

**STRATEGIC MARINE ECOSYSTEM RESTORATION
IN NORTHERN BRITISH COLUMBIA**

by

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ABSTRACT

Innovative methodology is developed for Back to the Future (BTF) restoration policy analysis to aid long-term strategic planning of ecosystem-based restoration in marine ecosystems. Mass-balance and dynamic ecosystem simulation models (Ecopath with Ecosim: EwE) are developed to represent the marine system of northern British Columbia as it appeared in 1750, 1900, 1950 and 2000 AD. Time series statistics are assembled for biomass and catch, incorporating local ecological knowledge (LEK) from community interviews and new estimates of illegal, unreported and unregulated (IUU) fishery catch. The dynamic behaviour of the historic models is fitted to agree with this time series information, when driven by historic catch rates and climate anomalies. Each historic period is evaluated in an optimal policy analysis for its potential to supply sustainable harvest benefits. Harvest benefits are quantified using socioeconomic and ecological indicators, including novel measures such as the Q-90 biodiversity statistic. Candidate goals for restoration are drafted based on these historic ecosystems. A new conceptual goal for ecosystem-based restoration is introduced, the optimal restorable biomass (ORB) that represents an optimized form of the historic ecosystems. It is structured to maximize sustainable harvest benefits, and to achieve a compromise between exploitation and the maintenance of historic abundance and biodiversity. Finally, restoration plans are drafted using a novel addition to Ecosim's policy search routine, the *specific biomass* objective function, which determines the pattern of fishing effort required to restore the depleted present-day ecosystem into one resembling a more productive ORB state. Cost-benefit analysis indicates that northern BC ecosystem restoration to an ORB state based on the 1950 ecosystem can deliver a rate of economic return, in terms of increased fisheries yields, that is superior to bank interest. The effect of fleet structure is paramount; reducing bycatch will greatly enhance the effectiveness of the fleet as a restoration tool. Restoration plans that sacrifice immediate fisheries profits tend to restore more biodiversity in a given amount of time, but a convex relationship between profit and biodiversity suggests there is an optimal rate of restoration.

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1 BACK TO THE FUTURE

The significant problems we face cannot be solved at the same level of thinking we were at when we created them.

Albert Einstein

Qu. Dukas and Hoffman (1979)

1.1 Introduction

For thousands of years, humans have been exploiting the seas for food. Paleoecological and archaeological evidence records the significant impacts that we have caused (Jackson *et al.* 2001). Fishing is thought to have become important to humans during the Upper Paleolithic period, 10 to 30 thousand years ago (Bar-Yosef, 2004), although fish may have contributed to our diet much earlier than that (Yellen *et al.*, 1995; Fiore *et al.*, 2004). From the earliest harpoons, nets and bone hooks, each advancement made in capturing fish must have opened up new habitats and new species to exploitation. But it was not until the development of industrial fisheries, less than 200 years ago, that the major depletion of marine systems began (e.g., Myers and Worm, 2003; Pauly *et al.*, 2005). With the advent of sail, steam and diesel powered boats, areas became accessible that were once out of reach. The end of the Second World War saw the modernization of fleets, including the addition of at-sea freezers, radar navigation, acoustic fish finders and other conveniences that increase catching power (Pauly *et al.*, 2002). The trend continues today with satellite navigation systems and communication networks that make fishing easier, safer and more efficient than ever before. Unfortunately, a step up in technology has proven to be a step down in the biodiversity and abundance of marine ecosystems (Pitcher and Pauly, 1998) (Fig. 1.1). The effect is cumulative. Globally, fisheries are in crisis (Pauly *et al.*, 1998; Myers and Worm, 2003).

Many factors can potentially contribute to the decline of fish stocks and the failure of fisheries. Climate is known to influence productivity of fish populations (e.g. Beamish *et al.*, 1995;

Polovina, 2005), and changes in climate may be related to long-duration environmental cycles that are poorly understood (Finney *et al.*, 2002). Other culprits like coastal development, land-based pollution and marine industries are also identified. In some cases, scientific error may contribute to fishery declines (e.g., Hutchings, 1996). However, it is overfishing that many scientists now believe has been the primary driver of fisheries collapse world-wide.

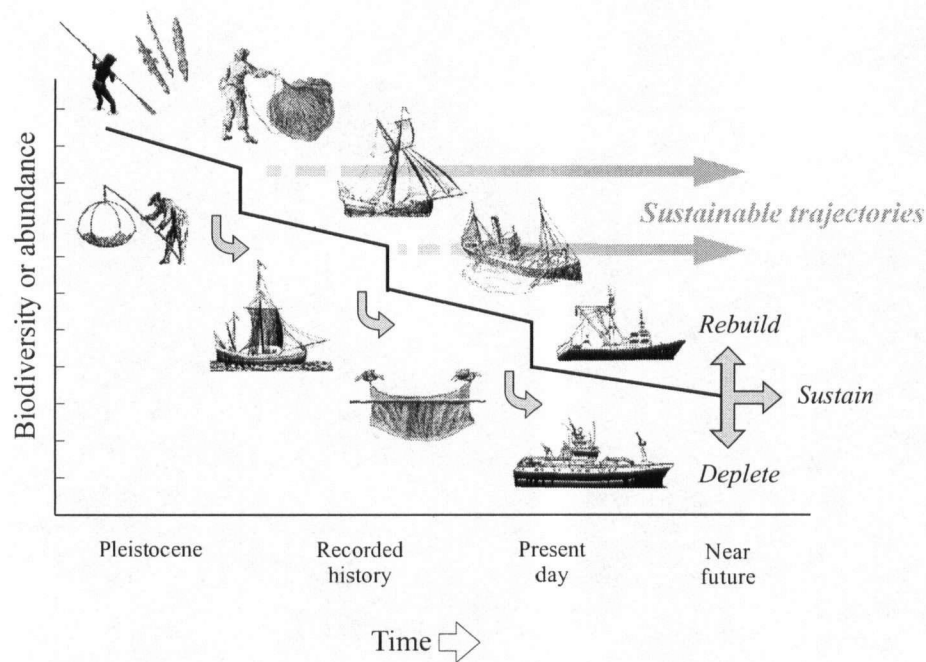


Figure 1.1 Biodiversity and species abundance decline caused by fisheries. The stepped downward line represents the serial depletion of marine ecosystems. Each fishing innovation, from simple harpoons to factory trawlers, opens up new species and habitats to exploitation. Horizontal arrows show sustainable use, which could have been achieved, in theory, at any level of ecosystem abundance. The three-way arrow shows policy options currently open to us. Modified from Pitcher and Pauly (1998).

Fishing overcapacity is viewed by some as the single greatest threat to sustainable fisheries (Mace, 1997; Gréboval and Munro, 1999; Ward *et al.*, 2001). Ludwig (1993) suggested that overcapitalization in the fishing fleet is driven by a dangerous bioeconomic ratcheting effect, where good fishing years encourage over-investment and bad fishing years demand government subsidies to keep the industry afloat. Compounding the problem, investors in the fishing industry may also expect a rate of return that is comparable to other types of enterprises, but cannot be supported sustainably by the natural growth rate of fish populations (Clark, 1973). Therefore,

overfishing is driven by complex social, economic and political factors. Any lasting solution will require cooperation across disciplines, and the commitment of many stakeholders groups.

To form this alliance we will need tools that can weigh the interests of all resource users, we will need to improve our understanding of human impacts on marine systems, and we will need to agree on a proper goal for fisheries management. Although sustainability is usually pursued as an explicit objective in regulated fisheries, repeated failures indicate that it is rarely achieved in practice (Ludwig *et al.*, 1993; Botsford *et al.*, 1997). When environmental conditions are favorable, sustainable fisheries may be achieved without careful restraints on human activities. But when climate turns against the interests of people, which may be increasingly of our own causing, our management systems need to operate according to strict precautionary principles. Sustainability is now too low of standard to aim for; we realize this when we look to the past as a reference point and understand the enormous benefits that a healthy ecosystem is capable of providing.

A new perspective on fisheries management

Many traditional target species have declined to only a fraction of their abundance prior to the industrialization of fisheries (Christensen *et al.*, 2003; Worm and Myers, 2003; Reid *et al.*, 2005; Rosenberg *et al.*, 2005; Ward and Myers, 2005). The public, and scientists as well, are generally unaware of the magnitude of the historic decline. It is perhaps because of Pauly's (1995) shifting baseline syndrome. He suggested that one's concept or perception of ecosystem abundance is based on a mental benchmark set at the beginning of the career. As the ecosystem is slowly degraded, each generation accepts a lower standard as the rule. This can apply to fisheries scientists as well as the general public. Considering the poor state of the oceans, it has been argued that the proper goal for fisheries management should not be to sustain current fish populations, but rather to restore them to historic levels (Pitcher *et al.* 1998; Pitcher and Pauly, 1998; Pitcher, 2001).

The Back to the Future (BTF) approach to restorative marine ecology offers a new perspective on what management objectives should be (see Pitcher, 2001a, 2004, 2005; Pitcher *et al.*, 1999,

2004, 2005; Ainsworth and Pitcher, 2005b). Under the BTF approach, an initial objective for any ecosystem-based restoration initiative should be to establish long-term goals for restoration. Candidate goals should be quantitatively evaluated for their potential to provide benefits to stakeholders and maintain ecological health.

Using ecosystem models, BTF simulates fishing of historic ecosystems to determine their long-term sustainable production potential. From this we can estimate what resource value has been lost due to human influences, and what a restored ecosystem might be worth to society. Fig. 1.2. shows a schematic illustration of the BTF concept. The symbols in Fig. 1.2 document many new and unconventional sources of information that must be relied upon to create whole ecosystem models of the past. Although there will be some aspects of historical ecosystems that are unknowable, multidisciplinary data on fish stocks and the environment can be used to form a picture of what the ecosystem looked like before heavy exploitation.

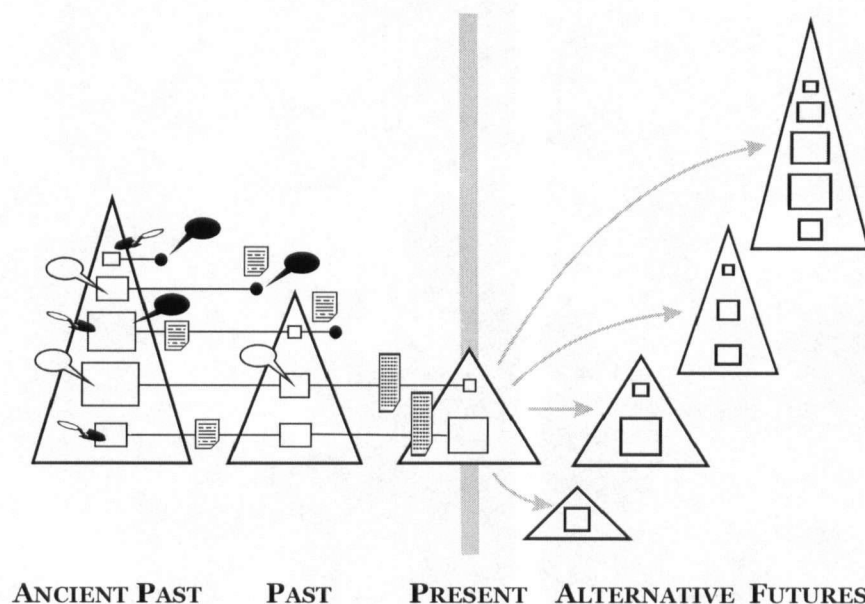


Figure 1.2 The Back to the Future approach to marine ecosystem restoration. Triangles represent trophic pyramids; height is directly related to biomass and internal connectance. Internal boxes show biomasses of representative species through time, with closed circles indicating extirpations. Ecosystems of the past contained longer trophic chains than they do now, greater biodiversity and predator biomass. The BTF approach advocates setting restoration goals based on historic ecosystems (right). Ecosystem models are constructed to evaluate various periods using historical documents (paper sheet symbol), data archives (tall data table symbol), archaeological data (trowel), the traditional environmental knowledge of indigenous Peoples (open balloons) and local environmental knowledge (solid balloons). Reproduced from Pitcher *et al.* (2004).

Historic ecosystems may hold special resonance with stakeholders as restoration goals if people can appreciate the long-term impacts that fisheries have had (Pitcher, 2000; Pitcher and Haggan, 2003). There may also be a scientific rationale for selecting restoration goals based on historic ecosystems. Because they existed, their relative species compositions may represent workable ecosystem goals, more so than an arbitrary design. If we can allow for environmental changes that have occurred since their time, then historic ecosystems can serve as an analogue for the future. The study of historic ecosystems can inform us as to what level of abundance and productivity can be expected from a natural system, given any constraints that regional oceanographic conditions impose.

Pitcher *et al.* (2004) imagined a bright future for marine fisheries, where the ecosystem is restored to something resembling a historic condition. They likened the reconstituted ecosystem to a *lost valley*¹, an untouched area as discovered in Sir Arthur Conan Doyle's "The Lost World". This *lost valley* offers humans a second chance to responsibly use the marine ecosystem. BTF asks the following questions: what might this *lost valley* look like, how might we sustainably harvest it, and what would be the costs and benefits of rebuilding to this goal? To answer these questions, a new methodology has been developed that makes use of the ecosystem simulator, Ecopath with Ecosim (EwE: Christensen and Pauly, 1992; Walters *et al.*, 1997; Christensen and Walters, 2004a).

A quantitative goal for ecosystem based approaches

Quantitative techniques are often called upon to help set safe removal rates. Numerical targets and reference points have been established to guide fisheries management and allow the responsible use of living marine resources. Historically, a widely used paradigm has been the maintenance of maximum sustainable yield (MSY) from fisheries. For a given stock size, it is the theoretical amount of catch that can be taken each year, under average environmental conditions, without influencing the abundance of the stock. The "puritanical philosophy" identified by Larkin (1977), to take only surplus stock production and forever maintain MSY

¹ The term *lost valley* was suggested by Prof. Daniel Pauly (Pitcher *et al.*, 2004).

once promised to solve all fisheries issues. Now people question whether MSY has ever been achieved in practice and whether it is achievable in theory (Larkin, 1977; Sissenwine, 1978; Punt and Smith, 2001). Amendments have been proposed to address the well-known inadequacies of MSY; for example, optimum sustainable yield (OSY: Roedel, 1975), maximum economic yield (MEY) and $F_{0.1}$ (see Hilborn and Walters, 1992). However, some question whether proper fisheries management is at all possible through a reductionist approach (Ludwig *et al.* 1993), which is the traditional mechanism of single species science. More and more, scientists are turning towards ecosystem based approaches in the hopes that a holistic view of ecosystem functioning will provide a better foundation for fisheries management.

Ecosystem based management (EBM) could benefit from a new objective reference point; one that considers the health and productivity of the ecosystem as a whole. Such a standard could do for EBM what indices like MSY, OSY, MEY and $F_{0.1}$ did for single species management – provide a quantitative policy goal that can potentially set the benchmark for sustainable use. This volume presents a new conceptual target for ecosystem based approaches. It is the optimal restorable biomass (ORB), an equilibrium biomass configuration for the ecosystem that maximizes sustainable harvest benefits, and is designed to meet specific criteria for ecosystem health.

ORB is calculated based on historic ecosystems. It is the species biomass vector, defining the relative abundance of each ecosystem component, that would naturally result after the long-term responsible use of historic ecosystems. Sidestepping the serial depletion of stocks witnessed in reality, it takes into account the activities of fisheries and determines the best compromise between maintaining historic abundance and diversity, while still providing for the needs of humans.

Mace (2001) pointed out that even if we could establish suitable goals for whole-ecosystem restoration, it is doubtful whether we would have the capability to manipulate the ecosystem into the desired state. The work presented in this volume offers a first step towards developing an integrated approach to management that can accomplish just that. Tools and techniques developed here for use with EwE models provide a strategic aid to help draft restoration plans

that would use selective fishing as a tool to manipulate the marine ecosystem, and ultimately restore it to some former level of abundance and productivity.

1.2 Ecopath with Ecosim

EwE provides a fresh tool to explore the complex interactions of marine organisms. To enable multi-sector fishery policy analysis, the competing effects of fisheries must be considered, as well as trophic interactions throughout the food web. Single species models, versatile and informative, are completely indispensable to whole ecosystem work, as they form the basis of our understanding for key ecosystem components. Nevertheless, they are limited in scope. Even traditional multi-species models can isolate and examine only a small number of interactions, and strict data requirements limit these analyses to well understood ecosystem components. Although ecosystem models offer no panacea, they can provide a new perspective on population dynamics and help us understand unintuitive processes. They can complement well-established analysis methods and provide an integrated overview of ecosystem functioning and the impact of fisheries. The mass-balance approach, in particular, makes it possible to construct a virtual ecosystem without the need for exhaustive supporting science.

Invented by Polovina (1984) and advanced by Christensen and Pauly (1992, 1993), Walters *et al.* (1997, 1998) and Christensen and Walters (2004a) among others, EwE is a mass-balance trophic simulator that acts as a thermodynamic accounting system. Summarizing all ecosystem components into a small number of functional groups (i.e., species aggregated by trophic similarity), the box model describes the flux of matter and energy in and out of each group, and can represent human influence through removals and other ways. There are now dozens of published articles that use EwE to describe ecosystems, qualify data, test hypotheses and demonstrate other applications (see review in Christensen and Walters, *in press*). EwE has been used in actual fisheries management, but to a limited extent. Reviews and criticisms of the EwE approach are provided by Fulton *et al.* (2003), Christensen and Walters (2004a), and Plagányi and Butterworth (2004).

Ecopath

The static model Ecopath (Polovina, 1984; Christensen and Pauly, 1992) implicitly represents all biotic components of the ecosystem. The model operates under two main assumptions. The first assumption is that biological production within a functional group equals the sum of mortality caused by fisheries and predators, net migration, biomass accumulation and other unexplained mortality. Eq. 1.1 expresses this relationship:

$$B_i \cdot (P/B)_i = Y_i + \sum_{j=1}^n B_j \cdot (Q/B)_j \cdot DC_{ij} + E_i + BA_i + B_i (P/B)_i \cdot (1 - EE_i) \quad \text{Equation 1.1}$$

Where B_i and B_j are biomasses of prey (i) and predator (j), respectively;

P/B_i is the production/biomass ratio;

Y_i is the total fishery catch rate of group (i);

Q/B_j is the consumption/biomass ratio;

DC_{ij} is the fraction of prey (i) in the average diet of predator (j);

E_i is the net migration rate (emigration – immigration); and

BA_i is the biomass accumulation rate for group (i).

EE_i is the ecotrophic efficiency; the fraction of group mortality explained in the model;

The second assumption is that consumption within a group equals the sum of production, respiration and unassimilated food, as in eq. 1.2.

$$B \cdot (Q/B) = B \cdot (P/B) + (1 - GS) \cdot Q - (1 - TM) \cdot P + B(Q/B) \cdot GS \quad \text{Equation 1.2}$$

Where GS is the proportion of food unassimilated; and TM is the trophic mode expressing the degree of heterotrophy; 0 and 1 represent autotrophs and heterotrophs, respectively. Intermediate values represent facultative consumers.

Ecopath uses a set of algorithms (Mackay, 1981) to simultaneously solve n linear equations of the form in eq. 1.1, where n is the number of functional groups. Under the assumption of mass-balance, Ecopath can estimate missing parameters. This allows modelers to select their inputs.

Ecopath uses the constraint of mass-balance to infer qualities of unsure ecosystem components based on our knowledge of well-understood groups. It places piecemeal information on a framework that allows us to analyze the compatibility of data, and it offers heuristic value by providing scientists a forum to summarize what is known about the ecosystem and to identify gaps in knowledge.

Ecosim

Ecosim (Walters *et al.*, 1997) adds temporal dynamics to turn the mass-balance model into a simulation. It describes biomass flux between groups through coupled differential equations derived from the first Ecopath master equation. The set of differential equations is solved using the Adams-Bashford integration method by default. Biomass dynamics are described by eq. 1.3.

$$\frac{dB_i}{dt} = g_i \sum_{j=1}^n f(B_j, B_i) - \sum_{j=1}^n f(B_i, B_j) + I_i - (M_i + F_i + e_i) \cdot B_i \quad \text{Equation 1.3}$$

Where dB_i/dt represents biomass growth rate of group (i) during the interval dt ;

g_i represents the net growth efficiency (production/consumption ratio);

I_i is the immigration rate;

M_i and F_i are natural and fishing mortality rates of group (i), respectively;

e_i is emigration rate; and

$f(B_j, B_i)$ is a function used to predict consumption rates of predator (j) on prey (i) according to the assumptions of foraging arena theory (Walters and Juanes 1993; Walters and Korman, 1999; Walters and Martell, 2004).

The principle innovation in Ecosim considers risk-dependant growth by attributing a specific vulnerability term for each predator-prey interaction. The vulnerability parameter is directly related to the carrying capacity of the system, and it describes the maximum increase in the rate of predation mortality on a given prey. A high value represents a top-down (Lotka-Volterra) interaction, a low value represents a bottom-up (donor-driven) interaction, and an intermediate value indicates mixed trophic control. Variable speed splitting enables Ecosim to simulate the

trophic dynamics of both slow and fast growing groups (e.g., whales/plankton), while juvenile/adult split pools allow us to represent life histories and model ontogenetic dynamics. A new multi-stanza routine in Ecopath (Christensen and Walters, 2004a) back-calculates juvenile cohorts based on the adult pool biomass and on life history parameters. The multi-stanza routine has replaced former the split-pool method; however, it was not available at the time of this work. As such, recruitment to juvenile stanzas in this model are determined by Ecosim using a Deriso-Schnute delay difference model (Walters *et al.*, 2000).

Ecospace

Ecospace (Walters *et al.* 1998) models the feeding interactions of functional groups in a spatially explicit way. A simple grid represents the study area, and it is divided into a number of habitat types. Each functional group is allocated to its appropriate habitat(s), where it must find enough food to eat, grow and reproduce - while providing energy to its predators and to fisheries. Each cell hosts its own Ecosim simulation and cells are linked through symmetrical biomass flux in four directions; the rate of transfer is affected by habitat quality. Optimal and sub-optimal habitat can be distinguished using various parameters such as the availability of food, vulnerability to predation and immigration/emigration rate. By delimiting an area as a protected zone, and by defining which gear types are allowed to fish there and when, we can explore the effects of marine protected areas (MPAs) and test hypotheses regarding ecological function and the effect of fisheries. Many authors have used Ecospace in this capacity (e.g., Walters *et al.*, 1998; Beattie, 2001; Pitcher and Buchary, 2002a/b; Pitcher *et al.*, 2001; Salomon *et al.*, 2002; Sayer *et al.*, 2005):

1.3 Northern British Columbia

Whenever viable fisheries are lost, communities and cultures that have traditionally relied on the sea can be impacted in deep and lasting ways. This is especially true when social and cultural values are tied closely to the sea. That is the case in northern British Columbia (BC). Fishery failures, such as the herring collapse of early 1960s, the Northern abalone collapse of the 1980s,

and the present decline of the salmon fisheries displaces workers, disrupts communities and sabotages a sustainable source of revenue.

This volume evaluates restoration scenarios for northern BC that would return the ecosystem to historic conditions of biodiversity and abundance. For this, I create ecosystem models of northern BC at various points in history: 1750, 1900, 1950 and 2000 AD. The models are described in Chapter 5. The 1750 model represents the marine ecosystem prior to contact by Europeans. It contains the most abundant array of marine fish and animals, although it does not represent an unexploited system since indigenous coastal human populations are thought to have relied on the sea to a great extent (Haggan *et al.*, *in press*; Turner *et al.*, *in press*). A model of 1900 represents the ecosystem as it appeared prior to the industrialization of fisheries, and before the advent of major advances in fishing technology such as steam trawlers. The 1950 model demonstrates what the ecosystem looked like during the heyday of the Pacific salmon fisheries, and before most major depletions of commercial fish populations. Finally, the present-day model, 2000, provides a contemporary representation of the ecosystem. It is from this vantage point that restoration plans are drafted.

Physical area

This study models the marine environment of northern BC, from the northern tip of Vancouver Island to the southern tip of the Alaskan panhandle, including the waters of Dixon Entrance (DE), Hecate Strait (HS) and Queen Charlotte Sound (QCS) (Fig. 1.3). It covers the shelf and continental slope, about 70,000 km² of ocean, using the same delineation as in Beattie (2001), including Department of Fisheries and Oceans (DFO) statistical areas 1-10. Oceanography of the region was described by Crean (1967), Thomson (1981), Ware and McFarlane (1989) and Crawford (1997).

The area roughly corresponds to the eastern region of the Coastal Downwelling Domain identified by Ware and McFarlane (1989). Water movement is influenced by the counterclockwise flow of the Alaska gyre, which creates a northeastern flowing Alaskan current year round. The Alaskan current enters QCS and extends northward along the coast into HS. In the south of the study region there is a transitional zone, where the clockwise flowing California Current diverges from the Alaska current and flows south. Coastal convergence occurs mainly on the west coast of Haida Gwaii and along the mainland shoreline of QCS and HS.

The shelf area is relatively shallow, more than two thirds of the total area is less than 200 m in depth. Three major gullies transect the continental shelf from east to west. Crossing HS and

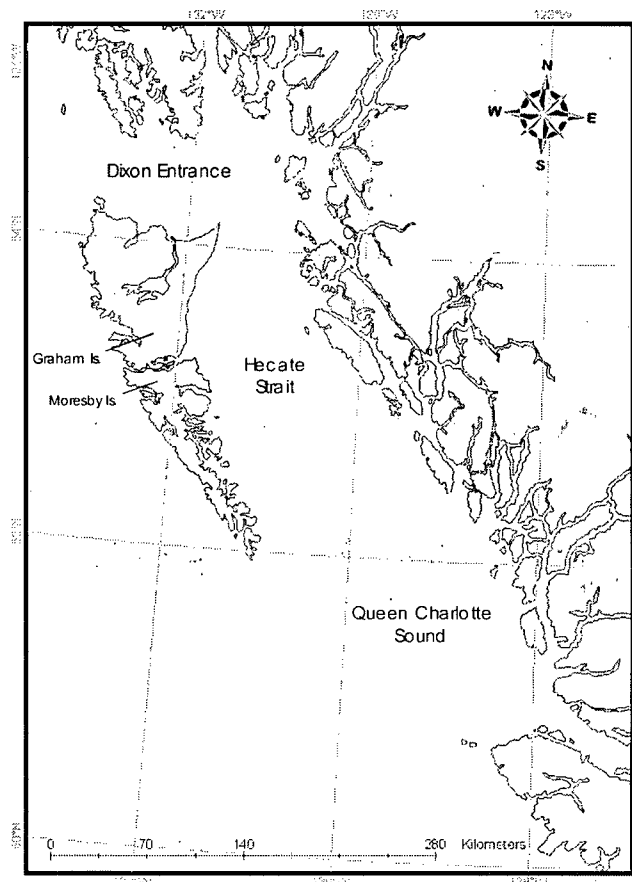


Figure 1.3 Northern BC study area. The study area includes the shelf waters of Queen Charlotte Sound, Hecate Strait and Dixon Entrance (DFO statistical areas 1-10).

terminating south of Moresby Island (S. Haida Gwaii) is the Moresby Trough. QCS is divided twice, by Mitchell's Trough in the north and Goose Island Trough in the south. The mainland coastline is rugged, with many islands and inlets.

Biological system

The waters of northern BC host a diverse marine biota. With the greatest human populations concentrated in the south of the province, the marine ecosystem of northern BC remains relatively intact compared to the Strait of Georgia and Southern BC. The complex coastline provides a range of habitats including rock, sand and mud flats, with various degrees of wave exposure. With its large expanse open to the Pacific Ocean, QCS offers an 'oceanic' habitat which is subject to oceanographic intrusions. HS and DE provide a more shallow and protected zone. Deep troughs and productive banks in QCS support large populations of rockfish, flatfish and demersal fish species. The coastal corridor is migrated annually by five salmon species, each an important commercial asset. Important nesting areas for seabirds, like cormorants (Phalacrocoracidae), gulls (Laridae) and auklets (Alcidae), are located along the coastal islands and on the mainland. Large kelp beds covering much of the coast provide habitat for juvenile fish, and support a large population of benthic invertebrates. Echinoderms like urchins, sea stars and sea cucumbers are common. Also occurring in the tidal and subtidal zones are massive beds of bivalves and barnacles. Seals and seal lions occur throughout northern BC. There are five species of pinnipeds: two Phocidae (true seals) and three Otariidae (eared seals). Cetacean species like killer whale (*Orcinus orca*), minke whales (*Balaenoptera acutorostrata*) and dolphins can occur throughout the year, and there are seasonal populations of migratory gray (*Eschrichtius robustus*) and humpback whales (*Megaptera novaeangliae*). Four hexactinellid sponge reefs in central QCS and HS are noted for their uniqueness and conservation utility (Conway, 1999; Sloan *et al.* 2001; Ardron, 2005).

Fisheries

Commercial fisheries in northern BC are conducted by seine boats, gillnetters, trawlers (or draggers), trollers, demersal traps, hook and line, scuba diving and other gear types. Commercial capture fisheries yielded a value of \$359 million in 2004 (DFO, 2004d), contributing a meagre 0.1% to the provincial gross domestic product. By comparison, recreational fisheries and their supporting industries contributed an estimated \$675 million in the same period, while aquaculture, mainly for Atlantic salmon (*Salmo salar*), contributed another \$287 million. Pacific salmon constitutes the most valuable component of the commercial catch. Salmon species include sockeye (*Oncorhynchus nerka*), pink (*O. gorbuscha*), chum (*O. keta*), chinook (*O. tshawytscha*) and coho (*O. kisutch*). The large majority of salmon captures is achieved by the seine net fishery, followed by gillnets and trollers. The halibut (*Hippoglossus stenolepis*) fishery is second in importance after the salmon species. It mainly uses longline gear and trolling methods. Herring roe purse seine fisheries and shrimp trawl and trap fisheries follow. Fisheries for rockfish, sablefish (traps), crabs, lingcod and other invertebrates also contribute to the coastal economy.

1.4 Structure of thesis

Chapter 1 summarizes the Back to the Future approach to restorative marine ecology. It describes the EwE ecosystem modeling software and provides background on the study area of northern BC. A new conceptual and quantitative target for ecosystem restoration is introduced: optimal restorable biomass (ORB).

Chapter 2 introduces quantitative indices used throughout this volume to evaluate harvest benefits in economic, social and ecological terms. Case studies are provided to demonstrate the use of these indices within the EwE framework and their application to restoration ecology. Economic valuation indices include net present value (NPV), calculated using conventional and intergenerational discounting approaches. A case study examines the Newfoundland cod collapse, and demonstrates that intergenerational valuation of fisheries resources advocates better maintenance of fish stocks than conventional valuation. An employment diversity index is

developed to help quantify social benefits of fishing, and a new ecological index is introduced to describe species biodiversity, the Q-90 biodiversity statistic. A case study compares biodiversity impacts of fishing policies using the Q-90 index across eight EwE models of the NE Pacific, and demonstrates that the index is invariant to model structure.

In Chapter 3, I describe the BTF community interviews conducted in northern BC, and explain the methodology used to turn the subjective comments of interviewees into a relative abundance trend for EwE functional groups. These trends help set the dynamics for data-poor functional groups in the northern BC models. The perceived changes in biomass are compared with stock assessment information and with preliminary model outputs as a diagnostic tool used to identify problem dynamics.

Chapter 4 quantifies illegal, unreported and unregulated (IUU) catch in BC for salmon and groundfish fleets using a new subjective methodology. It is part of a larger effort to establish reliable estimates of extractions, which can be used to tune the dynamic models. A timeline of BC fisheries is compiled that includes regulatory, technological and political changes likely to have affected the quantity of unreported catch. From this, a semi-quantitative Monte Carlo procedure provides estimates of IUU catch for each 5 year period between 1950 and 2000 based on qualified anchor points (i.e., real-world estimates of misreporting from the literature and expert opinion).

Chapter 5 explains the northern BC models in detail, including basic parameterization and all fitting procedures used to improve dynamic predictions. Climate factors are addressed that may have influenced observed ecosystem dynamics, and some generalizations are drawn concerning predator-prey trophic vulnerabilities: Ecosim's chief dynamic parameters. A novel procedure is introduced whereby the dynamics of ancient EwE models are tuned based on the fitted dynamics of more recent models. This assumes stationarity in density-dependent foraging tactics. It is demonstrated that this method improves predictions by the 1900 northern BC model over other common parameterization methods.

In Chapter 6, I demonstrate ORB as a new ecosystem-based goal for restoration. Various ORB restoration targets are determined from historic ecosystems. ORB equilibriums are structured to

maximize socioeconomic or ecological benefits in varying degrees, and a trade-off spectrum of available benefits is presented for each historical period. This analysis demonstrates what wealth we have sacrificed over the last 250 years through our unsustainable fishing practices, and it also demonstrates what restoration could be worth to stakeholders in monetary and non-monetary terms. New techniques are used to relate the geometry of the optimization response surface to various policy considerations. Uncertainties surrounding historic model parameter estimates are also considered in the ORB solutions through use of a Monte Carlo routine.

Chapter 7 describes a new procedure integrated into Ecosim that can be used to determine optimal restoration plans to transform the current ecosystem into a desired configuration. A new objective function called *specific biomass* is created for EwE's policy search routine, and possible restoration policies are evaluated that would turn the present-day depleted system into one resembling a more productive ORB state. Plans are tested that provide various degrees of continued harvest benefits during the restoration period. A cost-benefit analysis tests the economic feasibility of ecosystem restoration. A conservative approach to restoration is demonstrated to provide a better economic return than bank interest.

Chapter 8 offers conclusions on the strengths and weaknesses of this restoration approach, and suggests new avenues of research that could take this integrated methodology from theory into practice. A comparison is made between ORB as an ecosystem management target and Maximum Sustainable Yield (MSY), an analogous single species index. Criticisms of the BTF approach are addressed, and comment is made on the usefulness of EwE as a policy aid for restoration ecology. Finally, policy recommendations are provided based on the general conclusions of this study.

The appendices provide results and supporting information for each chapter. Appendix 2 includes a cost-benefit analysis of education, as an existing example of a multigenerational enterprise, that can be used to set the intergenerational discount rate for valuation of fisheries resources. Appendix 3 presents qualitative trends of relative abundance for EwE functional groups based on LEK information. Appendix 4 provides supporting materials for the IUU analysis, including a timeline of BC fisheries, a table summarizing influences in the rate of misreporting, as well as reported landings and species weights used to estimate the IUU trend.

Appendix 5 presents parameters used in the Ecopath and Ecosim models of northern BC for all time periods. Time series data for biomass and catch are presented; other information includes model outputs such as dynamic biomass and catch, and an equilibrium analysis of the present-day (2000) model. Appendix 5 also compares the present-day 2000 model with the one derived from the 1950 model (following a 50 year simulation). Appendix 6 first presents the parameters used in the policy optimization routine in Chapter 6, and then presents the results of the optimizations, listing harvest benefits of ORB ecosystems measured using various indices of harvest utility. Appendix 7 provides supporting information used to parameterize the policy search in Chapter 7, and biomass trajectories are presented for restoration plans that vary the speed of restoration and the level of sustained harvest benefits. Appendix 8 provides a spatial analysis of the consequences of marine protected area (MPA) zonation in northern BC. Various harvest strategies are analyzed for the National Marine Conservation Area (NMCA) surrounding Moresby Island in southern Haida Gwaii. Appendix 9 lists references cited in the appendices.

The published materials appearing in this thesis are presented in Table 1.1.

Table 1.1 Published materials appearing in this thesis. Articles in review or in preparation are available from this author (contact: c.ainsworth@fisheries.ubc.ca)

Thesis section	Subject	Reference	Journal or publisher	Description
Chapter 1	Back to the Future policy approach.	Pitcher et al. (2004)	American Fisheries Society Symposium	* Conference proceedings
Chapter 2	Application of Q-90 biodiversity statistic to EwE models of NE Pacific.	Ainsworth and Pitcher (<i>in press</i>)	Ecological Indicators	* Primary literature
	As above.	Ainsworth and Pitcher (2004b)	Fisheries Centre Research Reports	Grey literature
	Economic valuation techniques.	Ainsworth and Sumaila (2004a)	Fisheries Centre Research Reports	Grey literature
	Employment diversity index.	Ainsworth and Sumaila (2004b)	Fisheries Centre Research Reports	Grey literature
Chapter 3	Use of local ecological knowledge in ecosystem models.	Ainsworth and Pitcher (2005a)	Alaska Sea Grant	* Conference proceedings
	Interview methodology.	Ainsworth (2004)	Fisheries Centre Research Reports	Grey literature
Chapter 4	Estimation of IUU catch in BC	Ainsworth and Pitcher (2005c)	Fisheries Research	* Primary literature
	As above.	Ainsworth and Pitcher (2005d)	State of the Environment Reporting (MWLAP)	Internal government report
Chapter 5	Preliminary Northern BC models for 1750, 1900, 1950 and 2000 AD.	Ainsworth et al. (2002)	Fisheries Centre Research Reports	Grey literature
	As above.	Alcock et al. (<i>in prep.</i>)	McGill-Queens University Press	* Book
	Analysis of predator-prey vulnerabilities for northern BC models	Ainsworth and Pitcher (2004a)	Fisheries Centre Research Reports	Grey literature
Chapter 6	Evaluation of restoration goals based on ORB concept.	Ainsworth and Pitcher (2005b)	Alaska Sea Grant	* Conference proceedings
	As above.	Pitcher et al. (2005)	NATO Science Series IV: Earth and Env. Sciences.	Book
	Introduction of ORB concept.	Baker et al. (<i>in prep.</i>)	McGill-Queens University Press	* Book
	Trade-off analysis of ORB benefits.	Pitcher and Ainsworth (<i>in review</i>)	Proceedings of the 4th World Fisheries Congress	* Conference proceedings
	Policy search methods.	Ainsworth et al. (2004)	Fisheries Centre Research Reports	Grey literature
Chapter 7	Restoration policy optimization; introduction of <i>specific biomass</i> algorithm.	Ainsworth and Pitcher (<i>in review</i>)	Proceedings of the 4th World Fisheries Congress	* Conference proceedings
	Cost benefit analysis of ecosystem restoration to various ORB states.	Ainsworth and Pitcher (<i>in review</i>)	ICES Annual Science Conference Proceedings (2005)	* Conference proceedings
	Demonstration of ecosystem-based population viability analysis.	Pitcher et al. (<i>in review</i>)	ICES Annual Science Conference Proceedings (2005)	* Conference proceedings
Appendix 2	Intergenerational discounting case study: Newfoundland Northern cod collapse.	Ainsworth and Sumaila, (2005)	Canadian Journal of Fisheries and Aquatic Sciences	* Primary literature
	As above.	Ainsworth and Sumaila (2003)	Fisheries Centre Research Reports	Grey literature
Appendix 8	Spatial analysis of Gwaii Haanas NMCA zonation options.	Ainsworth (2004)	(Available from author)	Workshop proceedings

* Peer reviewed contribution

2 HARVEST POLICY EVALUATION TECHNIQUES

The prudent heir takes careful inventory of his legacies and gives a faithful accounting to those whom he owes an obligation of trust.

John F. Kennedy

State of the Union Address, 1961

2.1 Introduction

To evaluate economic, social and ecological benefits of harvest policies for the BTF procedure, I have adapted standard evaluation techniques and developed new ones for use with EwE models. The indices described in this chapter include an economic index, net present value (NPV) calculated under conventional and intergenerational discounting equations (Ainsworth and Sumaila, 2004a; Sumaila, 2004; Sumaila and Walters, 2005), a social utility index based on employment diversity (Ainsworth and Sumaila, 2004b) and an ecological index used to represent biodiversity (Q-90 statistic) (Ainsworth and Pitcher, 2004b; *in press*). Chapters 6 and 7 use these indices along with standard EwE outputs to compare candidate ecosystem goals for restoration, and evaluate the success of fishing plans to achieve those goals.

2.2 Economic index: Net present value (NPV)

Although benefits of marine ecosystem restoration may be measured in ecological and social terms, economic considerations will likely weigh heavily in determining the practicability of any long-term restoration agenda. The NPV term is used to summarize the economic success of harvest plans because it condenses the flow of future benefits into a single expression, while introducing a time component through discounting that reflects the preference of an investor for immediate benefit and delayed payment. The conventional discounting NPV term weights immediate harvest benefits heavily, but the present value of benefits to be received far in the future is reduced exponentially with time. Under the intergenerational discounting approach

(Sumaila, 2004; Sumaila and Walters, 2005), future benefits are discounted less, and the welfare of future generations is considered explicitly in the present value term.

Conventional discounting

Under the conventional model discounting, the flow of fishery benefits is summarized in the NPV term using the expression in eq. 2.1.

$$NPV = \sum_{t=0}^T (d^t \times NB_t) \quad \text{Equation 2.1}$$

Where NB is net benefit accruing in year t ; d is the discount factor defined in eq. 2.2,

$$d = \frac{1}{(1 + \delta)} \quad \text{Equation 2.2}$$

Where δ is the discount rate.

Intergenerational discounting

The intergenerational discounting equation considers a continuous interlacing of generations, where the discounting of future benefit is countered each year by the addition of $1/G$ stakeholders (G is generation time). These new entrants bring with them a renewed perspective on future earnings, partially resetting the discounting clock. The equation requires a standard annual discount factor (d) and a discount factor to evaluate benefits destined for future generations (d_{fg}). NPV is represented as in eq. 2.3.

$$NPV = \begin{cases} \sum_{t=0}^T NB_t \left(d^t + \frac{d_{fg} \cdot d^{t-1}}{G} \cdot \left[\frac{1-\Delta}{1-\Delta'} \right] \right) & \text{if } \delta \neq \delta_{fg} \\ \sum_{t=0}^T \frac{NB_t}{(1+\delta)^t} \cdot \left(1 + \frac{t}{G} \right) & \text{otherwise} \end{cases} \quad \text{Equation 2.3}$$

G is assumed to be 20 years, the average age at which a Canadian woman has her first child, and

$$\Delta = \frac{d_{fg}}{d} \quad \text{Equation 2.4}$$

The conventional approach to discounting will favour fishing policies that provide immediate benefits to stakeholders, while the intergenerational approach will assign a higher NPV to policies that spread out benefits over several decades.

The need for a new resource valuation method

In cost-benefit analysis (CBA), standard discounting is often unable to sanction long-term environmental policies that fulfill the frequently stated mandate to provide for the needs of future generations (e.g., DFO, 2001; EC, 2002). Scaling down the value of future benefits exponentially through time ensures that immediate costs will outweigh far-off benefits at any practicable level of discounting, so that only myopic policies can result (Clark, 1973; Sumaila, 2001; 2004). In valuing the stream of benefits from a fisheries resource, use of conventional discounting may lead to early profit taking at the expense of sustained productive potential. Evidence of this type of ‘front-loading’ of fisheries benefits is clear in the harvest record of Northern cod (*Gadus morhua*) in the years before the 1992 collapse.

Appendix 2.1 presents a case study on the Newfoundland cod fishery that suggests conventional valuation of fishery resources may have contributed to the decline and collapse of the Atlantic cod fishery. The case study also demonstrates that intergenerational valuation of fisheries resources could make long-term conservation an affordable prospect. The discount rates I use to

evaluate fishery benefits are based on a CBA of education (Appendix 2.2). Schooling of children serves as an example of an existing multigenerational investment. By applying the apparent discount rate that people use to value the education of their children, I implicitly account for a variety of non-monetary benefits which could also apply to resource conservation.

2.3 Social utility index: Employment diversity

Ainsworth and Sumaila (2004b) used an employment diversity index to evaluate harvest plans after the methodology of Attaran (1986). Based on the Shannon's entropy function (Shannon and Weaver, 1949), this measure describes the diversity of employment across fishing sectors. The entropy function is defined as in eq. 2.5:

$$D(E_1, E_2, \dots, E_N) = - \sum_{i=1}^n E_i \log_2 E_i \quad \text{Equation 2.5}$$

Where n is the number of (possible) fishing sectors active in the ecosystem, and E is the proportion of total employment that is located in the i^{th} fishing sector.

The measure is normalized across sectors with respect to their maximum possible diversity so that $D(E_1, E_2, \dots, E_n)$ ranges from 0 to 1, as in eq. 2.6.

$$D(E_1, E_2, \dots, E_n) = \left(- \sum_{i=1}^n E_i \cdot \log_2 E_i \right) / \text{MAX}(D(E_1, E_2, \dots, E_n)) \quad \text{Equation 2.6}$$

When $D = 0$, this indicates that all fishing activity is concentrated in a single sector; $D = 1$ indicates the maximum possible employment diversity with all sectors contributing equally to employment (all E_i are equal).

Application to Ecosim

A VB algorithm uses this descriptor to assess the annual employment diversity of the dynamic 50-year harvest schedule for each optimal policy suggested by the EWE policy search routine. Beginning with Ecosim's output CSV file, total value per gear type is calculated as the sum of all functional group landings, multiplied by gear-specific prices (Chapter 5; Appendix Table A5.1.6). Total value per gear type is converted to relative number of jobs using an estimated "jobs per catch value". It is considered equal for all fleets, so employment is proportional to landed value. Employment per sector (E_i) is then calculated as a fraction of total employment.

2.4 Ecological indices

Although the commercial value of fishing a restored ecosystem may offset the costs of rebuilding (Chapter 7), any practical restoration agenda will need to include ecological criteria for ecosystem improvement. A range of ecological indicators is useful for forecasting non-monetary benefits in fishing scenarios, and many have been developed or adapted for use with EWE models.

It can be difficult to define appropriate indices to summarize ecosystem model outputs (Fulton *et al.*, 2003), but considering the generic nature of EWE, its wide availability and comparatively simple implementation, there is a need to develop standardized outputs that can help users interpret ecosystem effects of experimental harvest scenarios. Ecosystem modelers have begun to realize that functional group aggregation styles and other nuances of model structure can have significant impacts on the dynamic predictions (Fulton *et al.*, 2003; Pinnegar *et al.*, 2005). Output indices therefore need to be robust and deliver consistent results despite subjective variations in model structure.

Ecosystem modelers must often make judgments on the applicability of imperfect data, but there are also fundamentally subjective components in EWE. 1) Functional groups of species are aggregated depending on the objectives of modeling, fishery and policy targets and availability of data. 2) When time-series data are unavailable for fitting, flow parameters may be set according to rules of convention. 3) The model diet matrix is usually based on incomplete and

imprecise data, and arbitrary manipulation of the matrix may be required to achieve mass-balance. Some attempts have been made to standardize the EwE model construction process (e.g., automatic mass-balance: Kavanagh *et al.*, 2004; semi-automated data retrieval from Fishbase: Froese and Pauly, 2005).

Existing EwE outputs

Ecological indicators automated in EwE include the Finn cycling index (Finn, 1976), indices relating to energy and primary production required (Odum, 1988; Christensen and Pauly, 1993), trophic flow indices (Ulanowicz, 1986), resource niche overlap (based on Pianka, 1973), system omnivory index (Pauly *et al.*, 1993), fishing-in-balance index (Pauly *et al.*, 2000), mixed trophic impacts (Ulanowicz and Puccia, 1990), among other system state and trophodynamic indicators. Indicators developed for the BTF approach include an ecosystem resiliency index based on information theory (Heymans, 2004), a fuzzy logic algorithm to estimate local extinction risk based on fish life history parameters (Cheung and Pitcher, 2004; Cheung *et al.* 2005) and a biodiversity statistic, Q-90, which is described here (also see Ainsworth and Pitcher, *in press*).

Q-90 biodiversity statistic

The Q-90 biodiversity statistic is a variant on Kempton's Q index (Kempton and Taylor, 1976) that has been adapted for use with EwE, where taxonomic species are grouped into aggregate biomass pools of functionally similar organisms. When used in conjunction with other indicators, the Q-90 index offers a useful method to evaluate consequences of alternative fishing plans, track the effect of climate fluctuations and changes on biodiversity, estimate the non-consumptive value of ecosystems, and generally inform the ecosystem-based approach to marine science. Although ecological indicators of all varieties are of scientific interest, biodiversity holds special appeal to the public and is often addressed directly by policy - even though the appropriate scientific definition may not be made explicit (Harper and Hawksworth, 1994; Hamilton, 2005). In this section, I refer to biodiversity as organismal diversity at the level of species functional groups.

Definition

Kempton's Q index describes the slope of the cumulative species abundance curve (Fig. 2.1). As applied here, each functional group in the EwE model represents one 'species', and the biomasses of these groups, sorted into bins, serves as a proxy for the number of individuals in that species. Kempton and Taylor (1976) suggested using the inter-quartile slope of the species abundance curve in order to avoid problems arising from the inclusion of tails, which, in field sampling, may be long and include a high number of low-abundance species. In applying this methodology to Ecosim, tails are less of a problem since modelers do not normally represent a large number of low abundance functional groups. I therefore used the slope between the upper and lower 10-percentiles rather than quartiles.

The Q-90 statistic is defined as in eq. 2.7.

$$Q_{90} = 0.8S / [\log(R_2/R_1)]$$

Equation 2.7

Where S is the total number of functional groups in the model; R_1 and R_2 are the representative biomass values of the 10th and 90th percentiles in the cumulative abundance distribution.

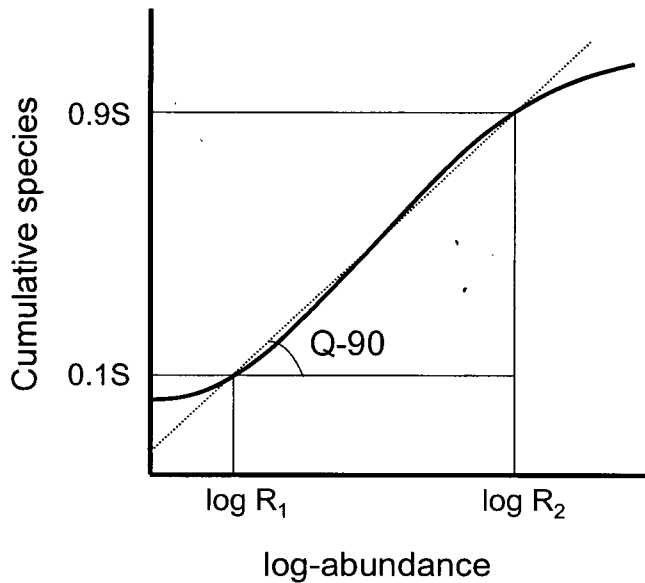


Figure 2.1 Q-90 statistic definition. S is number of functional groups in reference model; R_1 and R_2 are lower and upper 10-percentiles of the species abundance distribution. Modified from Kempton and Taylor (1976).

The 10th and 90th percentiles are determined by eq. 2.8 and eq. 2.9, respectively,

$$\sum_1^{R_1-1} n_R < 0.1 \cdot S \leq \sum_1^{R_1} n_R \quad \text{Equation 2.8}$$

$$\sum_1^{R_2-1} n_R < 0.9 \cdot S \leq \sum_1^{R_2} n_R \quad \text{Equation 2.9}$$

Where n_R is the total number of functional groups with abundance R .

Magurran (1988) describes the qualities of Kempton's index that make it well suited to this application. Kempton's index is not dependent on the assumption of a particular species abundance model, which makes it generically applicable to a wide variety of ecosystem types. It is not biased by very abundant or very rare species, and this can be advantageous if there are highly aggregated functional groups, as is sometimes the case with data-poor models. It expresses both species richness and evenness, which allows it to discriminate ecosystem effects among harvest plans (since exploitative fishing strategies can result in depletions or extirpations), while also capturing changes in the ecosystem that occur outside of harvested functional groups. In field studies, Kempton's index is robust against changes in sample size if very small samples are avoided, but this is not critical with EwE models since the entire ecosystem is represented explicitly or implicitly.

The following case study evaluates the effect of fisheries on ecosystem biodiversity, and demonstrates that the Q-90 statistic delivers consistent results regardless of model structure.

2.5 Q-90 case study: NE Pacific ecosystems

I use the Q-90 statistic to evaluate biodiversity after 25 years of fishing for eight ecosystem models of the northeastern Pacific under a variety of fishing plans. I test the ability of Q-90 to respond to fishing influence on the ecosystem, and compare predictions made using simple and complex ecosystem models. By choosing similar shelf ecosystems, biodiversity predictions

should be comparable across models. Any real differences in biodiversity among the ecosystems should be minimized so that we can examine the effects of model structure on the index.

Applying index to Ecosim output

Using a Visual Basic algorithm, a user-defined number of bins is established that represents the complete range of functional group biomasses. The biomass of each functional group is then sorted into its appropriate bin as a count; this serves as a proxy for the number of individuals in that group. Bins may be linear or logarithmic. The Q-90 index is the slope of the cumulative species abundance curve is determined between the 10- and 90-percentiles; the Q-90 value may be plotted for each year in the simulation.

At present, EwE does not permit absolute extinctions; it returns a low but non-zero biomass value for critically depleted groups. Therefore, every fishing scenario at its conclusion will contain the same number of functional groups as the base model. To increase the sensitivity of the index to group depletions, a filter is passed over group biomasses for each year of the simulation. If the biomass of a given functional group falls below a reference value, that group is omitted from the Q-90 calculation, reducing the overall biodiversity score.

In previous applications of this index, the depletion filter threshold has been set as an arbitrary 60% of the unfished biomass (B_0) and pristine biomasses represented in models of ancient ecosystems have been used as a proxy for B_0 (e.g., Ainsworth and Pitcher, 2005b; also see Chapter 6). Setting a high threshold makes the index more sensitive to group depletions; the Q-90 value therefore drops off quickly as fishing plans tend towards heavy exploitation and the index provides greater discrimination between conservative and exploitative fishing plans. The filter threshold may be reduced when evaluating severely depleted ecosystems; alternatively, one may set the threshold at a fraction of the baseline biomass.

Methods

Using eight EwE models of present-day ecosystems in the northeastern Pacific (Table 2.1), I compare the effects of three simple fishing policies on biodiversity: a reduction in fishing mortality to one-half the model baseline (0.5 F), baseline fishing mortality (1 F) and a five-times increase in fishing mortality (5 F). Baseline represents an estimate of current real-world fishing mortality. In lieu of biomass estimates for unfished populations (i.e., that correspond to the species group aggregation style used by the original modelers), the depletion filter is set here as a proportion of baseline group population size for all simulations. If groups fall below 80% of their initial biomass, they are removed from the Q-90 calculation.

Table 2.1 Eight EwE models of the NE Pacific.

Abbreviation	Model area	# of groups	Reference
WCVI	West Coast Vancouver Is.	15	Pauly <i>et al.</i> (1996)
SNBC	Northern BC – small model	26	Ainsworth, C. (<i>unpublished manuscript</i>) ¹
SOG	Strait of Georgia	27	Dalsgaard <i>et al.</i> (1998)
ALU	Aleutians	40	Heymans (2005)
HEC	Hecate Strait	50	Beattie (2001)
PWS	Prince William Sound	51	Okey and Pauly (1999); Okey and Wright (2004)
LNBC	Northern BC – large model	53	Ainsworth <i>et al.</i> (2002)
NCC	Northern California Current	65	Field (2004)

¹ Contact: c.ainsworth@fisheries.ubc.ca

Results

Long-term fishing simulations show a relationship between biodiversity maintenance and the overall level of fishing mortality applied. Fig. 2.2 shows Q-90 biodiversity predictions from the Ecosim model of the Northern California Current (Field 2004). As we increase extractions from the ecosystem, biodiversity is sacrificed. Under the exploitative fishing policy described by the (5 F) scenario, there is an initial 50% drop in ecosystem biodiversity.

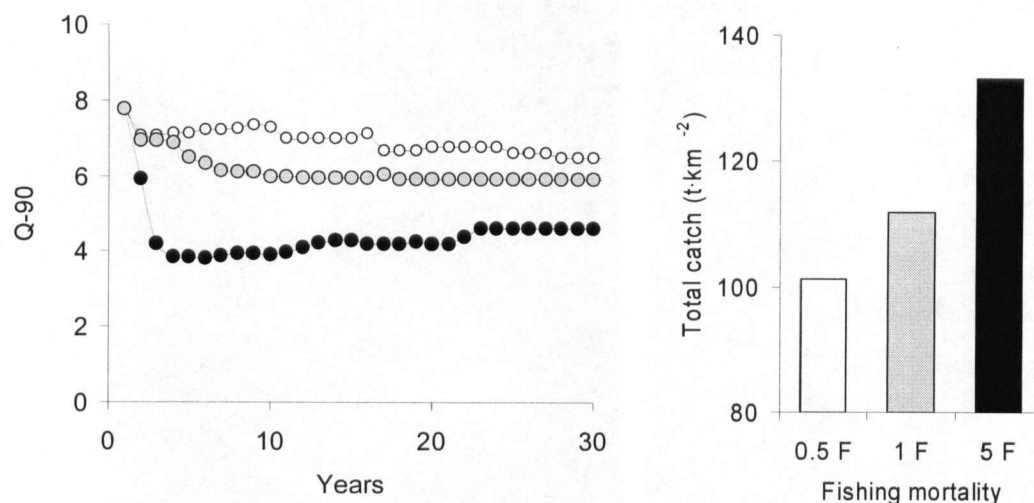


Figure 2.2 Dynamic ecosystem biodiversity (Q-90) of three example Ecopath with Ecosim simulations. A.) White circles show reduced fishing mortality from model baseline (0.5 F); grey circles show baseline fishing mortality (1 F); black circles show increased fishing mortality (5 F). B.) Bar graph shows total catch for these policies. Model of Northern California Current ecosystem (Field, 2004).

Fig. 2.3 compares baseline biodiversity among Ecopath models of the NE Pacific constructed by various authors using independent group aggregation criteria. The absolute value of the Q-90 statistic increases in direct relation with the total number of functional groups. The scatter around the trendline represents differences in functional group aggregation style and real ecological differences, although I have tried to minimize this factor by using models of similar ecosystems. Fig. 2.4 suggests that the relative change in the Q-90 statistic is not dependant on model complexity. However, model complexity itself can affect dynamic function if functional groups are over- or under- aggregated and a key ecological interaction is misrepresented (Fulton *et al.*, 2003). In that case, the Q-90 index will report the errant model behaviour. Because of this, we may expect a small degree of variation around the trend line in Fig 2.3, owing to inherent behavioural differences between models of varying complexity. However, Q-90 measurements for complex models (containing many functional groups and interactions) should be resistant to the compounded data uncertainty (see Håkanson, 1995) if errors surrounding the slope line in Fig 2.3 tend to cancel.

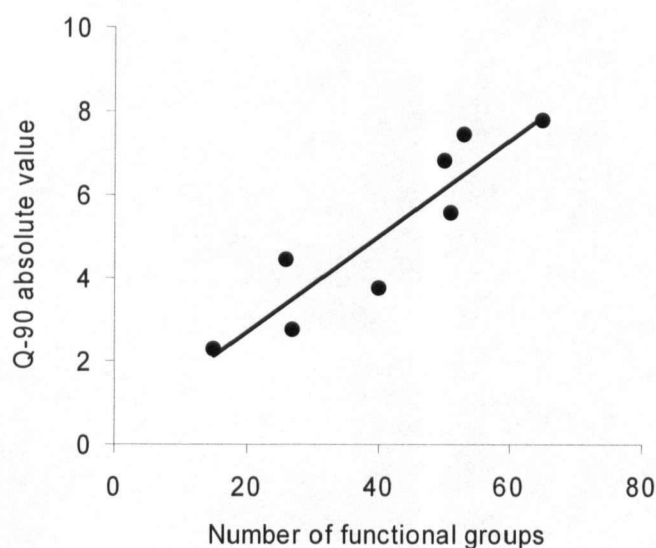


Figure 2.3 Absolute Q-90 value at baseline (year zero) for eight northeastern Pacific Ecopath models. The simplest Ecopath model represents the ecosystem using only 15 functional groups, while the most complex model uses 65 functional groups.

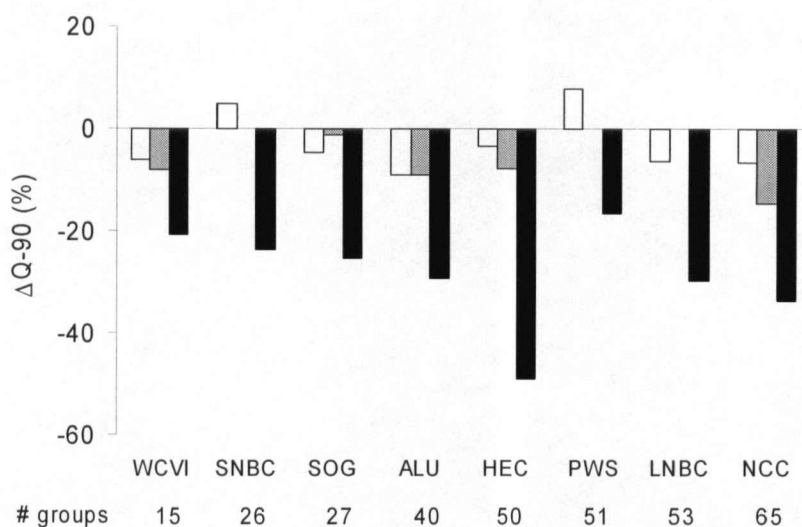


Figure 2.4 Change in Q-90 index after 30 years of fishing for eight EwE models of the NE Pacific. From left to right, models increase in number of functional groups. White bars show reduced fishing mortality from model baseline (0.5 F); grey bars show baseline fishing mortality (1 F); black bars show increased fishing mortality (5 F). West Coast Vancouver Is. (WCVI); small-Northern British Columbia (SNBC); Strait of Georgia (SOG); Aleutian Islands (ALU); Hecate Strait (HEC); Prince William Sound (PWS); large-Northern British Columbia (LNBC); Northern California Current (NCC).

Fig. 2.4 also shows the effects of fishing on the biodiversity of the ecosystem. At five times model baseline fishing mortality, every EwE model predicts a drop in biodiversity over 30 years. Except for SNBC, LNBC and PWS, which were designed to be steady state under baseline fishing mortality, all models predict a biodiversity decline under baseline fishing mortality. Several models suggest that even halving the exploitation rate will not prevent biodiversity from declining over the long-term. However, the fishing scenarios tested are simplistic because all assume a constant level of fishing mortality without regard to changing stock size and the fishing rates tested also assume a uniform change in fishing mortality across all gear sectors.

Index resolution

The Q-90 statistic tends to change in a step-wise fashion with dynamic biomass predictions. Models containing many functional groups allow the index to resolve more precise changes in species composition, but models containing fewer functional groups tend to produce coarse changes in the biodiversity index over time reflecting only large-scale changes in species composition. Resolving power of the index is therefore reduced in models containing fewer functional

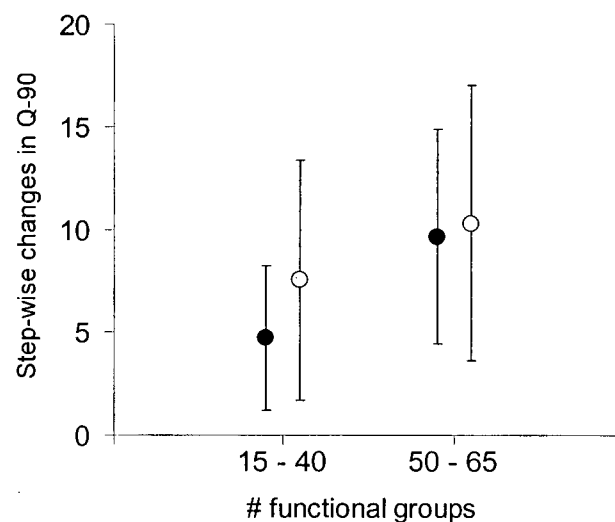


Figure 2.5 Q-90 sensitivity to changes in system biomass structure.

Q-90 sensitivity is compared in small (< 40 functional groups) and large models (> 50 groups). Y-axis shows mean number of step-wise changes in Q-90 value (i.e., resolving power) for a standard set of harvest simulations (30 year simulations at 0, 0.5, 1 and 5 times baseline fishing mortality). Closed circles show logarithmic bins, open circles show linear bins (error bars; 1 SD).

groups (Fig. 2.5). A one tailed Student's *t* test indicates that resolving power is significantly less for small models (< 40 functional groups) than large models (> 50 groups) ($p < 0.05$). Linear

bins provide better resolution for small models than logarithmic bins (t test; $p < 0.01$), but logarithmic bins produce less variable results overall (F test; $p < 0.05$).

Fig. 2.6 shows application of the depletion filter at 30%, 50% and 80% of baseline functional group biomass. A high filter threshold causes functional groups to fall out of the Q-90 calculation, and increases the sensitivity of the index to ecosystem changes. Under high depletion filter thresholds, linear bins may be slightly better at resolving biodiversity changes than logarithmic bins (t test; $p = 0.104$).

Discussion

This application of Kempton and Taylor's (1976) index to ecosystem models considers both evenness and richness in the biodiversity score. Although most ecological studies determine biodiversity based on occurrence and abundance of taxonomic species ('speciosity'), the number of functional groups in a EwE model is fixed and species-level population changes are not considered in the dynamics unless those species are explicitly represented. The method introduced here therefore provides an approximation to the original Kempton index, which was developed for field studies.

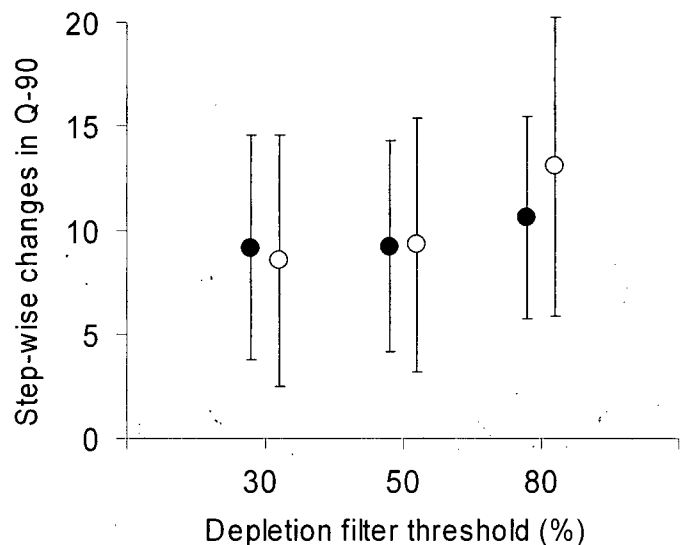


Figure 2.6 Q-90 sensitivity to changes in ecosystem structure using three depletion filter thresholds. Thresholds are set at 30%, 50% and 80% of baseline functional group biomass. Y-axis shows mean number of step-wise changes in Q-90 value (i.e., resolving power) for a standard set of harvest simulations (30-year simulations at 0, 0.5, 1 and 5 times baseline fishing mortality). Closed circles show logarithmic bins, open circles show linear bins (error bars; 1 SD).

Evenness can be represented in the ecosystem models, with biomass serving as a proxy for the number of individuals in each functional group. Under some circumstances this proxy could produce a bias; for example, if the average weight of animals changes suddenly as a result of fishing, as new technologies are introduced or in response to market influences. When comparing ecosystem models of different time periods, evolutionary changes in response to fishing could also cause a bias. Calculating richness is less straightforward. Since the number of model functional groups is fixed, the depletion filter is used to drop groups from the calculation and the total number of functional groups active in the calculation is therefore analogous to species richness. By setting a high depletion filter threshold we increase the contribution of species richness to the overall biodiversity score, but without the filter the index solely represents evenness.

Eliminating groups from the computation with the filter increases the sensitivity of the index to depletion events or effects and reduces the overall Q-90 value. However, as functional groups are removed, the remaining biodiversity calculation is based on fewer groups and the ability of the index to recognize small changes in biodiversity is compromised. I suggest using a high threshold to increase sensitivity of the index for models containing many functional groups, which can stand to lose a few from the calculation, or to exaggerate small differences in ecosystem biodiversity when comparing similar models or fishing plans.

The algorithm could be adapted to work with any static or dynamic multispecies or ecosystem model that represents species biomass in aggregated functional groups; see Fulton *et al.* (2003), Hollwed *et al.* (2000) and Whipple *et al.* (2000) for reviews of multispecies and ecosystem models. Model dynamics do not need to be based on trophic flows, but the biomass of functional groups must be accessible. Models which are primarily oceanographic or biogeochemical likely could not benefit; nor could EwE models that use nutrients as the currency of group exchange instead of biomass (e.g., Watkinson, 2001).

The Kempton Q index is now automated in EwE V5.1 and is available as a dynamic output for simulations (Christensen and Walters, 2004b). However, the integrated version is not exactly as described here. It considers only high trophic level functional groups ($> TL\ 3$), it uses the inter-quartile slope of the cumulative abundance curve rather than 90-percentiles. It can also

accommodate only linear species biomass bins, and as it does not employ a depletion filter it mainly serves as an indicator of biodiversity evenness.

Contribution to ecosystem studies

The use of ecological indicators is recognized as a critical component of EBM (e.g., FAO, 2003; Cury *et al.*, 2005; Garcia and Cochrane, 2005), although firm ecological theory is needed to relate changes in ecological indices to proper remedial management actions (Hall and Mainprize, 2004). As the relatively new field of ecosystem modeling continues to advance, facilitated by an increase in inexpensive computing power and the current drive towards ecosystem-based approaches in marine systems (Link, 2002), standardized indices will make ecosystem models tools that are more effective toward understanding fisheries and climate effects on marine communities.

Not only can ecosystem models be used to evaluate potential repercussions of fishing on non-target organisms, broad indicators which describe the state of the natural environment may hold special resonance with the general public (Rogers and Greenway, 2005); and public buy-in is critical since fishery stakeholders become a far more encompassing group once the entire marine ecosystem is factored in to management decisions.

The next chapter will summarize work done with communities in northern BC. Community members helped evaluate candidate restoration goals and suggested fisheries that could be used to harvest a restored ecosystem. Through interviews, they provided local ecological knowledge to supplement scientific information and help satisfy the vast data requirements of the ecosystem models used in BTF research.

3 COMMUNITY INTERVIEWS

All our knowledge has its origins in our perceptions.

Leonardo de Vinci

Qu. E. MacCurdy (1954)

3.1 Introduction

In modeling whole marine ecosystems, data deficiencies become especially apparent among species that hold no commercial appeal. Stock assessment records exist for only a small minority of species so modelers must borrow parameters from other ecosystems, or rely on guesswork. Although EWE grants modelers some reprieve by automatically estimating biomasses of data-poor groups based on the assumption of mass-balance, there is a clear need to reduce uncertainty in our estimates by incorporating supplemental information, particularly for historic ecosystems. Local ecological knowledge (LEK) held by fishing community members is one such resource.

LEK can be used to fine-tune static Ecopath models, to confirm dynamic Ecosim function, or to inform us how the ecosystem might have been structured decades ago - before time-series data began for most species. LEK therefore holds obvious application for BTF, which seeks to quantify ecosystem changes over time. The key step in adapting LEK to our modeling needs comes in producing a quantitative data series from qualitative accounts. This section describes how that was done for the northern BC models, and how the LEK trends are used to improve dynamics in the northern BC models. I also compare LEK trends with stock assessment in the hope that fishers' perceptions can help establish criteria by which we can assess the quality of scientific data - by challenging it with an independent authority and identifying where fishers' perceptions depart from the scientific understanding. Interview methods used in this chapter are published in Ainsworth (2004); results are in Ainsworth and Pitcher (2005a).

3.2 Methods

Interviews

Under approval of the University of British Columbia Ethical Review Committee, workers from the Fisheries Centre² interviewed forty-eight community members from the Prince Rupert region and Haida Gwaii, BC in two community workshops in 2002 (Pitcher *et al.*, 2002b; Pitcher, 2004). The processed anonymous data is searchable online at [www.fisheries.ubc.ca/projects/btf/]. Interviewees represented a broad cross-section of commercial, recreational and aboriginal fishers as well as processors and others who are familiar with the marine system in Hecate Strait, Dixon Entrance and Queen Charlotte Sound. As the aim was to improve the northern BC models, participants were not selected randomly; snowball sampling was used to find the most knowledgeable contributors as recommended by partners and participants.

One hundred and eighteen flashcards of marine mammals, birds, fish and invertebrates were shown to each interviewee. LEK information recorded included species population changes, fisheries interactions and spatial information - such as animal aggregations and seasonal movements. These data along with career and demographic information were processed to ensure anonymity, and entered into the BTF Historical and Interview Database (Erfan, *in press*):

Creating a time-series of relative abundance

Respondents were asked whether the abundance of marine creatures had increased, remained the same or decreased during their careers. This method assumes that respondents made implicit allowance in their answers for any changes in catchability arising from new methods or fishing technology. To create a numerical trend, an interviewee's comment of increase, stable or decrease is assigned the numerical value of +1, 0 or -1, respectively. Every year that the respondent fished receives one numerical 'vote' for that organism. Summing votes from all

² Dr. Tony Pitcher (P.I.), Dr. Ussif Rashid Sumaila, Dr. Sheila Heymans, Dr. Melanie Powers, Nigel Haggan, Russ Jones (Haida Fisheries Council), Eny Buchary, Cameron Ainsworth, Pablo Trujillo, Louisa Wood, Richard Stanford, Erin Foulkes and Aftab Erfan.

respondents, the annual total is assumed to indicate the average perception for that year. A value greater than zero therefore indicates that the fishers perceived an increase in biomass during that period, while a value less than zero indicates a perceived decline. The resulting time-series provides an index of the rate of change for each organism, which is converted into a running total to serve as a proxy for relative abundance.

Data trends for organisms are compiled into Ecopath functional groups (see Chapter 5 for group descriptions). Some functional groups include multiple species, so I assume that the abundance trend of the group closely follows the species mentioned most often by respondents. For example, only eight comments out of 59 concerning the functional group Odontocetae mentioned the Northern right whale dolphin, while 36 comments were made for orca. The abundance trend of Odontocetae therefore more closely reflects the trend for orca; it is a weighted average of the relative number of comments. Ideally, one would weigh the contribution of each species to the overall functional group abundance trend using some independent estimate of relative abundance. However, in the base Ecopath model, important and commercial species (i.e., species for which independent abundance data are available), are typically assigned their own dedicated functional group.

Weighting by expertise

The interview data captures a diverse sample of local knowledge. Many fisheries (and industries) were represented at the interviews, and each sector is expected to carry its own special expertise in species of particular importance to the specialization. I therefore applied weighting to the votes offered by each participant according to their expertise. 'Expert' opinions were taken to be worth twice as much towards calculating the average (i.e., +2 and -2 for increasing and decreasing votes), 'Novice' opinions were taken to be worth half as much (i.e., +0.5 and -0.5).

The following criteria are used to define 'expert' and 'novice' comments:

1. Fishers are considered expert in their target functional groups;
2. Group interviews are novice in all functional groups;
3. First Nation group interviews remain unchanged in First Nation specialties;
4. Non-fishers are novice in all functional groups;
5. Recreational fishers are novice on all functional groups except their specialty, in which they are expert;
6. Interviewee #20 was judged expert in all functional groups;
7. Interviewee #21 was judged expert in all rockfish functional groups.

Group interviews operated on consensus; their vote is reduced in importance to limit the effect of influence between respondents in the analysis. However, one exception is made. Since the majority of First Nations respondents participated in group interviews, I do not want to reduce the impact of their comments on the LEK abundance trends. Comments made during First Nations group interviews therefore remain 'unchanged' in importance regarding the abundance of traditionally harvested species. I assume that non-fishers and recreational fishers spend less time at sea than commercial harvesters, so their contribution to the overall trend is weighted half as much. In addition to years of fishing experience, interviewees 20 and 21 had formal ecological training.

Alternatively, a weighting scheme based on years of experience could be used, although some degree of ranking by gear specialization should still be included. Information from experienced fishers does actually influence the LEK abundance trend more than information from less experienced fishers under the current methodology, since their comments apply to more years in the analysis.

Qualitative agreement of LEK versus stock assessment

To determine how often comments agreed with stock assessment records, I compare the qualitative change in abundance offered by each interviewee with time series biomass data assembled from stock assessment. For the period in which a respondent fished, an Excel macro consults time series stock assessment records assembled from various DFO publications³. The algorithm determines whether the abundance of the subject functional group had increased, stayed the same or decreased in the stock assessment record. It compares this result against the suggested population change provided by the interviewee to determine agreement. This procedure is conducted for functional groups that have continuous stock assessment information. Comments are used from only the respondents whose career spanned a period covered by stock assessment data.

For each comment made concerning a particular functional group, the span of the interviewee's career at sea is divided into two halves. The average abundance of that functional group in the first and second halves of the fisher's career is determined from stock assessment records. If the average abundance was greater in the second half than in the first, the functional group is said to have increased. If the fisher had indicated an increase in abundance, then their comment is considered 'true' (indicating only agreement between datasets). Similarly, if the stock had declined according to stock assessment, then comments that indicate a decrease in abundance are considered 'true'.

³ Vasconcellos, M. (2001). *Unpublished manuscript*. Marine Resources Service. Food and Agriculture Organization, Viale delle Terme di Caracalla, 00100 Rome, Italy. Contact Marcelo.Vasconcellos@fao.org.

An arbitrary threshold is assigned so that if only a slight increases or decreases in abundance had occurred during the fisher's career according to stock assessment records, the functional group could be considered 'stable'. Then comments providing that response would be considered 'true'. By decreasing the threshold required for a change in biomass to be considered significant, fewer comments indicating 'stable' become true. The threshold used for the analysis is set as a fraction of the total amplitude of change seen in that group's abundance since stock assessment began (see Table 3.1).

It was found that when considering all functional groups together, a threshold of 15% change in absolute biomass yielded agreement equally often between increasing, stable and decreasing votes ($\sigma^2 = 0.0018$). This approach assumes that fishers are equally likely to agree with stock assessment data regardless of the direction of abundance change. That threshold is used for all calculations. For fishers whose careers were shorter than the 61 year maximum (the most experienced interviewee), the required threshold was decreased proportionately. For example, the biomass would need to have increased or decreased by only 7% of its maximum amplitude over the course of a 30-year career to be considered significant.

Correlation of LEK time series versus stock assessment

I compare the LEK relative abundance trend of commercial groups with stock assessment records. Time series are available for the following Ecopath functional groups: chinook, coho, transient salmon, flatfish, halibut, herring, lingcod, Pacific cod, sablefish and seals and sea lions. In order to compare the LEK information with stock assessment, I convert the relative abundance time series suggested by the interviews into absolute abundance by assuming the same mean and amplitude of change as in stock assessment data. The correlation of the LEK information to the assessment records was then measured using a non-parametric Spearman's rank sum test for the weighted and unweighted interview information. I also tried dividing the time series into two periods, before and after 1965, and repeated the correlation analysis.

Challenging the model

The models are challenged with the LEK data to verify their structure and dynamic functioning. LEK information is used here in two ways: first, as a test of the relative (static) structure of the 1950 and 2000 Ecopath models, and second, as a test of the dynamic function of a 50-year Ecosim simulation beginning in 1950.

Static structure

LEK information can serve as an independent check to compare Ecopath models of different periods. Here I attempt to verify that the relative abundance has increased, remained the same, or decreased between the 1950 and 2000 models in accordance with the fishers' average perception. By weighing the credibility of our model data source against the magnitude of the disparity with the LEK information, a judgment can be reached whether or not to accept an alternate value, if one is available, or allow Ecopath to estimate that parameter. Table 3.2 compares model biomass parameters with the LEK trend, and includes Ecopath's data ranking pedigree as a measure of data quality (see Christensen *et al.*, 2004a). The pedigree describes the following ranking of data quality, where 1 indicates the lowest quality data and 6 indicates the highest:

1. Estimated by Ecopath;
2. From other model;
3. Guesstimate;
4. Approximate or indirect method;
5. Sampling based, low precision;
6. Sampling based, high precision.

Ecopath's pedigree considers any user input to be more reliable than an internally generated value. However, the six criteria listed above are only established by convention – if the user has reason to believe that a value estimated by Ecopath is reliable, a higher data quality ranking can be entered manually. Similarly, if a 'guesstimate' is made by expert opinion, it may warrant a better ranking than 3.

Dynamic function

LEK data is used to verify dynamic group interactions occurring in a 50-year harvest simulation from 1950 to 2000. The simulations are driven by historic production and fishing mortality trends (see Chapter 5) to reconstruct real-world population dynamics observed since 1950 (for groups where stock assessment records exist). For all functional groups, Ecosim's predicted abundance trend is compared with the suggested biomass trend from the LEK interviews, and stock assessment information assembled in Chapter 5 (for commercial groups only). A Spearman's rank correlation test determines whether abundance trends are in concordance with the two datasets.

3.3 Results

Qualitative agreement of LEK versus stock assessment

Appendix Fig. A3.1.1 shows the LEK trends for functional groups that had adequate coverage in the interview materials; the trends are standardized to a mean of zero. Table 3.1 records the fraction of instances where the interviewee comments qualitatively agree with stock assessment records, varying the threshold of abundance change required to be considered significant. As that threshold is decreased, fewer 'stable' comments become true.

Table 3.1 Percentage of interviewee comments that agree with stock assessment records. Shown at left is the threshold of abundance change required for 'increase' or 'decrease' votes to be considered true, as a fraction of the total amplitude of abundance change seen in stock assessment. If the abundance change is less than this amount, then 'stable' votes are considered true. This threshold is proportionately reduced for fishers whose career is shorter than the maximum (61 years). Fisher's exact test shows that 'increase' votes agree more often with stock assessment at low threshold values. Based on ($n = 234$) comments.

Threshold	Increase	Stable	Decrease	Total	Variance	Exact p
	(% values)					
0	58	0	36	34	0.086	0.002
0.05	58	22	32	37	0.035	0.0004
0.10	49	31	30	36	0.012	0.005
0.15	37	38	30	34	0.002	0.082
0.20	37	40	29	34	0.004	0.063
0.25	28	42	26	30	0.007	0.129
0.30	19	56	23	30	0.041	0.126
0.35	15	66	17	28	0.082	0.158
0.40	15	73	14	28	0.113	0.171
0.45	15	78	13	29	0.139	0.159
0.50	12	80	9	27	0.162	0.161

Overall, agreement between LEK and stock assessment datasets is poor, with a maximum of only 37% of comments agreeing with official records. This highest level of agreement occurs when the abundance change threshold is set to 5% of the total amplitude of change observed in stock assessment records. Votes that indicate increase, stable and decrease are true equally often when the threshold is set at 15% ($\sigma^2 = 0.002$). At most threshold levels, the 'increase' votes show agreement with stock assessment more often than 'decrease' votes. The discrepancy is significant ($\alpha = 0.05$) at all levels of threshold below 15%, according to Fisher's exact test.

Fig. 3.1 shows the fraction of comments that agree with stock assessment records per functional group at a biomass threshold level of 15%. Not shown, flatfish comments ($n = 16$) were never in agreement with stock assessment. Transient salmon is a composite functional group; the

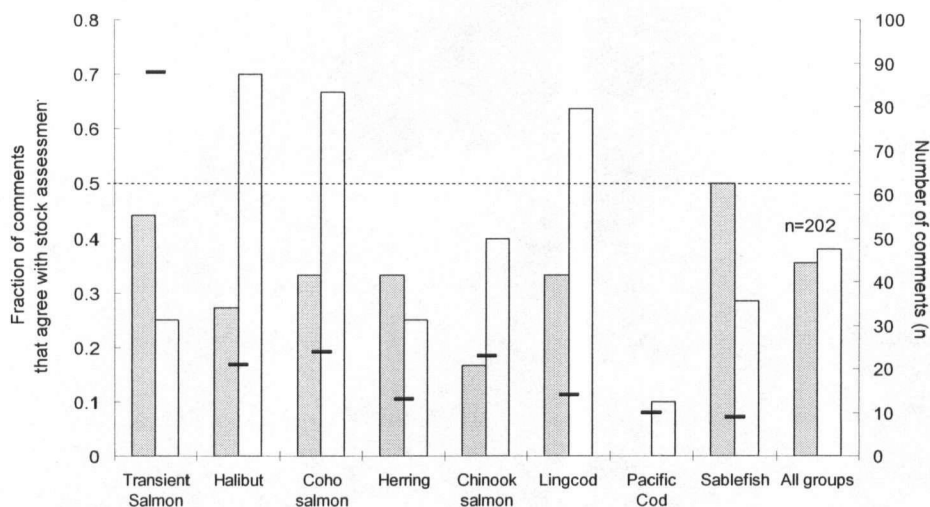


Figure 3.1 Fraction of comments that agree with DFO records by functional group.

Dark bars represent expert comments; light bars represent non-expert comments. Broken line at 50% indicates proportion of correct responses expected by chance. Crossbars show the number of comments received for each functional group. Biomass threshold level is 15%.

comments ($n = 90$) refer to sockeye (*Oncorhynchus nerka*), chum (*O. keta*) and pink salmon (*O. gorbuscha*).

A binomial test shows that comments provided for three functional groups disagree with stock assessment more often than could be expected by chance at $\alpha = 0.05$. By experts, the interview trend for chinook contradicts stock assessment ($p = 0.004$; Spearman's rank correlation); by non-experts, the interview trend for transient salmon and Pacific cod contradicts stock assessment ($p = 0.002$ and 0.011 respectively). Non-expert comments agree with stock assessment more often than expert comments for all groups except transient salmon, herring and sablefish. Although non-experts are in agreement with stock assessment more often than experts, they are not as consistent across functional groups ($\sigma^2 = 0.010$ for experts; $\sigma^2 = 0.046$ for non-experts). Still, there is little evidence to support the division between expert and non-expert (non-expert including both unchanged and novice votes).

Fig. 3.2 tests whether experienced interviewees are in agreement with stock assessment more often than less experienced ones. There is a trend suggesting that agreement between datasets improves as fishers' experience increases. Fisher's exact test reveals that interviewees with 40 or more years of experience provide a significantly ($p = 0.045$) higher fraction of comments that agree with stock assessment (41%, $n = 74$) than less experienced interviewees (31%, $n = 148$).

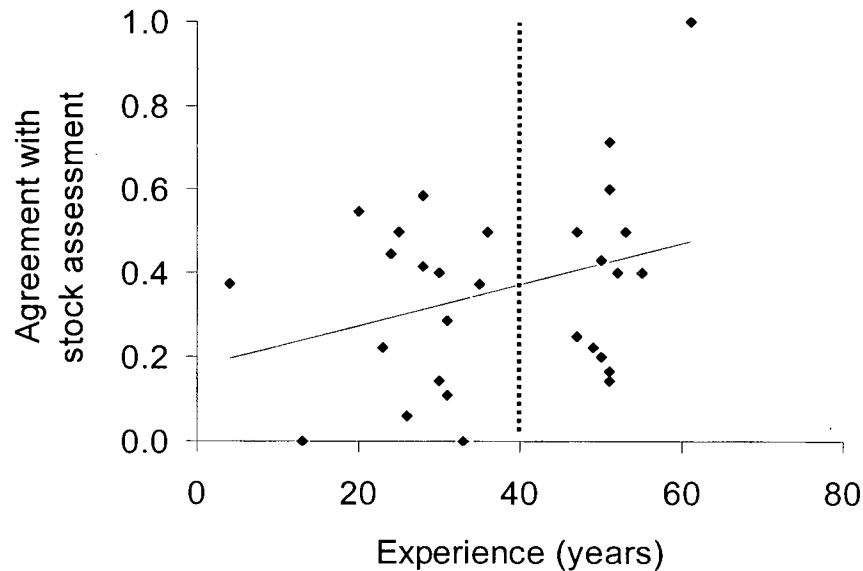


Figure 3.2 Interviewee agreement with stock assessment data by career length. Y-axis shows the fraction of comments that agree with the qualitative stock assessment trend (increase, stable or decrease); trend line is shown (solid). Dotted line shows 40-year division for Fisher's exact test.

Correlation of LEK time series versus stock assessment

Fig. 3.3 presents the absolute abundance for ten functional groups estimated from the interview materials, and from DFO stock assessment information. The average and amplitude have been scaled to match the stock assessment record. Unweighted abundance trends are shown. Fig. 3.4 summarizes correlation of weighted and unweighted LEK trends with stock assessment.

There is a significant positive correlation between the interview trends and stock assessment for four groups using the unweighted LEK trend, and only three groups using the trend weighted for

expertise. In fact, there are significant negative correlations for several groups, indicating that the average fisher perception is in contradiction to the scientific dataset. The abundance trend weighted by respondents' expertise outperforms the unweighted trend only for chinook salmon.

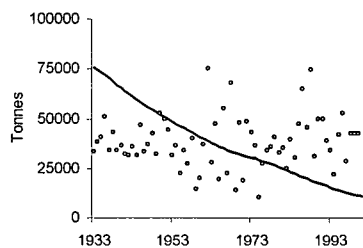
With functional groups that display a large degree of inter-annual variability, the correlation of the LEK trend versus stock assessment can be expected to suffer, since the abundance trend from the interview data is not suited to detect fine-scale (e.g., annual) changes in abundance. The LEK trend may be better suited to detect decadal changes. In fact, the best correlation occurs in lingcod and sablefish, two long-lived species whose abundances even under long-term trends do not fluctuate greatly from year to year.

Since fewer interviewees could contribute to the early years of the analysis, I tried dividing the abundance datasets into 2 series (1933-1965 and 1965-2000). The expectation was that the LEK trend would match stock assessment information better, and for more functional groups, in the latter time series than in the former. However, not only did the LEK trend from the 1965-2001 data series achieve agreement with stock assessment less often than the 1933-1964 series, significant negative correlations were found for six functional groups, as opposed to three for the combined data set in Fig. 3.4. Even when there were a maximum number of respondents contributing to the LEK trend, their perceptions of change in abundance still disagreed, and more frequently, with stock assessment information.

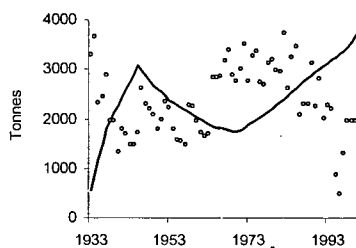
The failure of these experts to perfectly recreate the scientifically assessed population trends is not necessarily an indication that the information is generally untrue or unhelpful. The discrepancy with data may partly be due to animal biology, observational bias or psychological factors, and some of these effects could be mediated by a more refined analysis (see discussion). It is worth noting that the judgments of 'experts' can also contradict analytical information in other fields of human endeavor, yet the judgments of experts are nonetheless accorded their appropriate respect. For example, the skilled predictions made by economists appear subject to bias when compared to time series data (Brown 2001, Richardson et al. 2004), and can sometimes be less accurate than 'naïve' forecast models that use simple prediction rules (e.g., Ciccone, 2004). In no way does this diminish the employability of financial analysts. Similarly, researchers studying the accuracy of weather predictions found that, on average, meteorologists

tended to outperform a simple persistent temperature model (which assumes that tomorrow's temperature will be the same as today's) in only ~60% of U.S. cities (Anon. 2005). Even fishery scientists, with their qualifications and expertise, frequently disagree on the interpretation of stock status based on immediate indicators. Moxnes (1998) confirmed that fishers, fishery bureaucrats and scientists judged stock status equally well in simulated stock management. I take this as evidence that the interpretation of the interviewees is a viable source of information, and that their opinions are as reliable as the anecdotal judgments of other, more formally recognized marine experts.

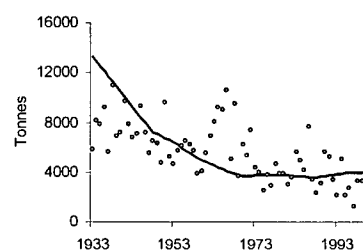
Transient salmon



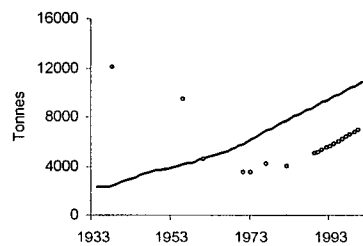
Chinook



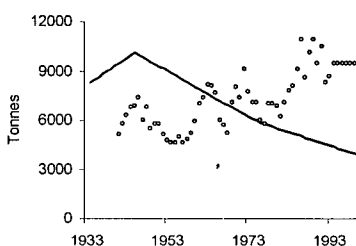
Coho



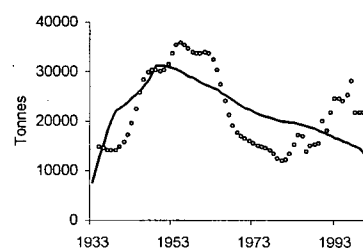
Seals and sea lions



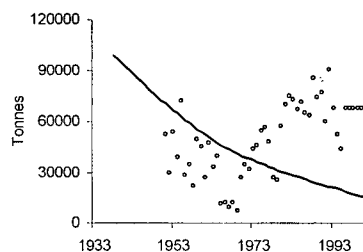
Flatfish



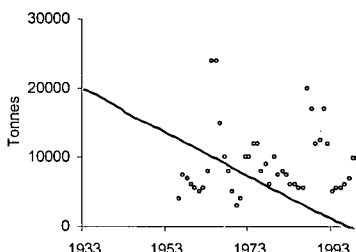
Halibut



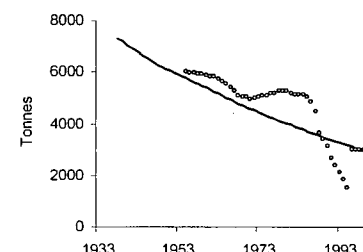
Herring



Pacific cod



Lingcod



Sablefish

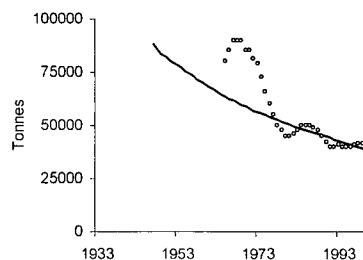


Figure 3.3 LEK abundance trend versus stock assessment. Open circles show stock assessment; solid line shows unweighted LEK trend. Absolute LEK trend is scaled using the same mean and amplitude as stock assessment.

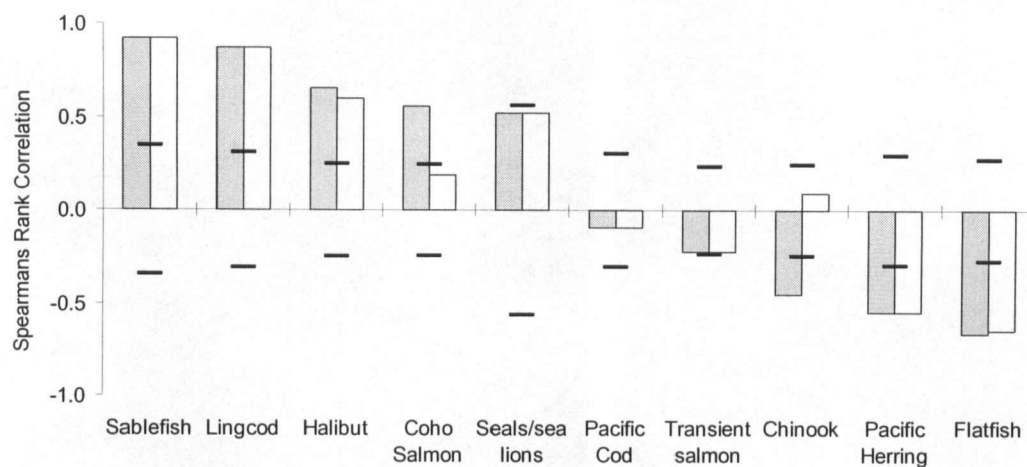


Figure 3.4 Rank correlation of LEK abundance trend versus stock assessment. Dark bars show unweighted LEK trend; light bars show weighted LEK trend; crossbars indicate correlation required for significance at $\alpha = 0.05$.

Challenging the model

Static structure

Table 3.2 shows biomass estimates used in the 1950 and 2000 Ecopath models. Biomass change between these periods is compared to the trend suggested by LEK materials. The LEK column indicates the net change in fisher's perception of abundance since 1950 according to the trends determined in Appendix Fig. A3.1.1. The data pedigree in the right column indicates the quality of data used in the 2000 model. Data quality in the 1950 model is generally poor for non-commercial groups. Not shown are the functional groups whose biomass remains constant between modelled periods.

Table 3.2 Biomass estimates ($t \cdot km^{-2}$) used in Ecopath models compared to LEK trend. LEK column indicates if interviewees perceived a positive or negative change in species group biomass from 1950 to 2000; the LEK trend is averaged across respondents without weighting; the LEK trend for composite functional groups is weighted toward species most often mentioned in interviews.

	1950	2000	Change	LEK	Agree?	Data pedigree ¹
Seals and sea lions	0.130	0.256	0.126	+	Y	6
Transient salmon	0.500	0.208	-0.292	-	Y	5
Coho salmon	0.100	0.024	-0.076	+	N	6
Chinook salmon	0.090	0.036	-0.054	+	N	6
Dogfish	0.417	0.909	0.492	+	Y	2
Forage fish	7.600	8.478	0.878	-	N	1
Eulachon	1.893	1.660	-0.233	-	Y	1
Herring	1.001	0.658	-0.343	-	Y	6
Piscivorous rockfish	0.541	0.654	0.113	-	N	1
Flatfish	0.535	0.236	-0.299	-	Y	5
Halibut	0.429	0.628	0.199	-	N	6
Pacific cod	0.348	0.163	-0.185	-	Y	4
Sablefish	0.600	0.269	-0.331	-	Y	4
Lingcod	0.104	0.039	-0.065	-	Y	2
Large crabs	0.506	0.456	-0.050	-	Y	2
Epifaunal invertebrates	11.191	13.448	2.257	-	N	1

¹Data pedigree column indicates the quality of data. This scale measures from 1, the lowest quality (data point estimated by Ecopath) to 6 (high precision sample based measures).

For 10 of the functional groups analyzed (63%), the qualitative LEK trend agrees with the biomass data used to construct the 1950 and 2000 models, which is a marked improvement over the findings of Ainsworth and Pitcher (2005a); they found agreement in only 31% of functional groups. This indicates that recent revisions to the 1950 and 2000 models have reduced discrepancies with the LEK information. However, a one-tailed Fisher's exact test shows that agreement between LEK and the models' change in biomass is independent of the data pedigree

($p = 0.549$), when data quality is divided into two categories, high and low. LEK information is no more likely to contradict low quality data than high quality data.

However, agreement between the LEK trend and model data is apparently related to the direction of biomass change, in contrast to the findings of Ainsworth and Pitcher (2005a) who detected no bias⁴. Of the 16 functional groups in Table 3.2, respondents consistently guessed the direction of biomass change correctly for groups that had experienced a stock decline between 1950 and 2000; (the change in 8 out of 10 functional groups was represented accurately by the LEK information). However, fewer interviewees answered in accordance with the models if the biomass had increased during that period (4 out of 6 functional groups are contradicted by LEK). A one-tailed Fisher's exact test indicates a weak bias ($p = 0.092$).

Dynamic function

A 50-year Ecosim simulation was run using the 1950 model as the starting point. Fig. 3.5 shows how well LEK and stock assessment correlate with the Ecosim model's predicted biomass trend using a non-parametric Spearman's rank sums test.

⁴ The analysis of Ainsworth and Pitcher (2005a) is based on older versions of the 1950 and 2000 Ecopath models of northern BC from Ainsworth *et al.* (2002).

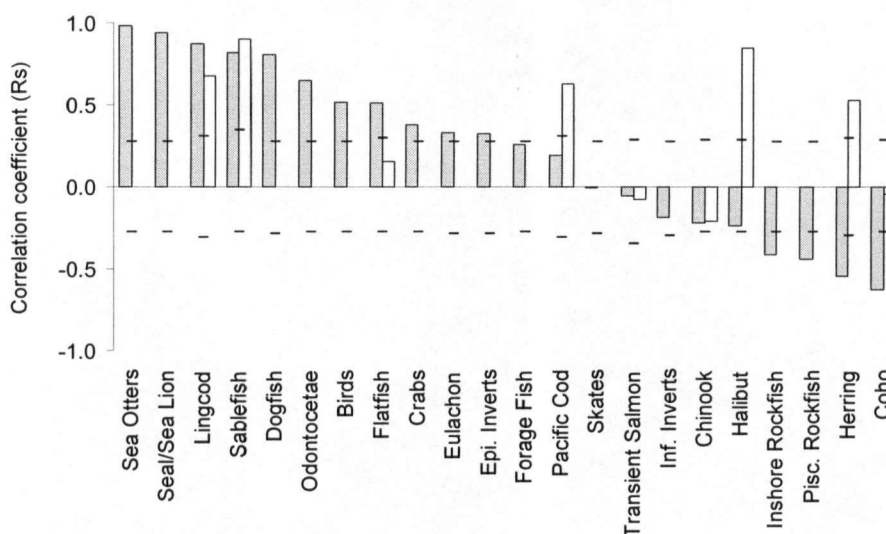


Figure 3.5 Correlation of LEK relative abundance trend and stock assessment with model outputs. Dark bars show correlation of LEK relative abundance trend versus Ecosim predictions; light bars show correlation of stock assessment versus Ecosim predictions. The 50-year biomass trajectories predicted by the 1950 model are driven by historical fishing mortalities and production modifiers. Crossbars show significant correlation at $\alpha = 0.05$.

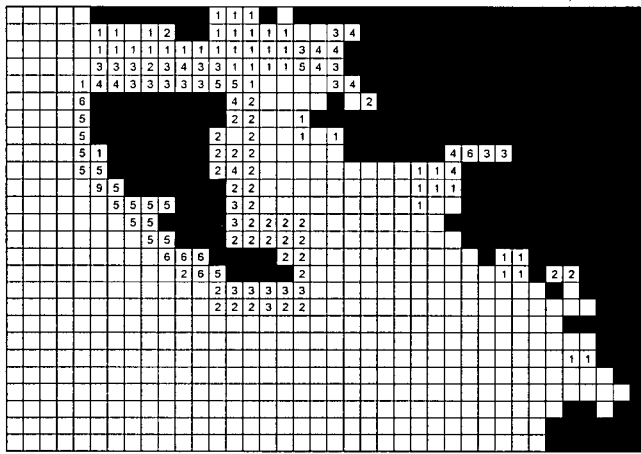
Spatial analysis of LEK information

Many of the comments received in interviews included a spatial reference. Although modeling efforts using Ecospace (Appendix 8) do not use this information in any quantitative way, the data has been summarized in maps showing the location of areas often cited. If, for any functional group, one assumes that an area mentioned often in interviews is likely to contain a higher abundance of animals than an area mentioned less frequently, then the number of comments concerning a particular region can be viewed as an indicator of relative abundance in that group. Certainly, the spatial distribution of comments is biased to include common fishing areas and travel routes. The study area was divided into cells, and comments mentioning species presence in specific areas were tallied in the corresponding grid square (Table 3.3). Fig. 3.6 provides an example for Odontocetæ and seabirds. Charismatic and commercial functional groups tended to receive the most comments overall, but similar maps have been constructed for a number of functional groups.

Table 3.3 Place names mentioned during interviews. Each area was assigned a corresponding set of grid cells on the map of the study area for a spatial representation of relative abundance.

2 peaks	Finlayson Arm	Metlakatla	S. Dixon Entrance
Alice arm	Flamingo Inlet	Milbanke Sound	S. QCI
Aristazabal Island	Freeman's Pass	Mill rocks	Sandspit
Banks Island	Gardner Channel	Moresby Island	SE. Dundas
Bella Bella	Gilttoyees Inlet	N. Danger rocks	SE. QCI
Birnie Island	Goose Island	N. Dixon Entrance	Skedans Point
Bonilla Island	Grenville Channel	Nass River	Skeena River
Browning Entrance	Grenville rocks	NE. Graham Island	Skidegate
Burke Channel	Hartely Bay	NE. Moresby Island	Smith Island
Burnt cliff	The 'horseshoe'	N. QCI	Spiller Channel
Butterworth rocks	Hudson Bay Pass	Ogden Channel	Saint Johns
Caamano Sound	Hunts Inlet	Oval Bay	Sue Channel
Cape Calvert	Kennedy Canal	Porcher Island	SW. Moresby
Centre Hecate St.	Kennedy Island	Port Dundas	Thurston Harbour
Chatham Sound	Kildala	Port Edward	Triple Island
Chief Matthews Bay	Kitasu Bay	Port Hardy	Ursula Channel
Collins Bay	Kitimat	Port Simpson	W. Banks Island
Devestation Channel	Kitkatla	Portland Inlet	W. Dixon Entrance
Digby Island	Klemtu	Princess Royal Island	W. Dundas
Dixon Entrance	Langara Island	Principe Channel	W. Gill Island
Dogfish Banks	Laredo Sound	Promise Island	W. Graham Island
Douglas Channel	Lema Pass	Queen Charlotte Islands	W. Moresby Island
Dundas Island	Louise Island	Roland rocks	Whales Island
E. Dixon entrance	Lucy Island	Rose Inlet	Work Channel
Eddy Pass	Macintyre Bay	Rose Spit	W. QCI
E. QCI	McNichol Creek	Rupert Harbour	Wright Sound
Estevan Island	Melville	S. Chatham Sound	

Odontocetae



Seabirds

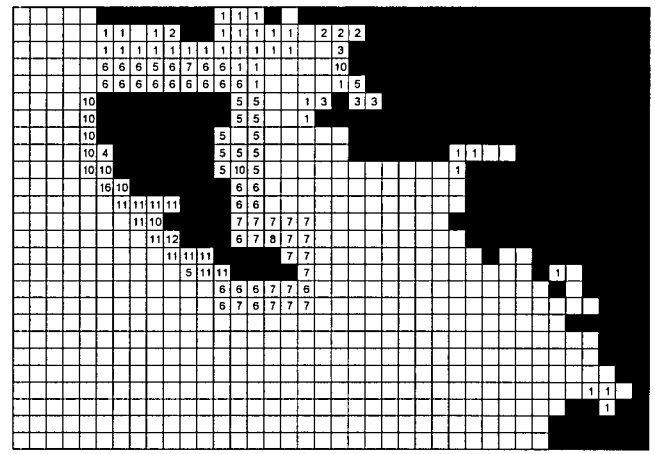


Figure 3.6 A map of the study area showing the number of LEK comments indicating species presence. Place names mentioned in interviews were assigned to a corresponding set of grid cells; the frequency that an area was mentioned may indicate relative abundance, although distribution is likely biased to include fishing locations. Examples for Odontocetae and seabirds are provided; detailed spatial information was recorded mainly for charismatic and commercial functional groups.

The collated spatial information may be used as an independent data source regarding distribution of species to complement existing spatial information. For example, DFO has spatial information regarding groundfish distribution from Hecate Strait trawl survey records (e.g., Schnute and Haigh, 2000), and there are upcoming Dixon Entrance and Queen Charlotte Sound surveys (Sinclair *et al.*, *in press*). Relative abundance indices from catch per unit effort (CPUE) estimates are also available from observer records (NWFSC, 2004) (i.e., GF TRAWL and PAC HARV datasets).

As this chapter compares temporal species abundance information between two data sources, the LEK responses and official statistics, so too may an analogous spatial analysis allow us to identify areas where fishers perceptions differ from the scientific understanding concerning aggregations and community structure. Eventually, spatial information from these sources, particularly spatial information forming a time series, could be used to parameterize Ecospace in the same way that abundance time series are used to parameterize Ecosim models. Such an analysis should be carefully considered as far as the application of LEK data is concerned. It would be prudent to restrict the analysis only to popular fishing areas to reduce any bias in

spatial reporting – and only functional groups that received wide mention in the interviews should be considered quantitatively. However, there is an immense volume of spatial information contained in the BTF Historical and Interview Database, and only a fraction of it has been utilized in the present investigation. Future efforts to model northern BC fisheries using spatial techniques would be well advised to consider this resource.

3.4 Discussion

Validating LEK

In comparing the LEK trend with stock assessment, agreement is poor, only 37% of comments agree with the qualitative trend (increase or decrease) as indicated by stock assessment, although agreement does increase with years of experience. There are a number of possible explanations. For example, retired fishers may be inclined to answer differently than those who continue to rely on the resource. As one reviewer of Ainsworth and Pitcher (2005a) pointed out, abundance trends suggested by local ecological knowledge may also partly reflect changes in the spatial distribution of species – particularly if LEK is more spatially restricted than broad-scale survey data. However, there are enough site-specific references in the interview materials that we could compare abundance trends with stock assessment by area to estimate this potential bias.

It is clear that LEK information is better suited to detect long-scale trends in population abundance. The best correlation with stock assessment occurs in slow growing species, whose abundance does not fluctuate greatly from year to year. However, considering that LEK comments indicate a decrease in abundance for the majority of functional groups, regardless of fishing experience, it is likely that fishers' perceptions come to resemble the scientific understanding only when considering the long-term trend. Presumably, a steady depletion (particularly among commercial species) becomes obvious over the course of several decades, where a short-term trend can be mired in fluctuations, and is open to interpretation. Interestingly, I found that interviewees were more likely to contradict stock assessment if they were reporting a decrease in abundance. This suggests that respondents are more likely to err on the side of pessimism and/or stock assessment is more likely to err on the side of optimism. That is not to suggest that one dataset is more accurate than the other, only that there is persistent

bias in that direction. At any rate, discrepancies show where stock assessment records are in contradiction with fishers' perceptions.

Challenging the models

I tested the relative biomass values used in the static 1950 and 2000 Ecopath models against the LEK trend. The LEK trend verifies the change in biomass for 9 out of 16 functional groups. Of those that disagree with LEK, we can remain confident in the data sources concerning well-studied groups such as coho, chinook and halibut. However, the LEK trend also disagrees with data-poor groups like forage fish, piscivorous rockfish and epifaunal invertebrates. However, as will be demonstrated in Chapter 5 (see Appendix Fig. A5.4.1), the relative abundance trend predicted by the LEK information does fall within the confidence limits assigned to these data-poor groups.

I next compared the LEK and stock assessment datasets with the output of a 50-year simulation. Four functional groups show a significant negative correlation with the LEK trend: inshore rockfish, piscivorous rockfish, herring and coho salmon. Of these, the herring group is vindicated by a strong positive correlation with stock assessment data. Suspect groups are therefore coho salmon, inshore rockfish and piscivorous rockfish. For coho, as with all salmon groups, the problems involved in modeling a highly migratory stock make it difficult to recreate observed dynamics in Ecosim when the model is driven by only local mortality and production series. For that reason, salmon biomass was forced in all fitting procedures (Chapter 5). Predictive forecasts concerning salmon may suffer from inaccuracies however, and LEK information calls rockfish dynamics into question as well. The uncertainty regarding these groups can be addressed using broad confidence intervals in Monte Carlo procedures used to verify dynamic output (*vis. Pitcher et al., in press*).

Future work

The analysis presented in this chapter represents a rough, first attempt to quantify abundance information provided by LEK interviews. Poor agreement between stock assessment time series

information and the LEK abundance trends estimated here raises questions. It is possible that disagreement between the community members and scientific data sources reflect legitimate discrepancies between the scientific understanding and the perceptions of resources users. Before we make that conclusion, the methodology should be reconsidered.

Future revisions to this work should consider several elements. First, the methodology should account for the location of fishery activities, as the interview information probably contains a strong bias towards these areas relative to stock assessment data. Considering stock assessment information from only the most popular fishing areas may be one way to resolve the disagreement. Alternatively, a more thorough analysis of the interview responses may be required. One suggestion would be to apply a fuzzy logic approach to estimate the relative abundance trend from qualitative remarks. Another meeting with community members and presentation of the current results would also help us decide whether or not stock assessment information is in conflict with local knowledge, or whether the current methodology is too basic to accurately reflect fishers' perceptions.

If after additional analysis and consultations, there still exists a strong discrepancy between LEK trends and stock assessment information, several factors could be at work. Observational selection effects may bias the trend from interviews when compared to stock assessment unless the scientific data refers to a similar area, season and environment in which the experts are familiar. Perhaps the perceptions of resource users differ from the conclusions of formal science, or perhaps there are other psychological factors at work that limit the usefulness of the LEK information to a detailed quantitative analysis of ecosystem structure. In any case, the application of this information is of limited value from the prospect of strict biological modeling. The data may still be useful from a sociological perspective but since the scope of the present investigation is not able to satisfactorily resolve the discrepancy between the two data sources, the LEK data is used sparingly throughout this volume to guide the modeling.

Scientific abundance information from surveys, models and other sources were always considered more reliable than the LEK information. Specific instances are mentioned throughout Chapter 5 where interview information was used to guide construction of the static Ecopath models, and Appendix Fig. A5.4.1 demonstrates a limited use of LEK information in

parameterizing ecosystem dynamics for data-poor functional groups. Specifically, the start and end points of model predictions were tied to our best biomass estimates from scientific sources, while system dynamics during the interceding years were guided by LEK trends.

Where this chapter contributes biomass trends to help guide the modeling process, the next chapter will improve estimates of removals through time by quantifying illegal, unreported and unregulated catches in the BC marine ecosystem.

4 ESTIMATING ILLEGAL, UNREPORTED AND UNREGULATED CATCH

When it was full, the fishermen pulled it up on the shore. Then they sat down and collected the good fish in baskets, but threw the bad away.

Matthew 13: 48

4.1 Introduction

To fully understand the ecological impact of fishing on the marine environment, it is necessary for biologists to have an estimate of total extractions from the ecosystem. In addition to nominal fisheries landings and reported discards, which are regulated and monitored, removals will include a certain amount of IUU catch. Quantifying these removals can present a significant technical challenge if, for certain fisheries and species, regulatory agencies hold no mandate to record catch statistics. The difficulty is compounded by the politically sensitive nature of the question. Sometimes governments are reluctant to reveal the scope of catch that escapes their notice, or to embarrass particular sectors engaged in 'dirty' or illegal activities, and fisheries agencies may have few incentives to attempt to estimate the quantity of unreported catch.

In the absence of reliable estimates, some assume an implicit 'zero' quantity for these elements (Pitcher *et al.*, 2002d). It is a potentially dangerous assumption. If considered, the missing catch could profoundly affect estimates of stock abundance and safe removal rates. When managers are forced to set harvest goals without knowing at least the magnitude of catch left unaccounted for, we put fisheries and ecosystem services at risk. If the missing quantities are significant, then not only can 'blind' management jeopardize ecosystem structure, but fisheries benefits could also be compromised, since a precautionary and ecosystem-based approach to management demands conservative harvest regulations (Evans, 2000). Moreover, the presence of IUU fishing distorts and devalues information obtained from compliant sectors, often at their own expense.

Here I present a methodology to estimate the quantity of IUU catches over time, based on influences in the history of the fishery, and on independent estimates of misreporting. I use a Monte Carlo routine to determine missing catch with an associated error range for British Columbia salmon and groundfish fisheries. Although any reasonable estimate of IUU would be better than a zero-rate assumption, the values calculated here, using this subjective but transparent methodology, are intended to provide a starting point for further discussion and amendment. This chapter is published in Ainsworth and Pitcher (2005c; 2005d).

What is IUU?

For many fisheries, the largest component of IUU will be discarded bycatch, which may or may not be illegal, but is generally not recorded by fishery observers. Illegal catch refers to catch deliberately concealed, or misreported as other species to contravene regulatory limitations (such as time or area closures, species quotas, gear restrictions and so on). Illegal catch may also include unreported harvests landed in foreign ports or trans-shipped to foreign vessels at sea. Illegal catch is the most difficult component of IUU to quantify as an accurate record may be hard to obtain, even from surveys, if fishers are reluctant to contribute for fear of incriminating the industry – while the presence of onboard observers is likely to curb such activities altogether. Finally, IUU will include unregulated catches of species which authorities are not mandated to monitor, or harvests originating from certain vessels or gear types that are not subject to strict accounting.

BC Case study

In west coast Canadian fisheries, the only IUU component regularly assessed is discards for the groundfish trawl and hook and line fleets. Efforts to quantify discarding through use of on-board observers have been limited to large vessels. Although observer coverage is now very high (100% of vessels are covered, but not for 100% of the time) for groundfish trawl, coverage is not adequate in the hook and line fleet (Haigh *et al.*, 2002). Where it is assessed, time-series discard information extends back barely a decade. Although attempts have sometimes been made to quantify missing catch through statistical techniques (e.g., Patterson 1998), no reliable estimates

are published for BC (A. Sinclair, Department of Fisheries and Oceans. Pacific Biological Station. Nanaimo. *pers. comm.*).

Here I examine discards and illegal catch present in the major fishing sectors for salmon (gillnet, troll, seine and recreational) and groundfish (bottom trawl, hook and line and recreational). Discards in the salmon and groundfish fleets are assumed to contain both 'unreported' and 'unregulated' catch, but I do not try to distinguish them. A separate analysis of the salmon recreational fleet provided estimates of missing catch, which I call 'unreported'.

4.2 Methods

The estimation procedure

To estimate missing catch I use a technique similar to Pitcher *et al.* (2002d). For example, Pitcher and Watson (2000) estimated IUU for Atlantic Canada, Pitcher *et al.* (2002d) looked at Iceland and Morocco (also Forrest *et al.*, 2001), and Kalikoski *et al.* (*in press*) considered Chile. The methodology can be broken down into seven steps.

1. Create a timeline of the fishery - taking note of regulatory, technological and political changes that are likely to have affected the quantity of fishery discards, illegal, unreported and misreported catch.
2. Assign 'influence factors' to each event (usually increase or decrease), to describe the effect on IUU rates.
3. Based on the frequency and severity of influences, assign an 'incentive' rating (e.g., low, medium or high) to describe the overall incentive to misreport for each five or ten year period in the timeline.
4. Establish an absolute range of values for each 'incentive' rating (e.g., in percent IUU catch per target species catch) – these are based on fixed 'anchor points', quantitative estimates of IUU available from the literature or expert opinion.
5. Scale absolute IUU estimates for missing periods, based on relative 'incentive' rating.

6. Using the range established in step 4, provide an estimate of total extractions for each fishery (reported plus missing catch), weighing the contribution of each gear type to IUU by its mean reported catch. For each period, estimates will contain an upper and lower bound. If possible, determine a 'best guess' estimate within the total range.
7. Use Monte Carlo resampling to determine the mean weight of missing catch with associated confidence intervals for each period, based on the likely error range established in step 6. Previous authors have assumed an asymmetric triangular distribution around a specified mode (the 'best guess').

Revisions to the methodology

The method used in this article has been modified from previous case studies. Other authors combined fishery discards, illegal catch, and other unreported or unregulated sources of catch into a single quantity, IUU, assuming that influence factors affect each component equally. However, it is likely that certain regulatory, technological and political changes will affect the categories of IUU differently. For example, an area closure for the trawl fleet meant to protect sensitive benthic habitat may reduce discarding of non-target benthic organisms, but at the same time will introduce an opportunity for poaching groundfish. So, in this paper I expand the methodology to consider these IUU categories separately. I develop an independent history of influences for each type of IUU based on a literature review (Appendix Table A4.1.1). Categories of IUU examined are discards (including both unreported and unregulated catch), illegal catch,² and unreported catch. I construct three absolute trends using independent anchor points for discards, illegal, and unreported catch. Parallel analyses are conducted and then combined to provide an estimated sum of IUU. By keeping these IUU components separate in the analysis, I hope that the technique will be more flexible and adaptable to any fishery. The relative quantities of missing catch in each category will also signal to managers what actions are required to reduce misreporting.

Influence factors

I also try to introduce a more precise methodology to assign influence factors. Where other authors allocated to each historical event a simple influence factor indicating an 'increase' or 'decrease' in the rate of IUU catch, I refine the ranking here into minor and major influences. This system can be used to discriminate significant from routine changes in the fishery, or it can be used to restrict the influence of certain events if they affect only a portion of the fleet or study area. The assumption introduced is that minor influences have half the effect of major influences in determining the rate of misreporting.

I then create a numerical running total throughout the time series, where major positive influences add 1.0 to the cumulative score, and minor positive influences add 0.5; negative influences subtract the same. In this way, events that have duration will contribute, and then withdraw from the ranking; multiple events will be additive, and so on. The influence table (Appendix Table A4.2.1) considers 154 events in the history of BC fisheries since 1950 which are likely to have affected IUU rates, including changes in management and politics, as well as technological and market developments.

Quantifying incentive

Previous authors used a subjective and arbitrary technique to assign each historical period a 'low', 'medium' or 'high' incentive rating based on their general impression of the severity and frequency of influences within that period. To standardize the process of quantifying incentives, I divide the total amplitude of the numerical influence trend into five categories: low, low/medium, medium, medium/high and high. Fig. 4.1 shows an example. Total influences affecting unreported catch for groundfish trawl begin with a 'medium' incentive score in the 1950s, indicating median levels of missing catch. Changes in the fishery increase unreported catch throughout the 1970s and 1980s to 'high', but by the 1990s the trend has reversed and incentives quickly fall to 'low'.

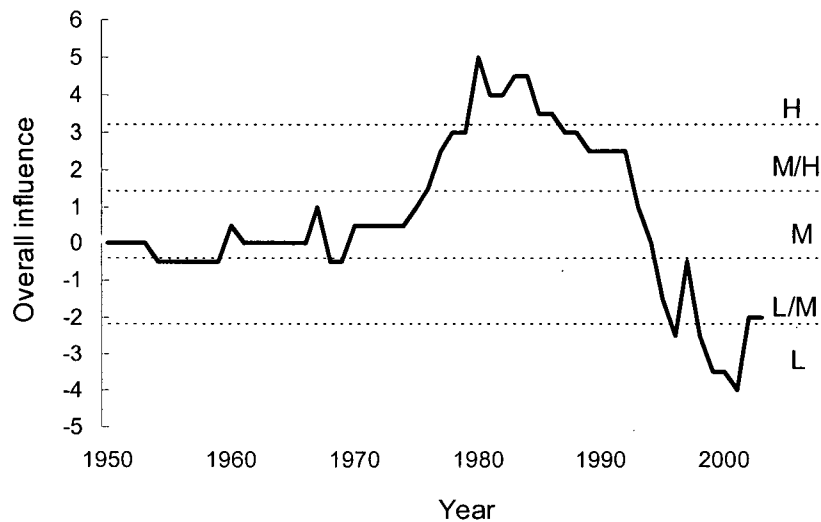


Figure 4.1 A time series of numerical influence factors assigned semi-quantitative ‘incentive’ ratings. Ratings are high (H), medium-high (M/H), medium (M), low-medium (L/M) or low (L). Example shows unreported catch for groundfish trawl.

As here, all previous IUU studies under this methodology used five categories to describe the magnitude of the incentive factor – most authors have labeled these categories low, low/med, med, med/high and high. Table 4.1 shows the predicted incentives for each period and gear sector used in the analysis. Grey cells indicate periods where anchor points exist.

Anchor points

To turn the incentive ratings into a series of absolute catch, it is necessary to ground the relative trend using anchor points – examples of known discards, illegal and unreported catch taken from the literature and other sources. Table 4.2 indicates the range of estimates available in the literature for each of these IUU categories, shown as a percentage of reported catch for each gear type. Absolute quantities for the lower and upper bounds were calculated based on official catch statistics summarized in Table 4.3. Data is averaged over five year periods. Assembled catch statistics are presented in Appendix 4.3; Table 4.3.1.

Discards

Information regarding discards for the salmon and groundfish fisheries includes data from experimental fisheries, onboard observer programs and predictive models. In some cases, data from outside BC is used. In the case of groundfish trawl, I often assumed that proportional discard data from the halibut fishery (which has the most information) can be applied without modification to other groundfish target species as well. See Table 4.2 for additional caveats.

Illegal catch

With regard to illegal catches, the anchor points provided in Table 4.2 represent a very rough first attempt at this quantity. Each year, DFO news releases record, for only a small proportion of incidents, the confiscated weight of illegally caught salmon, groundfish and other species taken by fishery officers during vessel inspections, road blocks, and other enforcement operations. No compendium exists, however, of total confiscated weight for any year, as the DFO Protection and Conservation Branch is not mandated to record that information along with the legal record of charges laid. In addition to the large number of incidents that presumably go unnoticed, the specific record of confiscated weights available in the news releases refers to only exceptional cases that are deemed newsworthy by the press (Anon., DFO. Victoria. *pers. comm.*). To accurately estimate the total weight of illegal catches occurring in the salmon and groundfish fisheries, rigorous surveys would be required. Under the scope of the present investigation, I therefore make a critical assumption: that DFO news releases account for 10% of the total weight of fish taken illegally (including weight confiscated but not reported, and including illegal catch that goes unnoticed by authorities). In reality, DFO news releases probably account for a much smaller percentage of the total illegal catch, so the final estimate is bound to be conservative.

Unreported catch

Anchor points for unreported catches of the salmon recreational fleet are based on the discrepancy between the two available datasets regarding sport catches in BC (Fig. 4.2). DFO Pacific Region conducts annual creel and logbook surveys (supplemented by aerial observations) to calculate recreational catch. These estimates can represent as little as one third of the total

amount estimated by mail-out surveys, conducted every five years by the DFO Statistical Services Branch. The disagreement is likely due to differences in methodology, as the Pacific Region's creel estimates do not account for landings in many ports and do not capture activity on the shore or at private docks.

The rigor with which creel surveys are conducted has declined in recent years as fewer financial and human resources are being dedicated to the process. Moreover, the actual coverage for all creels is unknown, and is only assumed to be complete (K. Brickley, DFO Statistical Services Branch. Ottawa *pers. comm.*). Forrest (2002) discusses the discrepancy further. I therefore assume that the Pacific Region sport catch estimates represent the lower bound of the possible catch range, while the Statistical Services Branch estimates represent the upper bound – the difference being called 'unreported'. Both datasets refer to pieces retained, so I converted to wet weight using ratios reported in Appendix 4.4; Table A4.4.1.

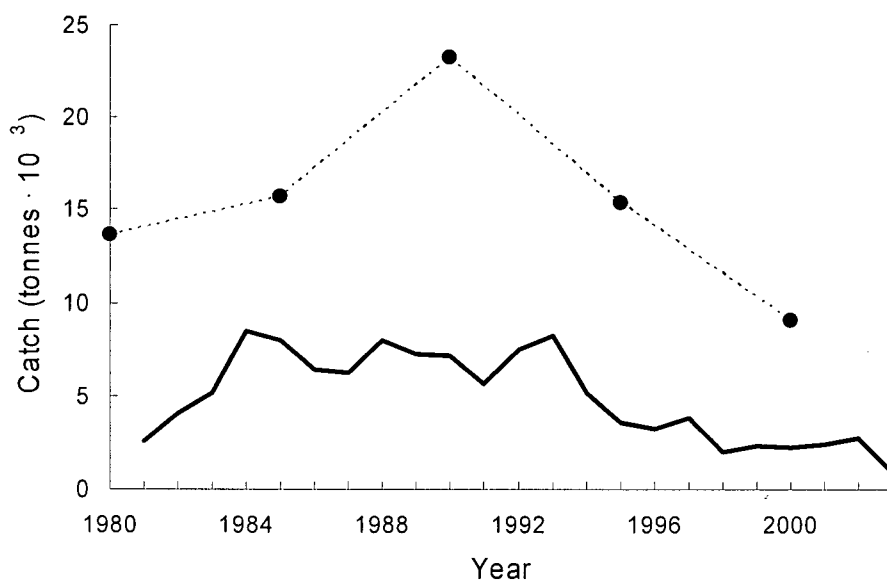


Figure 4.2 Salmon recreational catch estimates. Solid line shows DFO Pacific Region estimates from creel surveys and logbooks, dotted line shows DFO Statistical Services Branch estimates from mail-out surveys.

Table 4.1 Incentive ratings. Grey cells indicate periods for which anchor points exist. H = high; M = medium; L = low.

IUU Category	Target Sp.	Fleet	Period										
			1950-1954	1955-1959	1960-1964	1965-1969	1970-1974	1975-1979	1980-1984	1985-1989	1990-1994	1995-1999	2000-2003
Discards	Salmon	Gillnet	H	H	H	H	H	H	H	H	M/H	L/M	L
		Troller	H	H	H	H	H	H	H	H	M/H	L/M	L
		Seine	H	H	H	H	M/H	H	H	H	M/H	L/M	L
	Groundfish	Recreational	M/H	H	H	H	H	H	H	H	H	L/M	L
		Trawl	L	L/M	M/H	M/H	M/H	M/H	M/H	M	M/H	L/M	L/M
		Hook and line	L	L/M	L/M	M/H	M	M	M	M	M/H	M/H	M/H
		Recreational	L/M	M/H	M/H	M/H	M	M/H	M/H	M/H	H	M	L/M
Illegal	Salmon	H	H	H	H	H	H	H	H	H	M/H	L/M	L
	Groundfish	All	H	H	H	H	H	H	H	H	M/H	L	L
Unreported	Salmon	Recreational	H	H	H	H	H	H	H	H	H	L/M	L

Table 4.2 Anchor point range. Values show percentage (%) of catch per gear type.

IUU Category	Target Sp.	Fleet	Refa	Period										
				1950-1954	1955-1959	1960-1964	1965-1969	1970-1974	1975-1979	1980-1984	1985-1989	1990-1994	1995-1999	2000-2003
Discards	Salmon	Gillnet	190,193,203								5.0	6.0		0.0 – 10.0
		Troller	190								5.0	5.0		
		Seine	190								5.0	5.0		
		Recreational	194									7.6	7.6	
	Groundfish	Trawl b c d	182-190,203		4.3 – 25.0	7.7 – 25.0	12.0 – 26.1	8.3 – 26.8	6.8 – 25.0	5.9 – 22.2	6.6 – 21.5	3.2 – 17.5	10.3	
		Hook and linee	190-192							16.1 – 27.4	26.0 – 50.0	15.7 – 75.3	46.4 – 75.3	
		Recreationalf	203										21.0	
Illegal	Salmon	All	140-164,193								0.1 – 0.15	0.1 – 0.15	0.1 – 0.15	
	Groundfish	All	165-181								0.04	0.04	0.04	
Unreported	Salmon	Recreational	195-201						169	120	246	415	332	

a References listed in Appendix Tables 8.1,3-5

b Low estimate from 1962-1986 includes information from shrimp trawl fishery; we therefore assume GF trawl has similar discard/target ratio as shrimp trawl.

c Low estimate from 1974-1990 based on halibut bycatch and halibut landings; we therefore assume other groundfish have similar discard/landing ratio as halibut

d High estimate from 1960 to 1969 based on subsequent decade

e Low estimate from 1992-1994 Includes information from Bering Sea; 1988-1989 based on halibut hook and line;

f Datum from Oregon coast

Table 4.3 Mean reported catch. Values reported in tones (t).

IUU Category	Target Sp.	Fleet	Period										
			1950-1954	1955-1959	1960-1964	1965-1969	1970-1974	1975-1979	1980-1984	1985-1989	1990-1994	1995-1999	2000-2003
Discards	Salmon	Gillnet								1.0	1.5		0.0 – 2.7
		Troller								1.1	1.0		
		Seine								2.1	1.7		
	Groundfish	Recreational									1.7	2.5	
		Trawl			3.8 – 22.1	6.8 – 22.1	10.6 – 23.0	7.7 – 24.8	6.3 – 23.2	4.7 – 17.7	5.9 – 19.0	2.8 – 15.5	9.1
		Hook and line								1.1 – 1.8	2.8 – 5.5	1.7 – 8.2	5.1 – 8.2
		Recreational											2.3
Illegal	Salmon	All								0.1 – 0.15	0.1 – 0.15	0.1 – 0.15	
	Groundfish	All								0.04	0.04	0.04	
Unreported	Salmon	Recreational							169	120	246	415	332

Addressing uncertainty

Once I identify a likely range for the quantity of IUU using the subjective procedure (i.e., a lower and upper bound), I employ a Monte Carlo technique to estimate the mean of missing catch with error for each time period. The true amount of missing catch (X) will fall somewhere in the estimated range between the lower bound (A) and the upper bound (C) in eq. 4.1,

$$P[A \leq X \leq C] = \int_A^C f(X) dX = 1 \quad \text{Equation 4.1}$$

For values of X between A and C , the probability density function $f(X)$ of the triangular distribution is given by eq. 4.2,

$$f(X) = \begin{cases} \frac{2(X-A)}{(C-A)(B-A)} & \text{if } A \leq X \leq B \\ \frac{2(C-X)}{(C-A)(C-B)} & \text{if } B \leq X \leq C \end{cases} \quad \text{Equation 4.2}$$

B is the 'best guess'; the mode of the distribution. Sampling 5000 times, the Monte Carlo empirically determines the mean and 95% confidence intervals (Fig. 4.3).

Best guess

A 'best guess' estimate could be found only for groundfish trawl (Fig. 4.4), as there were several independent time series of discards available for that fishery. The lower and upper bounds and the best guess do not represent contiguous data series; they are each composites of three or more data series. The lower and upper bounds are set respectively by the largest and smallest estimate of discards found in the literature for a given year. The 'best guess' is based on an intermediate estimate, if available.

In my calculations, the 'best guess' estimate for groundfish trawl was extended to other groundfish fleets in proportion to their respective (independently scored) absolute ranges. I therefore assumed that the annual discard trend for trawl could be applied to other groundfish fleets, and that years of high discarding in trawl would correspond to years of high discarding in hook and line and the recreational fishery.

Sufficient data is not available to provide a 'best guess' estimate for salmon discards, salmon illegal catch, or groundfish illegal catch. Therefore, the 'best guess' for these IUU elements is said to be 20% of the total range offered by their lower and upper bounds (i.e., the Monte Carlo draws from an error distribution skewed to the right; 20% of the error falls to the left of the mode, see Fig. 4.3). I therefore assume that if the true quantity of missing catch is less than the best guess, then the amount will not be too far off, but if the true value is greater than the best guess, it is liable to be much greater. In other words, I am certain that there is at least an appreciable amount of missing catch. For unreported catch in the salmon recreational fleet, the 'best guess' is assumed to fall in the middle of the possible range, so that the Monte Carlo draws from a symmetrical distribution. Considering the potential magnitude of missing catch, I did not feel that the 'best guess' ought to be conservative.

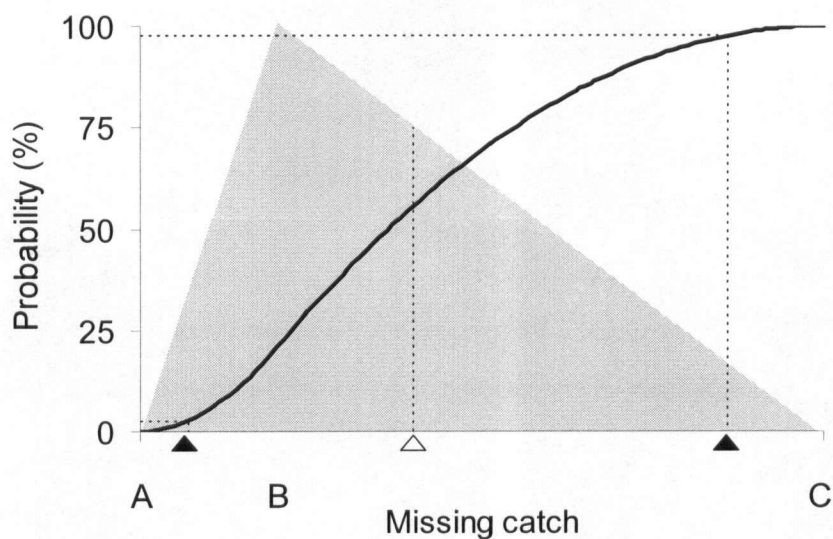


Figure 4.3 Cumulative probability distribution of missing catch. Line shows probability distribution. A) Lower bound; B) 'Best guess'; C) Upper bound. Triangle distribution provided for comparison (shaded area); the height of the triangle is $2/(C-A)$. Monte Carlo empirically estimates the mean (open arrow) and 95% confidence limits (closed arrows). The example distribution shows the error assumption used for most fisheries, where 20% of the error falls to the left of the mode.

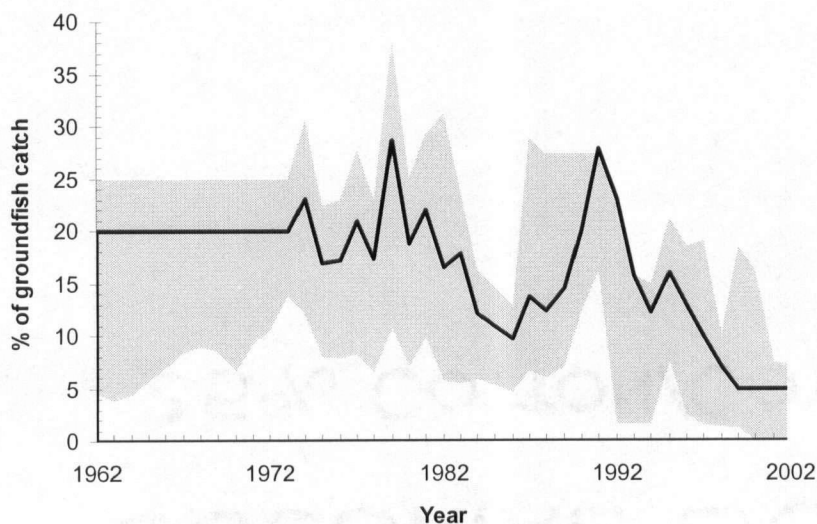


Figure 4.4 Likely range of groundfish discards. Shaded area shows full range of estimates available in the literature; black line shows 'best guess'.

4.3 Results

Determining absolute quantities of missing catch

Assigning absolute quantities to the incentive ratings is the most subjective component in this methodology - and one that will require additional refinement and review by experts if such an analysis is to contribute to management. Based on the range of discarding, illegal and unreported catch rates described by the anchor points, Table 4.4 defines the absolute quantity of missing catch for each incentive rating. The three IUU categories, discard, illegal and unreported catch, are treated independently.

Numbers listed in bold are anchor points based on one or more sources from the literature (values are taken from representative periods in Table 4.2). Numbers listed in italics are scaled based on these available estimates, such that the influence 'medium-high' represents 80% of the upper bound, 'medium' is 60%, 'low-medium' is 40 %, and 'low' is 20%.

Treating all salmon gear types the same, I assume that each sector will conform to this range, defined by an upper bound of 2.7% discards per weight of total catch, and 0.19% illegal catches per weight of total catch. Similarly, I assume that the unknown catch occurring in the groundfish fishery is comparable among sectors, and contained within an upper bound of 24.8% for discards, and 0.19% for illegal catch. The unreported catch range (upper bound 246%) refers to the recreational salmon sector only. However, the majority of 'unreported' catch for commercial salmon and groundfish fleets is probably considered within the calculation of discards. There will be some degree of overlap between categories.

Table 4.5 shows the product of the subjective portion of the methodology, a minimum and maximum estimate of missing catch for each period and gear type. Missing catch is presented as a percentage of known catch for each fleet. These values are converted into absolute quantities using catch statistics in Table 4.3 for input into the Monte Carlo.

Table 4.4 Absolute ranges of IUU catch rate for each incentive rating. Values show percentage (%) IUU per weight of target catch. Bold numbers are from literature (rated low to high based on time series), numbers in italics are scaled based on bold number.

IUU Category	Influences	Scaling factor	Salmon	Groundfish
Discard	H	1.0	2.7%	24.8%
	M/H	0.8	<i>2.2%</i>	<i>19.8%</i>
	M	0.6	<i>1.6%</i>	<i>14.9%</i>
	L/M	0.4	<i>1.1%</i>	<i>9.9%</i>
	L	0.2	<i>0.5%</i>	<i>5.0%</i>
Illegal	H	1.0	<i>0.19%</i>	<i>0.19%</i>
	M/H	0.8	0.15%	<i>0.15%</i>
	M	0.6	<i>0.11%</i>	<i>0.11%</i>
	L/M	0.4	<i>0.08%</i>	<i>0.08%</i>
	L	0.2	<i>0.04%</i>	0.04%
Unreported	H	1.0	246%	—
	M/H	0.8	<i>197%</i>	—
	M	0.6	<i>148%</i>	—
	L/M	0.4	<i>98%</i>	—
	L	0.2	<i>49%</i>	—

Table 4.6 shows the output of the Monte Carlo, the mean estimate of missing catch with 95% confidence limits in each category of IUU, by period and gear type.

Fig. 4.5 shows the estimated catch missing from official statistics for all BC salmon and groundfish fisheries. Discards are low in the pelagic salmon fishery compared to the demersal fleet, but discards for both salmon and groundfish are currently on the decline. Illegal catch is small in both fisheries compared to total IUU extractions, although the estimate is conservative. Missing catch from the salmon recreational fishery (called 'unreported') is not shown, but it comprises the large majority of total salmon IUU (lower-left graph).

Fig. 4.6 shows total estimated extractions from BC salmon and groundfish fisheries. The black area shows the official reported catch and the grey area shows the upper limit of missing catch at 95% confidence. IUU catch is currently on the decline for both salmon and groundfish sectors in proportion to recorded weight and in absolute terms. In 2000, catch missing from official salmon statistics appears negligible overall, but catch missing from groundfish records may be a concern.

Table 4.5 Monte Carlo input: IUU catch range. Values show percentage (%) of fleet catch by weight.

IUU Category	Target Sp.	Fleet	Period										
			1950-1954	1955-1959	1960-1964	1965-1969	1970-1974	1975-1979	1980-1984	1985-1989	1990-1994	1995-1999	2000-2003
Discards	Salmon	Gillnet	2.2 - 2.7	2.2 - 2.7	2.2 - 2.7	2.2 - 2.7	2.2 - 2.7	2.2 - 2.7	2.2 - 2.7	2.2 - 2.7	1.6 - 2.2	1.1 - 1.6	0.5 - 1.1
		Troller	2.2 - 2.7	2.2 - 2.7	2.2 - 2.7	2.2 - 2.7	2.2 - 2.7	2.2 - 2.7	2.2 - 2.7	2.2 - 2.7	1.6 - 2.2	1.1 - 1.6	0.5 - 1.1
		Seine	2.2 - 2.7	2.2 - 2.7	2.2 - 2.7	2.2 - 2.7	1.6 - 2.2	2.2 - 2.7	2.2 - 2.7	2.2 - 2.7	1.6 - 2.2	1.1 - 1.6	0.5 - 1.1
	Groundfish	Recreational	1.6 - 2.2	2.2 - 2.7	2.2 - 2.7	2.2 - 2.7	2.2 - 2.7	2.2 - 2.7	2.2 - 2.7	2.2 - 2.7	2.2 - 2.7	1.1 - 1.6	0.5 - 1.1
		Trawl	0 - 5.0	5.0 - 9.9	14.9 - 19.8	14.9 - 19.8	14.9 - 19.8	14.9 - 19.8	14.9 - 19.8	9.9 - 14.9	14.9 - 19.8	5.0 - 9.9	5.0 - 9.9
		Hook and line	0 - 5.0	5.0 - 9.9	5.0 - 9.9	14.9 - 19.8	9.9 - 14.9	9.9 - 14.9	9.9 - 14.9	9.9 - 14.9	14.9 - 19.8	14.9 - 19.8	14.9 - 19.8
		Recreational	5.0 - 9.9	14.9 - 19.8	14.9 - 19.8	14.9 - 19.8	9.9 - 14.9	14.9 - 19.8	14.9 - 19.8	14.9 - 19.8	19.8 - 24.8	9.9 - 14.9	5.0 - 9.9
Illegal	Salmon	All	0.15 - 0.19	0.15 - 0.19	0.15 - 0.19	0.15 - 0.19	0.15 - 0.19	0.15 - 0.19	0.15 - 0.19	0.15 - 0.19	0.11 - 0.15	0.04 - 0.08	0 - 0.04
	Groundfish	All	0.15 - 0.19	0.15 - 0.19	0.15 - 0.19	0.15 - 0.19	0.15 - 0.19	0.15 - 0.19	0.15 - 0.19	0.15 - 0.19	0.11 - 0.15	0 - 0.04	0 - 0.04
Unreported	Salmon	Recreational	197 - 246	197 - 246	197 - 246	197 - 246	197 - 246	197 - 246	197 - 246	197 - 246	197 - 246	49 - 98	0 - 49

Table 4.6 Monte Carlo output: Mean IUU catch with 95% confidence intervals. Values in tonnes (t).

IUU Category	Target Sp.	Fleet	95% CI	Period											
				1950-1954	1955-1959	1960-1964	1965-1969	1970-1974	1975-1979	1980-1984	1985-1989	1990-1994	1995-1999	2000-2003	
Discards	Salmon	Gillnet	Upper	895	681	629	638	751	444	393	565	489	177	107	
			Mean	818	621	574	583	685	404	359	515	426	151	79	
			Lower	763	579	535	542	638	377	335	479	379	132	58	
		Troller	Upper	307	295	301	432	453	453	486	592	422	106	21	
			Mean	281	269	275	395	413	413	443	540	366	90	15	
			Lower	262	251	256	368	385	386	412	504	325	79	11	
	Seine		Upper	955	650	557	525	565	635	823	1232	784	282	163	
			Mean	874	595	508	479	491	580	752	1127	680	240	119	
			Lower	814	554	473	446	438	541	700	1049	607	210	87	
		Recreational	Upper	87	81	149	159	191	153	135	188	177	46	21	
			Mean	76	74	136	146	174	140	123	172	161	39	15	
			Lower	68	69	126	136	162	130	115	160	150	34	11	
	Groundfish	Trawl	Upper	849	1589	4020	4010	2943	4290	8297	6997	13542	8044	7472	
			Mean	397	1197	3643	3631	2658	3927	7358	5707	12233	6709	5633	
			Lower	70	919	3234	3225	2363	3440	6642	4904	10861	4815	4314	
		Hook and line	Upper	98	210	254	526	288	253	329	592	1666	1986	1907	
			Mean	46	159	192	465	241	212	275	496	1471	1754	1685	
			Lower	8	122	149	421	207	183	238	427	1329	1587	1523	
		Recreational	Upper	-	-	-	-	-	-	257	268	164	127	77	
			Mean	-	-	-	-	-	-	227	235	148	107	55	
			Lower	-	-	-	-	-	-	205	213	137	92	42	
Illegal		Salmon	All	Upper	159	120	115	123	147	118	129	181	125	30	11
				Mean	143	108	103	111	132	106	116	163	109	22	5
				Lower	132	99	95	102	122	98	107	150	98	17	1
		Groundfish	All	Upper	36	36	43	43	32	44	84	99	114	32	32
				Mean	33	32	39	39	29	40	76	89	100	15	15
				Lower	30	30	36	36	26	37	70	83	89	3	3
Unreported	Salmon	Recreational	Upper	9886	7432	13599	14592	17437	14015	12315	17181	16173	2778	909	
			Mean	9107	6845	12510	13434	16048	12887	11334	15818	14898	2207	513	
			Lower	8327	6256	11422	12290	14658	11770	10362	14435	13611	1629	114	

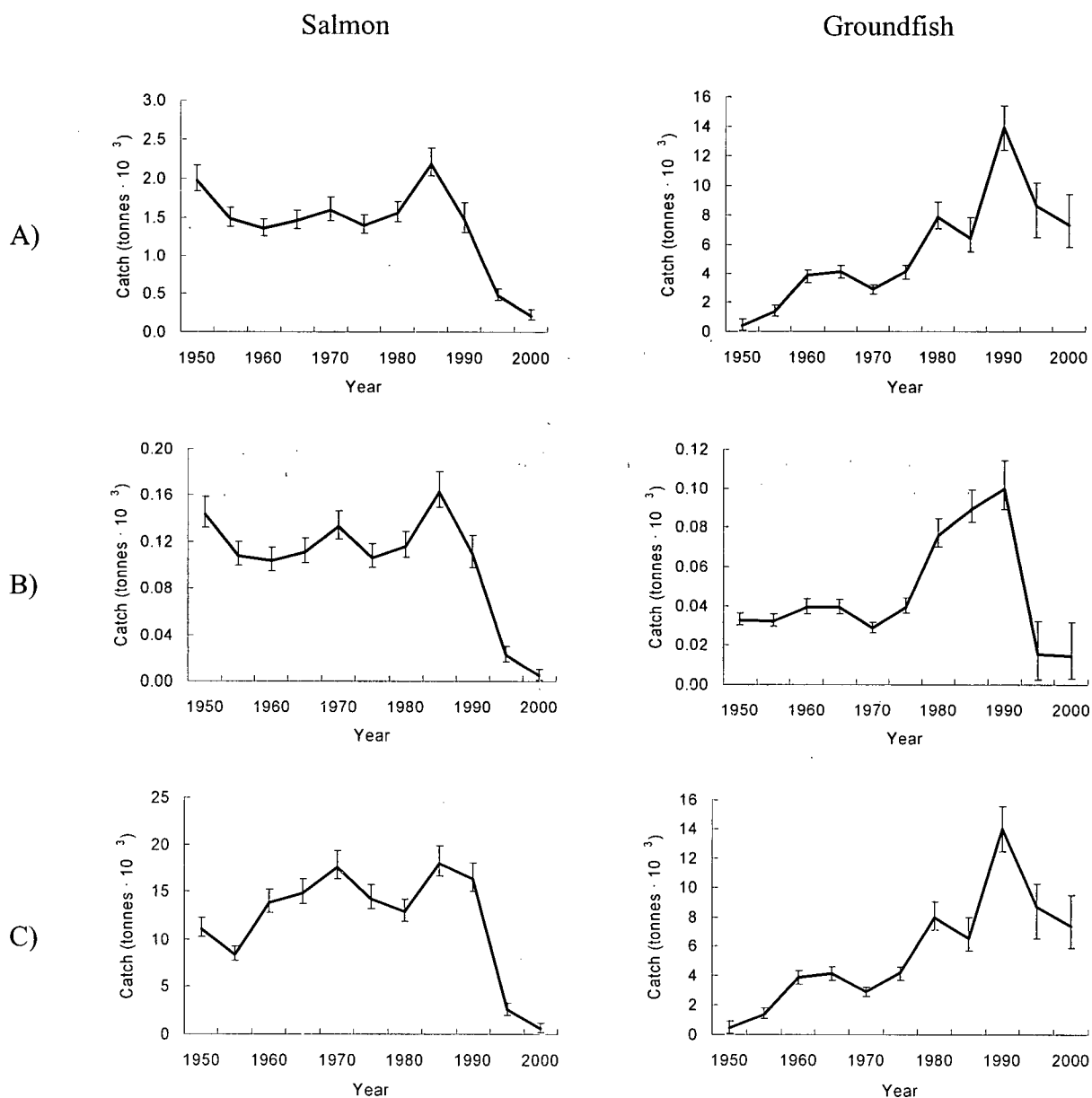


Figure 4.5 Estimates of missing catch for salmon and groundfish fisheries. Line shows the mean of 5000 Monte Carlo samples with 95% confidence intervals. A.) Discards; B.) Illegal catch; C.) Total IUU catch. Salmon (C) also includes unreported recreational catch.

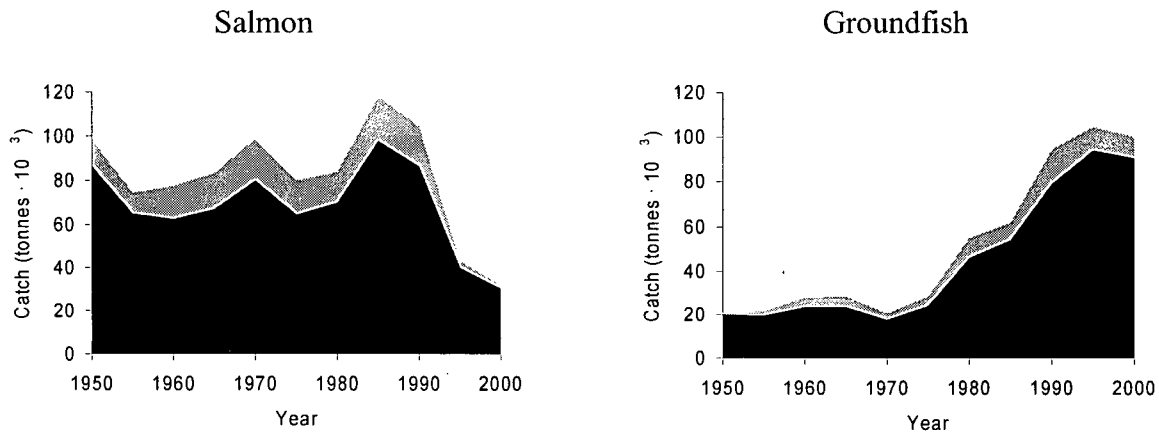


Figure 4.6 Total estimated extractions in BC salmon and groundfish fisheries. Black area shows official reported catch; grey area shows upper estimate of IUU catch at the 95% confidence limit.

4.4 Discussion

Discards

Discards in the salmon fleet appear small in comparison to the reported catch. The highest values in the literature suggest they are around 5% to 6% of total landings. This analysis suggests that discards hovered around 2.2% until the mid eighties, when they began to drop to current levels of less than 1%. Major political changes in the fishing industry would have contributed to this reduction, but the 1980s also saw technical changes in the way people fish for salmon. Gear modifications were introduced, like weedlines in gillnets and brailing boards in the seine fishery, and used in conjunction with new techniques to improve selectivity and reduce interception of non-target species. Ongoing experiments to improve gear selectivity, and the revival of effective traditional techniques in terminal fisheries (e.g., weirs, fish wheels) may be expected to reduce discarding even further in the future. Generally, the nature of pelagic salmon fisheries allows fishers to avoid bycatch more easily than the demersal fleet.

After shrimp trawl, the most unselective fisheries in BC are for groundfish. At its worst (from 1980-1985), I estimate that the hook and line fleet discarded a weight of fish equivalent to 22.6% of its recorded catch, and the trawl fleet (at its worst between 1975 and 1980) discarded about 17.8%. Unlike the fisheries for Pacific salmon, groundfish operations have seen a steady

increase in landings since the 1950s. As effort increases, so does incidental catch. However, in recent years, bycatch reduction initiatives have seen some success. Mesh size regulations, exclusion panels, grates, unhooking techniques and species-selective baits have been used to reduce incidental capture. Thanks to these modifications, I estimate that discards are now at their lowest levels in 20 years, despite a three-fold increase in groundfish landings.

Illegal catch

Illegal catch probably constitutes a very small portion of IUU in BC fisheries. The incentives and opportunities to poach in both salmon and groundfish fleets remained stable, according to this review, until about the 1990s. Observer programs began in this period for several fisheries, but more importantly, the significant quantity of illegal catch that had been occurring in an unofficial commercial aboriginal fishery was made legitimate by political changes (e.g., Sparrow decision, pilot sales program) (Wappel, 2003). BC salmon stocks are not what they once were however, and we may expect to see an increase in illegal catch in the coming years as restoration efforts restrict lawful fishing opportunities. This is certainly the case with northern abalone (*Haliotis kamtschatkana*), in fact rampant poaching has been blamed for preventing its recovery in BC (DCI, 1999).

Unreported catch

In recent decades, the recreational sector accounted for only 8 percent of salmon landings throughout the province (Table 4.3). Although the gross quantity of unreported salmon catch in the sport fishery may be small compared to commercial extractions, there exists a wide-spread (and probably well-founded) perception among commercial fishers that the recreational sector is not subject to the same strict accounting as the rest of the fleet (C. Ainsworth. *pers. obs.*). Data entry is not standardized, reporting is often voluntary, and large data gaps exist in the catch record – these factors combined make the estimate of unreported catch significant. The results here suggest that unreported sport salmon catches may have exceeded 220% of the official statistics until the mid 1990s. I estimate that since that time unreported catch has been reduced by almost an order of magnitude relative to recorded landings. As salmon stocks continue to

decline, particularly troubled chinook populations that have been the mainstay of anglers, the absolute quantity of missing catch may be expected to fall.

This analysis of unreported catch could be extended to include commercial salmon fisheries using a back-calculation technique employed by other authors. Patterson *et al.* (1990) estimated the unreported catch of the Ecuadorian tropical chub mackerel (*Scomber japonicus*) based on the output of fishmeal factories, and Castillo and Mendo (1987) estimated catch for Peruvian anchovy (*Engraulis ringens*) using the same method. In BC, packing records extending back to the 1950s are available from industry (BCSMC, 2004), and could be compared with official catch statistics to provide an estimate of unreported catch throughout the years.

Although this study has not attempted to quantify unreported recreational catch in the groundfish sector, it is worth noting that there is a discrepancy of almost 20-fold between the DFO Pacific Region creel estimates of recreational rockfish landings (22 tonnes in 2000), and the DFO Statistical Services Branch estimate (400 tonnes) (K. Brickley, DFO Statistical Services Branch. Ottawa. *pers. comm.*). Although the gross amount is small compared to commercial rockfish landings, unreported catch could be an important factor in this fishery considering the vulnerability of rockfish stocks to overexploitation, and the current drive to reduce landings. On the other hand, recreational catch of halibut seems to be well recorded. Catch records between Washington State (USA) data and DFO national survey results agree within 3% for the number of fish landed in Canadian waters.

Limitations to this methodology

The nature of IUU catch demands a subjective method for quantification. The most approximate step in this methodology, and the one most likely to arouse scrutiny, comes when I assign an absolute range of missing catch for each low, medium or high rank in the relative incentives table. Certainly, the approximation could benefit from further discussion with experts in BC fisheries, particularly with experts involved in each gear class. With their contribution, the general trend suggested by the influence table could be applied more relevantly to each gear type.

Also, it is difficult under this methodology to estimate the species-specific composition of IUU catch. In this chapter, I have aggregated all species together by weight to produce an estimate of IUU catch per tonne of fish landed. In fact, anchor points recorded cannot distinguish even target species discards (e.g., juveniles) from non-target fish or invertebrates. It would be possible to refine the estimate into species-specific categories of discards and illegal catch, but each series would require its own anchor points.

The estimation procedure presented here can be easily updated as more and better information becomes available. The analysis could be expanded to include additional anchor points, and improved 'best guess' estimates to define the distribution of likely error. There is still unused potential in the influences table; however, the availability of anchor points is limiting.

IUU in British Columbia

As fishing power increased from the 1950s to 1970s, IUU catch increased in proportion to recorded landings for the major fisheries in BC. During the 1970s, operational changes began to take effect that would begin to counteract the improved catching ability of fishers, and regulate fisheries that were previously open access. Exploitation increased throughout the 1980s, and the largest quantities of missing catch probably occurred in the early 1990s for most fleets. It was not until the mid 1990s when IUU catch began to fall in proportion to landings, but these days enforcement and monitoring are strict, and new regulations have likely reduced the amount of catch missing from official statistics.

From about the 1950s to the 1980s, I estimate that between 10 and 20 thousand tonnes of catch went unrecorded every year in the BC salmon and groundfish fisheries. IUU catch increased throughout the 1980s, and by 1990, the amount was probably closer to 30 thousand tonnes per year. That is equivalent to 18% of recorded landings. By 2000, IUU catch appears to have fallen to about 8 thousand tonnes per year, or 6.6% of landings. The influences table suggests that the downward trend has continued to the present for salmon fleets, due to better enforcement and data collection, but may have leveled off for the groundfish fleets. Continued fishery

closures and routine reductions in the legal capture size restrict the amount of catch that can be landed - encouraging discards. Despite compulsory observer programs, I suggest that wide scale implementation of the quota system in groundfish fisheries has also increased the motivation to high grade.

Global IUU issues

This method provides a simple and quick way to quantify IUU catch, and it can be done under data-limited conditions. It may therefore prove to be a useful tool for addressing the IUU problem on a global scale. So far, the methodology has been applied to ten countries or jurisdictions worldwide, and the estimates of missing catch are being integrated into the Sea Around Us Project (SAUP) Global IUU Database (R. Watson and T.J. Pitcher, UBC Fisheries Centre, Vancouver. *pers. comm.*). The database also currently contains 21591 individual entries; quantitative and semi-quantitative reports of missing catch indexed by year, country, gear type, taxon and other fields. The collected information should make it easier for future authors to apply this methodology to other parts of the world, while the new technique will offer a rigorous way to incorporate subjective information and expert knowledge into the database.

Future work

A significant ecological concern in BC is the effect that incidental catch and discarding is having on inshore rockfish populations (*Sebastes* spp.). Slow-growing and sensitive to the effects of overfishing, their numbers are in decline (Yamanaka and Lacko, 2001). In addition to directed harvests, mortality is kept high by incidental capture in both salmon and groundfish fisheries. Due to their anatomy, bycatch mortality can be a significant problem (Yamanaka and Lacko, 2001), and needs to be considered in setting the total allowable catch for directed fisheries. However, the quantity of discards is not well known. As with most groundfish, discards have only been recorded in logbooks since 2001; and compliance is poor (Haigh *et al.*, 2002). An emerging fishery for premium live rockfish (active in BC since the 1970s, but now gaining momentum; Love *et al.*, 2002), has been very poorly recorded because of the fragmentary nature of the fishery, and a drive for fishers to deliver the product quickly to market (Stevens, 2003).

This species group could benefit from a dedicated IUU analysis, although qualified anchor points may be difficult to find.

Northern abalone (*Haliotis kamtschatkana*) is important to First Nations culture, and was once a significant economic resource in northern BC. They have been listed by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) as 'threatened', and despite a coast-wide fishery closure since 1990, populations have shown no hint of recovery (Campbell, 2000; Lessard *et al.*, 2002). They are subject to a widespread black market fishery, and poaching has been blamed for the failure of the stock to recover (DCI, 1999). An IUU analysis using the present methodology would provide a rough estimate of illegal removals, and help us to understand whether poaching has sabotaged rebuilding efforts. Records of abalone confiscated in enforcement operations litter the DFO news releases. They could be used to produce an absolute series of estimates for anchor points, following the technique used here for illegal salmon and groundfish captures.

5 MODELING THE PAST AND PRESENT

We believe the food web modeling approach is hopeless as an aid to formulating management advice; the number of parameters and assumptions required are enormous.

Hilborn and Walters (1992) p 448.

5.1 Introduction

Four EwE models are developed to represent the marine ecosystem of northern BC at different periods in the past. The models represent a 'best guess' of what the historic ecosystems may have looked like. The historic periods were chosen to represent distinct eras in the development of west coast fisheries. These periods are prior to European contact (c. 1750), before the introduction of steam trawlers (c. 1900), during the peak of the Pacific salmon fishery (1950), and in the present day (2000). This chapter describes Ecopath functional groups for northern BC, and provides rationale and data sources for parameterization. Species that compose each group are listed in Appendix Table A5.1.1. Basic parameters for all groups are listed in Appendix Table A5.1.2, and diet composition is provided in Appendix Table A5.1.3.

History of northern BC models

The northern BC EwE models have had a long lineage, and this volume represents the fifth major revision. A present day model representing the marine system of Hecate Strait was developed by Beattie (1999) following a BTF workshop that assembled expert knowledge on the ecosystem and tried to quantify historical changes since the pre-contact period. The Hecate Strait model of Beattie (1999) was largely based on a southern BC shelf model (Pauly and Christensen, 1996). Major revisions were made by Beattie (2001), who increased the study area from 46,000 km² to 70,000 km², and included the waters of Dixon Entrance and Queen Charlotte sound. That area delineation has been preserved to the current version. Beattie (2001) also increased the number of functional groups from 25 to 49, including split-pool juvenile groups.

He expanded the salmon group to consider resident and transient populations separately, and he diversified rockfish and invertebrate groups. He also added a group for fishery discards. The next revision came from Vasconcellos and Pitcher (2002i) based on proceedings of a second BTF workshop. They increased the number of functional groups to 53, the current number. They also created preliminary models of 1750 and 1900 based on expert opinion. Ainsworth *et al.* (2002) revised the northern BC models again, particularly the fisheries, adding bycatch and including a new recreational fishing fleet, new costs and market prices. They also developed a preliminary 1950 model. In the current volume, basic model structure is maintained from Ainsworth *et al.* (2002), but parameters are revised and dynamics are greatly improved. A preliminary Ecospace model was created for demonstration at a Haida Gwaii workshop, organized by the Fisheries Centre and World Wildlife Fund. The spatial model was used to perform predictions regarding fishery closures in the proposed Gwaii Haanas National Marine Conservation Area (NMCA), and is included here as a demonstration of Ecospace in northern BC (Appendix 8).

EwE Model groups

Sea Otters

The pre-contact sea otter (*Enhydra lutris*) population is estimated to have been between 100,000 and 150,000 animals (Kenyon, 1975) before they were decimated by the fur trade (Nichol *et al.* 2003). Vasconcellos and Pitcher (2002a) estimated that there are 5,000 animals in northern BC by assuming that area covers 1/20th of the sea otter range. With an average weight of 22.4 kg (Bodkin *et al.*, 1998) the density of sea otters in the pre-contact period may have been about 1.6 kg·km⁻²; this value was used for the 1750 model. Remnants of the population were represented in the 1900 model with a biomass of 0.1 kg·km⁻². Although sea otters were extirpated by the 1950s, a small biomass value of 0.05 kg·km⁻² was given to that functional group to maintain a common model structure among periods. By 2000, the population is assumed to be back to the 1900 level of abundance, 0.1 kg·km⁻². This value agrees with a recent estimate by Nichol *et al.* (2003) if we can assume that one eighth of the BC population occurs in central and northern BC. Biomass accumulations were taken as zero for the 1750 and 1900 models, and a small (2%) increase per year was assumed in the 1950 model. With this base rate of population growth,

dynamic simulations beginning in 1950 can forecast the population increase mentioned by Riedman and Estes (1998). They suggested that sea otter populations grew at a rate of about 15% per year during the early phase of their reintroductions (late 1960s); dynamic predictions are also consistent with the biomass trend suggested by LEK interviews (Ainsworth and Pitcher, 2005a)⁵.

Bodkin *et al.* (1998) estimated an instantaneous mortality rate of 0.13 yr^{-1} , based on an average age of 7 years in the Prince William Sound (PWS) area; Okey and Pauly (1999) used this value for PWS. Vaconcellos and Pitcher (2002a) assumed that the mortality was representative of the production rate, P/B , and I extended that estimate to all periods. Consumption rate is based on Riedman and Estes (1998); it is assumed to be between 23 and 33% of body mass per day for adults. I used the average value, adjusted to an annual rate for Q/B in all periods (101.5 yr^{-1}). Sea otter diet was based on Riedman and Estes (1998), who suggested that it consists of 50% epifaunal invertebrates; also crabs, shallow water benthic fish, juvenile pollock and squid.

Pre-contact fishing mortality is based on Irwin (1984), who reported that First Nations hunted sea otters with harpoons and clubs. I assume that sustainable harvest rates were used in 1750, and so matching the population growth rate of 2.5% per year from Kenyon (1975). This equates to $0.2 \text{ kg} \cdot \text{km}^{-2}$. Sea otter kills in the other time periods are assumed to be negligible.

To represent sea otter dynamics accurately throughout recent decades it would be necessary to apply a time forcing pattern to represent the critical influence of reintroductions, or drive dynamics of the group using a biomass time series. This is not attempted. Neither were mediating functions applied that could capture keystone dynamics, such as sea otter interactions with kelp and urchins (*vis.* Steneck *et al.*, 2002).

⁵ BTF Interview and historical archive database. Searchable online: [www.fisheries.ubc.ca/projects/btf/]. Contact: C. Ainsworth, UBC Fisheries Centre, Vancouver, BC.

Mysticetae

The baleen whales include the blue whale (*Balaenoptera musculus*), fin whale (*B. physalus*), sei whale (*B. borealis*), humpback whale (*Megaptera novaeangliae*), right whale (*Eubalaena australis*), and gray whale (*Eschrichtius robustus*). GREGG (2002) gives historic population estimates for baleen whales and sperm whales. Using the mean weight per species from Trites and Pauly (1998), the biomass of the *Mysticetae* was calculated in 1750, 1900 and 2000 as 2.67 t·km⁻², 1.54 t·km⁻² and 1.34 t·km⁻², respectively. The value for 1950 calculated by this method was too low. Although in preliminary simulations of the 1950 model the collapse of baleen whales did follow expectations under historic fishing mortalities (such that whale biomass had been seriously depleted by 1967 when whaling was banned), the slow rate of production for this group would not thereafter permit the pace of recovery needed to rebuild biomass to 2000 levels and agree with time series. I therefore assume that baleen whale biomass had been higher in the 1950s than originally estimated, and has not yet recovered to that level in the present. A new value is adopted for 1950: 120% of the present-day estimate or 1.60 t·km⁻². I accept a small negative biomass accumulation rate in the 1950 model (-0.014 t·km⁻²) to influence Ecosim dynamics, and a small positive value in 2000 (0.004 t·km⁻²) based on the results of dynamic simulations using the 1950 model driven by historic production and mortality rates.

Trites and Heise (1996) suggested that the P/B ratio should be half of the 4% maximum rate of population increase, thus I use a P/B ratio of 0.02 yr⁻¹ in the 2000 model after Beattie (2001). That value is increased in the 1900 and 1950 models to 0.06 yr⁻¹ and 0.04 yr⁻¹ respectively to improve model fit to data. Trites and Heise (1996) suggested a Q/B ratio of 13 yr⁻¹ in summer and 5.1 yr⁻¹ in winter. For the 2000 and 1950 models I use the average (9.1 yr⁻¹), while for the 1900 and 1750 models, I use a value of 8 yr⁻¹ to represent an older population and one composed of larger blue and humpback whales present at that time. The diet of *Mysticetae* is adapted from Trites and Heise (1996).

First Nations people harpooned Gray whales according to Irwin (1984), and if one assumes that they caught about two per year then the calculation gives an annual pre-contact catch of 0.5 kg·km⁻². During the period 1908 to 1967, at least 18,643 baleen whales were caught and

processed at BC whaling stations at Sechart, Kyuquot and Coal Harbour in Vancouver Island, and Rose and Naden Harbours in the Queen Charlotte Islands (Gregar, 2002). Annual landings for 1900 are based on the records of Nichol and Heise (1992) for fin and humpback whales ($61 \text{ kg}\cdot\text{km}^{-2}$), and landings for 1950 are based on Gregor (2002) ($69 \text{ kg}\cdot\text{km}^{-2}$). I assume zero catch in the present-day. To incorporate the sizable amount of catch in 1950 stated by Gregor (2002), Ecosim dynamics demanded a small negative biomass accumulation ($<1\%$ per year). The result maintains an EE of 0.6, a reasonable figure for a highly migratory group.

Odontocetae

The toothed whales include the sperm whale (*Physeter macrocephalus*), Baird's beaked whale (*Berardius bairdii*), northern right whale dolphin (*Lissodelphis borealis*), Pacific white-sided dolphin (*Lagenorhynchus obliquidens*), Dall's porpoise (*Phocoenoides dalli*), harbour porpoise (*Phocoena phocoena*) and killer whale (*Orcinus orca*). Trites and Heise (1996) gave the number of toothed whales (excluding sperm whales) and average weight of each species in northern BC. The average sperm whale biomass is approximately 19 tonnes, and there are about 150 sperm whales according to Gregor (2002) giving a total biomass of $0.061 \text{ t}\cdot\text{km}^{-2}$. This value is used for present day and 1950. As in Beattie *et al.* (1999), I assume that the biomass of killer whales, dolphins and porpoises was about 20% larger during the early 1900s than at present time, and that the number of sperm whales was similar compared to today. Thus, the biomass in 1900 is estimated to be $0.066 \text{ t}\cdot\text{km}^{-2}$; I apply this value to 1750 as well.

The P/B of toothed whales is assumed higher than that of baleen whales, but lower than seals and sea lions. A P/B of 0.04 yr^{-1} is adopted for 1750, 1900 and 2000. P/B is lowered in 1950 to 0.02 yr^{-1} to fit population dynamics described in Preikshot (2005). Trites and Heise (1996) suggested a Q/B of 15.6 yr^{-1} for toothed whales in the summer and 15.3 yr^{-1} in the winter. The average is used for all periods (15.5 yr^{-1}).

For all periods, the diet of toothed whales is based on Beattie (2001). The Odontocetae group has a major influence on the system in this model and some changes were made in the diet composition to balance and fit dynamics, particularly with the 1900 model.

Seals and Sea lions

This group includes Steller sea lions (*Eumetopias jubatus*), harbour seals (*Phoca vitulina*), northern fur seals (*Callorhinus ursinus*) and northern elephant seals (*Mirounga angustirostris*). Biomass values for 1750 and 1900 are taken as $0.08 \text{ t}\cdot\text{km}^{-2}$, $0.069 \text{ t}\cdot\text{km}^{-2}$, respectively based on the suggestions of Vasconcellos and Pitcher (2002b). They used expert opinion to modify the present-day biomass estimate of Beattie (2001) for an estimate of historic abundance. Biomass estimates in the 1950 model ($0.13 \text{ t}\cdot\text{km}^{-2}$) and 2000 model ($0.27 \text{ t}\cdot\text{km}^{-2}$) are based on Preikshot (2005), who used BC sea lion data from Bigg (1985), Straight of Georgia seal data from Olesiuk (1999) and expert opinion. In all cases, abundance data is converted to wet weight using body mass figures supplied by Trites and Heise (1996). I assume that the BC and Straight of Georgia biomass density estimates can be applied to the study area without change.

Trites and Heise (1996) suggested that the maximum rate of population growth for pinnipeds is about 12%, and they assumed P/B was half that, 0.06 yr^{-1} . I use that value for all models except 1900, in which I increased P/B to 0.1 yr^{-1} to maintain the population under historic kill rates (Bigg, 1985). Trites and Heise (1996) estimated a Q/B for seals and sea lions of 15.3 yr^{-1} in summer and 14.8 yr^{-1} in winter, and I use the average in all models (15.1 yr^{-1}).

Seals and sea lions were hunted by First Nations people (Vasconcellos and Pitcher, 2002b), so I assume a catch of $0.1 \text{ kg}\cdot\text{km}^{-2}$ in the 1750 model. The 1900 catch value, $5 \text{ kg}\cdot\text{km}^{-2}$, is based on the average figure from 1912-1915 as supplied by Bigg (1985), with numbers killed converted into wet weight using data from Trites and Heise (1996). The same sources are used for catch in the 1950 model, although the year 1950 saw anomalously high landings compared to adjacent years. Rather than accept a spurious biomass accumulation in the 1950 base model, which would be required to reconcile that anomalous year, I instead assign catch as the average of surrounding years (1945-1955). This permits a more conservative biomass accumulation rate representing the long-term, $-0.009 \text{ t}\cdot\text{km}^{-2}$. The biomass accumulation rate is set to improve Ecosim's fit to time series.

The diet of seals and sea lions is adapted from Trites and Heise (1996) by assuming that 1/6th of the predation on forage fish is directed towards eulachon (Vasconcellos and Pitcher, 2002c);

other minor corrections were made to represent the assemblage in northern BC more accurately. However, seals and sea lions proved to be a controlling group in the ecosystem through simulation work, and so the diet composition is modified to adjust dynamics.

Seabirds

Seabirds present in northern BC include gulls (*Laridae*), grebes (*Podicipedidae*), Cassin's auklet (*Ptychoramphus aleuticus*), tufted puffin (*Fratercula cirrhata*), common murre (*Uria aalge*), rhinoceros auklet (*Cerorhinca monocerata*), marbled murrelet (*Brachyramphus marmoratus*), pigeon guillemot (*Cephus columba*), mergansers (*Mergus* spp.), pelagic cormorant (*Phalacrocorax pelagicus*), sooty shearwater (*Puffinus griseus*), northern fulmar (*Fulmarus glacialis*), double-crested cormorant (*Phalacrocorax auritus*) and the common loon (*Gavia immer*) (Kaiser, 2002). Kaiser (2002) suggested that until 1900, the effect of contact between native people and Europeans may have been of benefit to seabirds; their range expanded as epidemic and cultural disaster overtook the native people. Many parts of the coast became uninhabited by humans, and European foods became commonplace which may have relieved hunting pressure on birds and their eggs. However, in the twentieth century, human activity often had a negative impact on the marine birds of British Columbia (Kaiser 2002). Thus, it is assumed that the biomass of seabirds would be higher in 1900 than in 1750, or any subsequent years. Kaiser (2002) gives the biomass of seabirds that are currently feeding in the study area as 516 tonnes ($0.007 \text{ t}\cdot\text{km}^{-2}$), which is used for the 1750 model. As in Haggan *et al.* (1999), I double the 1750 biomass for the 1900 model ($0.015 \text{ t}\cdot\text{km}^{-2}$). Biomass for the 1950 model was taken as an intermediate value, the average of 1900 and present day to assume a gradual transition.

Wada and Kelson (1996) suggested a P/B of 0.1 yr^{-1} for seabirds and I use this ratio for all periods. They suggested a Q/B for seabirds of 112 yr^{-1} in summer and 98.4 yr^{-1} in winter. I use the average (105.2 yr^{-1}) in all models.

The diet of seabirds in all models was adapted from Beattie (2001). The amount of forage fish consumed was divided into the forage fish and eulachon groups in the northern BC models, based on Vasconcellos and Pitcher (2002c). Consumption of benthos from Wada and Kelson (1996) was divided into epifaunal invertebrates and small crabs. Minor changes were made to

the diet matrix in the 1750 and 1900 models to balance and fit dynamics. Also, fishery discards were removed from the diet.

Transient salmon

The transient salmon group includes sockeye (*Oncorhynchus nerka*), chum (*O. keta*) and pink salmon (*O. gorbuscha*), which migrate through the system on their way to spawning areas. Biomass for 1750 and 1900 ($1.0 \text{ t}\cdot\text{km}^{-2}$ and $0.84 \text{ t}\cdot\text{km}^{-2}$) is taken from Vasconcellos and Pitcher (2002d). They used the ratio between catch and exploitation rate to calculate biomass for 1900 (assuming an exploitation rate of 15%); then they assumed that biomass in 1750 was 20% higher than in 1900. Biomass for 1950 and 2000 ($0.5 \text{ t}\cdot\text{km}^{-2}$ and $0.208 \text{ t}\cdot\text{km}^{-2}$) is based on catch and escapement data (C.J. Walters, UBC Fisheries Centre, *pers. comm.*). Escapement data is from Wood *et al.* (1999) and Rutherford and Wood (2000). It is converted to wet weight using ratios in Groot and Margolis (1991). A negative biomass accumulation rate of $-0.025 \text{ t}\cdot\text{km}^{-2}$ (or -5%) is used in the 1950 model to improve fit to data. In the 2000 model, a negative biomass accumulation rate of $-0.028 \text{ t}\cdot\text{km}^{-2}$ was estimated through simulations of the 1950 model driven by historic mortality and production rates.

Newlands (1998) calculated a P/B value of 2.48 yr^{-1} for transient salmon. That is used for all periods except 1750, which uses a smaller value to represent an older population (0.517 yr^{-1}). It is based on the sum of fishing and natural mortality calculated by Ainsworth *et al.* (2002; Appendix B Table B1) using the empirical formula of Pauly (1980), and supplied with growth data from FishBase (Froese and Pauly, 2005). The Q/B estimates for 1750 and 1900 were calculated by Ainsworth *et al.* (2002; Appendix B Table B2) as 3.72 yr^{-1} using an empirical formula from Palomares and Pauly (1989) modified by Christensen *et al.* (2004a)⁶. Christensen (1996) gave annual Q/B ratios for pink, sockeye and chum of 12.2, 4.6 and 8.2 yr^{-1} respectively; an average value of 8.33 yr^{-1} was used in the 1950 and 2000 models.

Chum and humpback salmon were fished by First Nations people with hook and line, harpoon, spear, traps (weir, stone weir) dip nets, basket traps, or fall traps, and eaten fresh and smoked, or

⁶ The empirical formula for Q/B by Palomares and Pauly (1989) is modified by Christensen *et al.* (2004a).

dried (Irwin, 1984). Hewes (1973) estimated that First Nations caught approximately 6,400 tonnes of salmon in pre-contact times, and I split this catch equally between transient ($0.046 \text{ t}\cdot\text{km}^{-2}$) and resident salmon (i.e., coho and chinook groups). Vasconcellos and Pitcher (2002d) estimated the catch of transient salmon in 1900-1905 to be $0.126 \text{ t}\cdot\text{km}^{-2}$ and this value is used in the 1900 model. Landings for 1950 and 2000 models are based on official commercial statistics (DFO 2004a and b), Pacific region DFO creel surveys for sport catches (DFO 2004c) and estimates of IUU fishing made by Ainsworth and Pitcher (2005c). It was assumed that 30% of the sports catch reported as 'salmon' could be apportioned to the transient salmon group. Catch in numbers is converted into wet weight using ratios from Groot and Margolis (1991). 1950 and 2000 catches represent statistical areas 1-10, including Hecate Strait, Dixon Entrance and Queen Charlotte Sound. The historical record for transient salmon tends to apportion catch into gillnets, seine and troll. The latter was split evenly in the models between salmon troll and salmon freezer troll. Total catch for 1950 is $0.598 \text{ t}\cdot\text{km}^{-2}$ and total catch for 2000 is $0.230 \text{ t}\cdot\text{km}^{-2}$.

Transient salmon feed mostly on zooplankton, but outside of the ecosystem. Migratory species such as these are problematic during dynamic simulations since the abundance of their food is independent of system fluctuations. A large amount of diet import was found to cause serious dynamic instabilities in the group, and so import was limited to 60% for all periods. This could be an underestimate considering that Vasconcellos and Pitcher (2002d) used 100% diet import for the static model; however such values provide nonsense results in Ecosim. The diet of transient salmon remained the same for all four models.

Coho salmon

The 1750 biomass value for coho salmon (*Oncorhynchus kisutch*) is taken from Vasconcellos and Pitcher (2002d) ($0.096 \text{ t}\cdot\text{km}^{-2}$). Those authors estimated biomass of coho salmon in 1900 to be $0.08 \text{ t}\cdot\text{km}^{-2}$ based on the assumption that there has been a 75% decrease in the standing stock over the last century. However, this value may be low considering the more reliable 1950 estimate of $0.1 \text{ t}\cdot\text{km}^{-2}$, especially in light of the fact that exploitation rates had increased considerably in years following World War II. Moreover, time series estimates of biomass beginning in 1925, though uncertain, indicate a higher level of stock abundance throughout the 1920s and 1930s (ranging from $0.15\text{-}0.19 \text{ t}\cdot\text{km}^{-2}$) than could be accounted for under the assumed

levels of fishing mortality. As further evidence of a discrepancy, the observed catches from the 1920s and 1930s result in stock collapse when entered as a forced catch in the 1900 model. The 1900 estimate for coho salmon was therefore increased two-fold over the estimate of Vasconcellos and Pitcher (2002d) to $0.16 \text{ t}\cdot\text{km}^{-2}$. The 1950 biomass value ($0.1 \text{ t}\cdot\text{km}^{-2}$) is based on catch and escapement data (C.J. Walters, UBC Fisheries Centre, *pers. comm.*). Escapement data is from PSC (2002) and Holtby *et al.* (1999), converted to wet weight using ratios in Groot and Margolis (1991). The 2000 biomass value is retained from Beattie (2001) at $0.024 \text{ t}\cdot\text{km}^{-2}$ since it agrees closely with time series. A negative biomass accumulation rate ($-0.01 \text{ t}\cdot\text{km}^{-2}$) is entered into the 1950 model to improve dynamic fit to data.

Beattie (2001) uses monthly estimates of 23% for the increase in body size of coho (obtained from Newlands, 1998). This gives a P/B ratio of 2.76 yr^{-1} for coho which I use in the 1900, 1950 and 2000 models. The 1750 model received a lower value to account for an older population (1.16 yr^{-1}); this value was calculated by Ainsworth *et al.* (2002) as the sum of fishing and natural mortality, using the empirical formula of Pauly (1980) to determine the latter. This production value results in an EE of 0.95; which is appropriate for a resident population. Q/B was estimated for the 1950 and 2000 models as 13.8 yr^{-1} ; this value originates from Beattie (2001). He used Ecopath to estimate the figure assuming a P/Q ratio of 0.2. The Q/B value for 1750 (4 yr^{-1}) was taken from Ainsworth *et al.* (2002) who also assumed a P/Q ratio of 0.2. The value for 1900 (10 yr^{-1}) was chosen to fall between the 1750 and 1950 values.

Catches in 1750 are based on Hewes (1973), who estimated that First Nations caught about 6400 tonnes of salmon in pre-contact times. Of that, one half was assumed to be directed to transient salmon, and the remainder was divided among coho and chinook, producing an estimate of $0.023 \text{ t}\cdot\text{km}^{-2}$ catch for each. Catches in 1900 were calculated by Vasconcellos and Pitcher (2002d) as $0.012 \text{ t}\cdot\text{km}^{-2}$; they assumed that catches were directly proportional to fishing mortality, and that fishing mortality, arrived at through expert consensus, was close to 0.15 yr^{-1} at the turn of the century. Catches for 1950 and 2000 models are based on official commercial statistics (DFO, 2004a; 2004b), Pacific region DFO creel surveys for sport catches (DFO, 2004c) and estimates of IUU fishing made by Ainsworth and Pitcher (2005c). It was assumed that 30% of the sports catch reported as 'salmon' could be apportioned to the coho salmon group. Catch in numbers is

converted into wet weight using ratios from Groot and Margolis (1991). 1950 and 2000 catches represent statistical areas 1-10, including Hecate Strait, Dixon Entrance and Queen Charlotte Sound. Catch estimates for coho are $0.14 \text{ t}\cdot\text{km}^{-2}$ in 1950, and $0.05 \text{ t}\cdot\text{km}^{-2}$ in 2000. Discards in 1750 and 1900 and 1950 are assumed to be negligible; discards in 2000 is taken from Beattie (2001) ($0.001 \text{ t}\cdot\text{km}^{-2}$), and this value originates from Pacific region DFO's observer database 1997.

The diet matrix for all periods is adapted from Beattie (2001), with some changes made, particularly to the 1750 and 1900 models, to achieve mass balance and fit dynamics to data. It is assumed that 1/6th of the predation on forage fish in Beattie (2001) is directed at eulachon (Vasconcellos and Pitcher, 2002c).

Chinook salmon

Chinook (*Oncorhynchus tshawytscha*) biomass in 1750 comes from Vasconcellos and Pitcher (2002d) ($0.144 \text{ t}\cdot\text{km}^{-2}$). Biomass in the 1900 model is assumed to be $0.16 \text{ t}\cdot\text{km}^{-2}$, higher than the estimate of Vasconcellos and Pitcher (2002d) ($0.12 \text{ t}\cdot\text{km}^{-2}$). The value was chosen so that the dynamics would reconcile with the available time series (Eggers *et al.* 2003), which begins in 1925 at $0.19 \text{ t}\cdot\text{km}^{-2}$ when corrected for model area. The 1950 biomass value ($0.09 \text{ t}\cdot\text{km}^{-2}$) is calculated from catches, assuming an exploitation rate of 10%. The 2000 biomass value ($0.036 \text{ t}\cdot\text{km}^{-2}$) is based on catch and escapement data (C.J. Walters, UBC Fisheries Centre, *pers. comm.*). The escapement data is from PSC (2003), and numbers of fish are converted to wet weight using ratios in Groot and Margolis (1991).

For all models except 1750, I use the P/B estimate of Beattie (2001) (2.16 yr^{-1}), which he calculated from growth rates in Newlands (1998). The 1750 model uses a smaller value (0.366 yr^{-1}), as calculated by Ainsworth *et al.* (2002; Appendix B Table B1) using the empirical formula of Pauly (1980) and supplied with growth data from FishBase (Froese and Pauly 2005). The Q/B ratio of chinook used here for 1750 (2.82 yr^{-1}) was calculated by Ainsworth *et al.* (2002) based on the empirical relationship in Palomares and Pauly (1989). The value for 1900 (10 yr^{-1}) is set slightly lower than 1950 to represent an older population, while the 1950 and 2000 models (10.8 yr^{-1}) are based on Beattie (2001).

Catches in 1750 are based on Hewes (1973) ($0.023 \text{ t}\cdot\text{km}^{-2}$); chinook catch is assumed to constitute 25% of pre-contact salmon catch. Catches in 1900 ($0.018 \text{ t}\cdot\text{km}^{-2}$) are taken from Vasconcellos and Pitcher (2002d). Catches for the 1950 and 2000 models are based on official commercial statistics (DFO 2004a and b), Pacific region DFO creel surveys for sport catches (DFO, 2004c) and estimates of IUU fishing made by Ainsworth and Pitcher (2005c). It was assumed that chinook catch in the salmon recreational fishery accounted for 40% of total number of fish. Numbers of fish are converted to wet weight using ratios from Groot and Margolis (1991). Discards in the 2000 model ($0.01 \text{ kg}\cdot\text{km}^{-2}$) are based on Beattie (2001), who used data from the DFO observer program.

For all models, the diet of chinook was adapted from Beattie (2001), eulachon having been extracted from the forage fish compartment. It was assumed that 1/6th of the predation on forage fish in Beattie (2001) was directed at eulachon (Vasconcellos and Pitcher, 2002c).

Small and large squid

Squid are split into two functional groups, small and large, to alleviate cannibalism in the models but the groups are not ontogenically linked in Ecosim. The small squid group includes opal squid (*Loligo opalescens*); the large squid group includes flying squid (*Ommastrephes bartramii*), red squid (*Berryteuthis magister*), Nail squid (*Onychoteuthis borealijaponica*) and eight-armed squid (*Gonatopsis borealis*). In experimentation, rigorous manipulation of diet, feeding and vulnerability parameters was not able to reproduce the downward trend for squid and small squid from 1950 to 2000 reported by Ainsworth *et al.* (2002). Moreover, LEK Information obtained from community interviews did not support a downward trend. The biomass estimate in the 1950 model made by Ainsworth *et al.* (2002) was abandoned and replaced with the 2000 quantity to produce a stable trend; this 1950 biomass value maintained the EEs of small and large squid between 90-95%, which is appropriate for forage species. A negative biomass accumulation is used for both groups in 1950 to improve dynamics (small: $-0.109 \text{ t}\cdot\text{km}^{-2}$; large: $-0.038 \text{ t}\cdot\text{km}^{-2}$). Biomass accumulation in 2000 is based on simulations driving the 1950 with historic mortality and production rates (small: $-0.006 \text{ t}\cdot\text{km}^{-2}$; large: $0.027 \text{ t}\cdot\text{km}^{-2}$).

I use Beattie's (2001) estimate of P/B (6.023 yr^{-1}) for the flying squid (*Onychoteuthis borealijaponica*) for both squid groups. The same author calculates a Q/B ratio of 34.68 yr^{-1} for two other Loligo species (*L. peale* and *L. vulgaris*) and I use this ratio for both groups in all models.

No squid were caught in the 1750s, 1900s or 1950s. In 2000, opal squid (*Loligo opalescens*) are fished primarily as bait for sablefish, crabs and halibut using seine nets (DFO, 1999a) while a new fishery for the neon flying squid *Ommastrephes bartrami* is currently being promoted (DFO, 1999b). Based on DFO observer data, the groundfish trawl fishery is known to discard a small amount of squid ($0.002 \text{ kg}\cdot\text{km}^{-2}$) Beattie (2001).

The diet of squid is adapted from Beattie (2001) assuming that 1/6th of the predation on forage fish was directed at eulachon (Vasconcellos and Pitcher 2002c). The diet of adult squid remained the same for all four models, although significant changes were made in this group for balancing and fitting.

Ratfish

Ratfish (*Hydrolagus colliei*) biomass for 1750 and 1900 (0.262 and $0.183 \text{ t}\cdot\text{km}^{-2}$ respectively) is based on Ainsworth *et al.* (2002), who used Ecopath to estimate the figures. Values for 1950 and 2000 were taken from Beattie (2001) ($0.517 \text{ t}\cdot\text{km}^{-2}$) who based his estimate on central coast survey data in Fargo *et al.* (1990) using average values from 1984, 1986 and 1987. I therefore assume that ratfish biomass has remained quite stable over the last 50 years.

For the 1750 and 1900 models, P/B is based on natural mortality calculated in Ainsworth *et al.* (2002; Appendix Table B1) using the empirical formula of Pauly (1980) (0.199 yr^{-1}). Beattie (2001) suggests that the P/B of ratfish should be similar to dogfish and his figure is used here (0.099 yr^{-1}) for the 1950 and 2000 models. Beattie (2001) calculates a Q/B ratio for ratfish of 1.4 yr^{-1} using an empirical formula (Palomares and Pauly, 1989), and this figure is used in all models

Landings of ratfish are assumed negligible in all periods except the present day. It is now caught by the groundfish fishery in small amounts (Beattie, 2001); $0.052 \text{ kg}\cdot\text{km}^{-2}$ was recorded by the

DFO observer program database for 1997. Beattie (2001) obtained values on discards of ratfish by the groundfish trawl fisheries of about $0.01 \text{ t}\cdot\text{km}^{-2}$, and this value is used in the 1950 and 2000 models. Hay *et al.* (1999) suggest that ratfish are caught as bycatch to the shrimp trawl fishery. I use the estimate obtained by Hay *et al.* (1999) for ratfish bycatch ($1.25 \text{ kg}\cdot\text{km}^{-2}$) as a discard from the ratfish compartment in the 2000 and 1950 models.

Diet composition for ratfish is taken from Beattie (2001) for all periods. Benthic invertebrates consumed by ratfish were divided equally between carnivorous and detritivorous groups in the northern BC models. It is assumed that 1/6th of the predation on forage fish is directed towards eulachon (after Vasconcellos and Pitcher, 2002c).

Dogfish

The 1750 biomass of dogfish (*Squalus acanthias*) was assumed to be 50% higher than the biomass in the present day based on Vasconcellos and Pitcher (2002e) ($1.36 \text{ t}\cdot\text{km}^{-2}$). I assume that the 1900 biomass ($0.6 \text{ t}\cdot\text{km}^{-2}$) is 40% greater than in 1950 due to a revived fishery for liver oil following World War II (Anon, 1958). The 1950 figure ($0.427 \text{ t}\cdot\text{km}^{-2}$) is estimated by Ecopath so that EE equals 0.94; the biomass value agrees with the qualitative LEK information that suggests a clear increase in dogfish biomass over the last 50 years (Ainsworth and Pitcher, 2005a). Dogfish biomass from Beattie (2001) ($0.909 \text{ t}\cdot\text{km}^{-2}$) is used for the 2000 model. A small biomass accumulation rate ($5\% \text{ yr}^{-1}$) is set in the 1900 model, which works together with trophic effects to achieve suitable agreement with the 1950 biomass estimate upon simulation. The 1950 biomass accumulation rate ($4 \text{ kg}\cdot\text{km}^{-2}$) is in place to improve dogfish fit to data, and the 2000 biomass accumulation rate ($-0.06 \text{ t}\cdot\text{km}^{-2}$) is estimated through simulations of the 1950 model, driven by historic mortality and production.

For the 1750 model, P/B is represented by the natural mortality rate (0.11 yr^{-1}) calculated in Ainsworth *et al.* (2002; Appendix B Table B1) using an empirical relationship (Pauly, 1980). Their estimate for 1900, calculated in the same way, is revised upwards here to 0.2 yr^{-1} in fitting the 1900 model to data in order to avoid stock collapse in simulations that use the historical catch rate. The estimate for 1950 made by Ainsworth *et al.* (2002) is similarly revised upwards to 0.15 yr^{-1} ; this provides enough net production in the baseline model to accommodate the considerable

catch rate seen in the late 1950s and early 1960s and permit biomass growth in the 1960s and 1970s as suggested by LEK data (Ainsworth and Pitcher, 2005a). For 2000, I apply the P/B estimate of Beattie (2001) (0.099 yr^{-1}); he used the sum of natural mortality obtained from Wood *et al.* (1979) and fishing mortality of 0.005 yr^{-1} obtained from the DFO Fishery Observer Database. The production rate in 2000 should be lower than 1950 considering the assumed increase in stock size. For the 1900 and 1750 models, the Q/B calculated by Ainsworth *et al.* (2002; Appendix B Table B2) using an empirical relationship was used (3.33 yr^{-1}) (Palomares and Pauly, 1989). The Q/B ratio in the 2000 and 1950 models is based on Beattie (2001) (2.72 yr^{-1}).

Catch of dogfish is assumed negligible in the 1750 model, since the dogfish fishery (for liver oil) started around 1872 (Vasconcellos and Pitcher, 2002e). Longline catch in 1900 ($0.039 \text{ t}\cdot\text{km}^{-2}$) and 1950 ($0.032 \text{ t}\cdot\text{km}^{-2}$) is based on Ketchen (1986). Beattie (2001) obtained values on discards of dogfish by the groundfish trawl fisheries from DFO's observer program database for 1997, and I applied this value in the 1950 and 2000 models. Based on personal observation, I include a discard rate for the salmon gillnet fleet roughly equivalent to the rates from the groundfish trawl fleet, and I apply that to 1950 and 2000 models producing a total annual discard estimate of $0.019 \text{ t}\cdot\text{km}^{-2}$ and $0.018 \text{ t}\cdot\text{km}^{-2}$, respectively. The discard estimate for the present day model also includes a small amount from the shrimp trawl fishery (Hay *et al.*, 1999).

The diet obtained from Beattie (2001) was adapted for the 2000 and 1950 models by assuming that 1/6th of the predation on forage fish was directed to eulachon (Vasconcellos and Pitcher, 2002c) and the proportion of the diet attributed to benthic invertebrates was divided into infaunal carnivorous invertebrates and infaunal invertebrate detritivores. Transient salmon was also included in the diet of dogfish for these models, and the percentage of coho and chinook was reduced to balance those compartments. Dogfish are a controlling group in dynamic simulations, and so their diet was modified extensively to achieve mass-balance and improve fit to time series data (1900 and 1950 models).

Juvenile and adult pollock

Walleye pollock (*Theragra chalcogramma*) is split into adult and juveniles to reduce cannibalism in the model. Biomass values for juvenile and adult pollock are retained from Ainsworth *et al.* (2002) for the 1750 (juveniles 1.318 t·km⁻²; adults: 0.622 t·km⁻²) and 1900 models (juveniles: 0.926 t·km⁻²; adults: 0.479 t·km⁻²). Ecopath calculated the values. Beattie (2001) used the figure 11-22 thousand tonnes of pollock in Hecate Strait from Saunders and Andrews (1996), under the assumption that 37% of the pollock stock are juveniles (Niggol, 1982), to calculate both juvenile and adult walleye pollock biomasses (0.132 t·km⁻² and 0.359 t·km⁻²). The 1950 and 2000 models use these values.

Natural mortalities calculated by Ainsworth *et al.* (2002; Appendix B Table B1), based on the empirical relationship of Pauly (1980), are used as P/B estimates for the 1750 and 1900 models (juveniles: 0.23 yr⁻¹; adults: 0.15 yr⁻¹). The P/B estimates of 1.061 yr⁻¹ (for juveniles) and 0.263 yr⁻¹ (for adults) are obtained from Beattie (2001), and used in the 1950 and 2000 models. The 1750 and 1900 Q/B estimates for juvenile pollock remain unchanged from Ainsworth *et al.* (2002; Appendix B Table B2) at 5.05 yr⁻¹ for juveniles and 3.36 yr⁻¹ for adults; both values come from the empirical formula of Palomares and Pauly (1989). The rates for 1950 and 2000 are based on Beattie (2001) (5.31 and 1.17 yr⁻¹ for juveniles and adults, respectively), who assumed a P/Q ratio of 0.2 for juveniles, and used laboratory results to parameterize the adult group (Livingston *et al.*, 1986).

Catches of pollock are assumed negligible in the 1750 and 1900 models. Groundfish trawl catch in the 1950 and 2000 models (0.007 t·km⁻²) is taken from Beattie (2001), who based his information on the DFO's observer program database for 1997. Likewise, I use his discard rate from groundfish trawl (0.002 t·km⁻²) and include a small bycatch for the shrimp trawl fishery in the 2000 model (based on Hay *et al.*, 1999).

Diet estimates are obtained from Beattie (2001). Decapods, euphausiids and mysids are assumed to be euphausiids, while larvaceans, amphipods and gastropods are assumed to be epifaunal invertebrates. Fish are considered to be forage fish and are split into 1/6th eulachon and 5/6th

forage fish groups based on Vasconcellos and Pitcher (2002c). The 1750 and 1900 models required moderate alteration to balance this group.

Forage fish and eulachon

Forage fish consist mainly of sandlance (*Ammodytes hexapterus*), although pilchard (*Sardinops sagax*), anchovy (*Engraulis mordax*), capelin (*Mallotus villosus*), chub mackerel (*Scomber japonicus*), shad (*Alosa sapidissima*) and smelts (Osmeridae) are also present (Beattie 2001). Eulachon (*Thaleichthys pacificus*) receives its own functional group due to its cultural importance among First Nations. All diet references to forage fish were split into 1/6th eulachon and 5/6th forage fish as per Vasconcellos and Pitcher (2002c). The biomass of forage fish and eulachon in all four time periods was estimated by Ecopath in Ainsworth *et al.* (2002), and those values are used here.

The P/B ratios for forage fish in 1750 and 1900 (0.595 and 0.588 yr^{-1}) are based on calculations in Ainsworth *et al.* (2002; Appendix B Table B1). Those authors used the empirical relationship of Pauly (1980) to estimate natural mortality. The P/B ratio for eulachon is based on these values, but is increased to 0.6 yr^{-1} in the 1750 model and 0.7 yr^{-1} in the 1900 model to account for additional mortality caused by traditional native fisheries. Ecopath calculated production rate of forage fish in 1950 as 1.17 yr^{-1} by assuming an EE of 0.95,. Beattie (2001) used the average of adult and juvenile herring P/B ratios for forage fish (1.43 yr^{-1}), and that value is applied here to forage fish and eulachon groups in the 2000 model, and to eulachon in the 1950 model. The Q/B rate of 6.61 yr^{-1} calculated by Ainsworth *et al.* (2002; Appendix B Table B2) using the empirical formula of Palomares and Pauly (1989) for both forage fish and eulachon is employed in the 1750 and 1900 models. Beattie (2001) uses the average of adult and juvenile herring Q/B ratios for forage fish (8.4 yr^{-1}), and I use that value for both forage fish and eulachon groups in the 1950 and 2000 models.

There was a small recreational fishery for capelin in the past in the Georgia Strait area and this is probably also true for northern BC (Vasconcellos and Pitcher, 2002f). A seine net reduction fishery for sardine began in 1917, and catch increased to 80,558 tonnes by 1943. In 1947 that amount was down to 444 tonnes (Schweigert, 1987). The 1943 value was used for the 1950

forage fish group ($6 \text{ kg} \cdot \text{km}^{-2}$). However, at present no forage fish are caught except for eulachon, for which the total catch in BC is approximately 366 tonnes, and I assume that 3/5ths of that is taken from northern British Columbia ($3 \text{ kg} \cdot \text{km}^{-2}$). First Nations people harvest eulachon with rakes, seine, dip and bag nets after which they are dried or smoked and oil is extracted (Irwin 1984). Vasconcellos and Pitcher (2002c) estimate a catch of 3,000 tonnes per year for the early 1900s by assuming that catches were one order of magnitude higher than at the present time. Those authors also suggested that pre-contact (1750) catch was probably similar ($0.043 \text{ t} \cdot \text{km}^{-2}$), and their estimate is used here. Beattie (2001) obtained values on discards of forage fish by the groundfish trawl fisheries from DFO's observer program database for 1997, and I split the discards into forage fish ($0.04 \text{ kg} \cdot \text{km}^{-2}$) and eulachon ($0.007 \text{ kg} \cdot \text{km}^{-2}$) assuming that 1/6th of the discards can be attributed to eulachon groups after Vasconcellos and Pitcher (2002c). Eulachon is also discarded by the shrimp trawl fishery, and Hay *et al.* (1999) calculated that shrimp trawlers on the central coast discard approximately 90 tonnes ($1 \text{ kg} \cdot \text{km}^{-2}$) annually.

The diet of forage fish was obtained from Beattie (2001), and was used in all four models. This value was adapted for eulachon, reducing the proportion of euphausiids in their diet in order to balance the model. I also assume that they do not feed on detritus and that copepods are more important in their diet.

Juvenile and adult herring

Herring (*Clupea pallasii*) was split into adult and juvenile compartments to reduce the effects of cannibalism in the model. Biomass of juvenile and adult herring in 1750 (5.45 and $7.50 \text{ t} \cdot \text{km}^{-2}$) was taken from Ainsworth *et al.* (2002), who used Ecopath to estimate the figures, as were the 1900 biomass estimates (3.73 and $2.48 \text{ t} \cdot \text{km}^{-2}$). The 1950 biomass estimate for the juvenile group ($1.32 \text{ t} \cdot \text{km}^{-2}$) was estimated assuming a similar proportion of juveniles to adults as in the present day. The 1950 estimate of adult herring biomass ($1 \text{ t} \cdot \text{km}^{-2}$) is based on Schweigert (2004). The 2000 biomass of juvenile herring was obtained from Beattie (2001) at $2.265 \text{ t} \cdot \text{km}^{-2}$, and the adult biomass ($0.658 \text{ t} \cdot \text{km}^{-2}$) comes from Schweigert (2004). A biomass accumulation rate of -5% is applied to the adult group in 1950 to represent the damaging reduction fishery that continued until the late 1960s. In 2000, a small negative biomass accumulation (-0.5%) was

estimated for the adult group through simulations with the 1950 model under historical mortality rates.

For 1750, natural mortality is used to represent P/B for juvenile and adult groups, as calculated in Ainsworth *et al.* (2002; Appendix B Table B1) based on the empirical relationship of Pauly (1980) (juveniles: 1.173 yr^{-1} ; adults: 0.792 yr^{-1}). The P/B ratio for 1900 had to be increased over this estimate in both juvenile and adult groups to satisfy the feeding demands of higher predators and achieve mass-balance (juveniles: 2 yr^{-1} ; adults: 0.9 yr^{-1}). Juvenile production rates for the 1950 and 2000 models were taken from Beattie (2001), who based his estimate for the present day on calculations done by Robinson and Ware (1994). Adult production rates are taken for the 1950 model as 1.54 yr^{-1} from Prince William Sound (Okey and Pauly, 1999) because the (present-day) value used by Beattie (2001) (0.683 yr^{-1}) was found to be irreconcilable with the reported catch series and resulted in stock collapse under historic fishing pressure. The adult production rate for the 2000 model (0.683 yr^{-1}) is from Beattie (2001). It results in an EE of 0.95, which is appropriate for herring. The Q/B ratios for 1750 and 1900 were calculated in Ainsworth *et al.* (2002; Appendix B Table B2) based on the empirical relationship of Palomares and Pauly (1989) as 11.3 yr^{-1} for the juvenile group and 7.5 yr^{-1} for the adult group. Consumption rates for 1950 and 2000 are taken from Beattie (2001), who cited Robinson and Ware (1994) (juveniles: 5.84 yr^{-1} ; adults: 3.21 yr^{-1}).

BC First Nations caught a quarter million pounds of herring annually in pre-contact times (Carrothers 1941), which equals $2 \text{ kg}\cdot\text{km}^{-2}$. By the turn of the century, there was less aboriginal fishing and the commercial herring fishery in Prince Rupert (DFO, 2001a) and on the Central Coast (DFO, 2001b) was only getting started; total catch in northern BC was $0.65 \text{ kg}\cdot\text{km}^{-2}$ at that time (Daniel *et al.*, 2001). Catch increased during the 1930s with the start of the dry salt fishery (DFO, 2001c; Jones, 2000). By 1950 annual landings topped $0.922 \text{ t}\cdot\text{km}^{-2}$ and remained at high levels throughout the reduction fishery until it ended in 1967. A more conservative fishery for roe now exists, that captures $0.241 \text{ t}\cdot\text{km}^{-2}$ (Daniel *et al.*, 2001). Beattie (2001) reports that herring is now caught and discarded by the groundfish trawl fishery at a rate of $0.003 \text{ t}\cdot\text{km}^{-2}$, based on DFO observer data.

It is assumed that juvenile herring eat 90% copepods and 10% euphausiids, and that adult herring eat 10% copepods and 90% euphausiids, as in Beattie (2001).

Juvenile and adult Pacific ocean perch

For 1750, 1900 and 1950, biomass for juvenile and adult Pacific ocean perch (POP) (*Sebastes alutus*) are taken from Ainsworth *et al.* (2002) who used Ecopath to estimate values for 1750 (juveniles: $0.213 \text{ t}\cdot\text{km}^{-2}$; adults: $1.404 \text{ t}\cdot\text{km}^{-2}$), for 1900 (juveniles: $0.153 \text{ t}\cdot\text{km}^{-2}$; adults: $1.011 \text{ t}\cdot\text{km}^{-2}$) and 1950 (juveniles: $0.036 \text{ t}\cdot\text{km}^{-2}$; adults: $1.019 \text{ t}\cdot\text{km}^{-2}$). These values yield reasonable EEs for an unexploited and long-lived group in 1750 and 1900 (adult EE $\sim 0.4\text{-}0.5$); EE is also acceptable for 1950, by which time exploitation had increased (adult EE ~ 0.8). The 2000 biomass (juveniles: $.062 \text{ t}\cdot\text{km}^{-2}$; adults: $0.561 \text{ t}\cdot\text{km}^{-2}$) is based on a catch-age model of Goose Island gully fishery data (Schnute *et al.*, 2001). The downward trend throughout periods agrees with scientific consensus (Beattie, 1999). Biomass for the juvenile group represents invulnerable biomass reported in Schnute *et al.* (2001), and the adult group represents vulnerable biomass.

For 1750 and 1900, natural mortality is used to represent P/B, as calculated by Ainsworth *et al.* (2002; Appendix B Table B1) based on the empirical formula of Pauly (1980). For adults, it is 0.227 yr^{-1} and for juveniles it is 0.338 yr^{-1} . I assume that a lower P/B is justified for juvenile groups in the ancient models as compared with the present day, since many are now killed as bycatch in fishing activities. The 1950 and 2000 P/B ratios for juveniles (0.672 yr^{-1}) and adults (0.144 yr^{-1}) are based on Beattie (2001), except the juvenile P/B for 1950 has been adjusted (0.5 yr^{-1}) to facilitate dynamic fitting; the directional change is logical considering that populations are now depleted over 1950 levels. Q/B estimates for 1750 and 1900 are based on the empirical formula of Palomares and Pauly (1989); the calculations are presented in Ainsworth *et al.* (2002; Appendix B Table B2) (juveniles: 6.12 yr^{-1} ; adults: 4.08 yr^{-1}). As recommended in Ainsworth *et al.* (2002), P/Q ratios for the pre-contact and 1900 models are maintained for both juvenile and adult groups at around 0.05, which is appropriate for a long-lived and unexploited species. Q/B rates for 1950 and 2000 models are borrowed from Beattie (2001), who also used the empirical formula.

There was no fishery for Pacific ocean perch during 1750 or 1900, but 1950 catches ($2.6 \text{ kg} \cdot \text{km}^{-2}$) are based on red and rock cod landings from historical DFO catch statistics from 1951 (DFO 1995). Landings in 2000 ($0.06 \text{ t} \cdot \text{km}^{-2}$) are based on the catch-age model of Schnute *et al.* (2001), but also include data from the sport fishery (DFO, 2004c) and IUU catch (Ainsworth and Pitcher, 2005c). Pacific Ocean perch was caught and discarded in the 1950 and 2000 model by the groundfish trawl fishery ($0.002 \text{ t} \cdot \text{km}^{-2}$ – obtained from Beattie (2001) from observer data).

Diet is taken from Beattie (2001), and includes primarily euphausiids (90%); the remainder is divided among copepods and squid groups.

Inshore rockfish

Inshore rockfish include copper (*Sebastes caurinus*), quillback (*S. maliger*), tiger (*S. nigrocinctus*), China (*S. nebulosus*) and yelloweye rockfish (*S. ruberrimus*). Biomass in 1750 and 1900 (0.096 and $0.081 \text{ t} \cdot \text{km}^{-2}$) is based on Ainsworth *et al.* (2002), who used Ecopath to estimate the figures. Biomass in 2000 is based on Beattie (2001) ($0.1 \text{ t} \cdot \text{km}^{-2}$), which produces a reasonable EE of 0.96. The 1950 value is also based on Beattie (2001), but was modified to $0.095 \text{ t} \cdot \text{km}^{-2}$ to increase EE to 0.85. A small negative biomass accumulation is set in the 1950 model ($-0.002 \text{ t} \cdot \text{km}^{-2}$) to improve dynamic fit to data, and a small positive biomass accumulation is estimated for the 2000 model ($0.001 \text{ t} \cdot \text{km}^{-2}$), based on the end-state of a 50-year simulation with the 1950 model (using historical mortality and production drivers).

The P/B ratio used in 1750 and 1900 is assumed to be represented by natural mortality, and is based on calculations made by Ainsworth *et al.* (2002; Appendix B Table B1) using the empirical formula of Pauly (1980). In 1950 and 2000, the P/B ratio (0.19 yr^{-1}) is based on the value suggested by Beattie (2001), who used data from Oregon in Rogers *et al.* (1996). The Q/B ratio in 1750 (3.7 yr^{-1}) is based on the empirical formula of Palomares and Pauly (1989) as calculated in Ainsworth *et al.* (2002); it is an average value for copper, yelloweye and quillback. The Q/B value for 1900 (5.54 yr^{-1}) is reduced slightly from Beattie's (2001) present-day estimate to represent an older population, while the 1950 and 2000 values are taken directly (5.69 yr^{-1}) (represents copper and quillback populations described by Murie *et al.* (1994).

Catches of inshore rockfish are assumed to be negligible in the 1750 and 1900 models. Catch for 1950 ($3.7 \text{ kg} \cdot \text{km}^{-2}$) was assumed to represent one half of the amount landed by the commercial fleet in 2000 (from Kronlund and Yamanaka, 1999), while landings in the 2000 model ($10.3 \text{ kg} \cdot \text{km}^{-2}$) also includes IUU catch from Ainsworth and Pitcher (2005c), and sports catch reported in Forrest (2002) based on a personal communication. The total recreational catch for rockfish in Forrest (2002) was divided evenly among inshore rockfish and planktivorous rockfish.

Diet composition is based on Beattie (2001), who cited Murie *et al.* (1994). Beattie's (2001) forage fish diet item is divided into forage fish and eulachon groups in the northern BC models, and other changes were made for balancing, especially in the 1750 and 1900 models.

Juvenile and adult piscivorous rockfish

Piscivorous rockfish include species that feed mainly on fish and large invertebrates: rougheye (*Sebastes aleutianus*), shortraker (*S. borealis*), black (*S. melanops*), blue (*S. mystinus*), chillipepper (*S. goodei*) and dusky rockfish (*S. ciliatus*); also shortspine thornyhead (*Sebastolobus alascanus*) and longspine thornyheads (*S. altivelis*). Biomasses in 1750 (juveniles: $0.02 \text{ t} \cdot \text{km}^{-2}$; adults: $0.137 \text{ t} \cdot \text{km}^{-2}$) and in 1900 (juveniles: $0.016 \text{ t} \cdot \text{km}^{-2}$; adults: $0.119 \text{ t} \cdot \text{km}^{-2}$) were estimated by Ecopath in Ainsworth *et al.* (2002). Biomass in the 2000 model (juveniles: $0.007 \text{ t} \cdot \text{km}^{-2}$; adults: $0.654 \text{ t} \cdot \text{km}^{-2}$) was estimated by Ecopath in Ainsworth *et al.* (2004), as was adult biomass in the 1950 model, but juvenile biomass in the 1950 model was calculated assuming the same ratio of juveniles to adults as seen in 2000 (juveniles: $0.008 \text{ t} \cdot \text{km}^{-2}$; adults: $0.541 \text{ t} \cdot \text{km}^{-2}$). A small biomass accumulation was accepted in the 1950 to improve dynamic fit to data ($0.004 \text{ t} \cdot \text{km}^{-2}$), and a small negative biomass accumulation was used in 2000 ($-0.006 \text{ t} \cdot \text{km}^{-2}$), as estimated through simulations of the 1950 model under historic mortality and production drivers.

The P/B ratio for all time periods is taken from Beattie (2001) (juveniles: 0.261 yr^{-1} ; adults: 0.037 yr^{-1}), who based his estimate on total mortality (Z) for rougheye and shortraker from Leaman (1986). The Q/B value used for the adult group in all models (1.26 yr^{-1}) was calculated by Beattie (2001) using the empirical relationship of Palomares and Pauly (1989). He assumed

that the consumption rate of juveniles was 50% higher than adults (1.89 yr^{-1}), and I use this estimate in all models.

I assume that no fishery existed for piscivorous rockfish in 1750 or 1900. Landings in 1950 are based on Ainsworth *et al.* (2002) ($0.011 \text{ t}\cdot\text{km}^{-2}$). Those authors assumed that commercial landings were 50% of the present day amount, and that recreational catch was 9% (based on Forrest, 2002). Landings in 2000 ($0.027 \text{ t}\cdot\text{km}^{-2}$) are based on DFO observer data reported in Beattie (2001), plus rougheye catch from Kronlund *et al.* (1999) and recreational catch from an unpublished DFO survey reported in Forrest (2002).

Diet is based on Beattie (2001). The prey item squid was divided in the northern BC models into small and large squid, and forage fish were divided into forage fish and eulachon. Further changes were made, particularly to the 2000 model, to balance.

Juvenile and adult planktivorous rockfish

Planktivorous rockfish include yellowmouth (*Sebastes reedi*), redstripe (*S. proriger*), widow (*S. entomelas*), yellowtail (*S. flavidus*), darkblotch (*S. crameri*), canary (*S. pinniger*), splitnose (*S. diploproa*), sharpchin (*S. zacentrus*), Puget sound (*S. emphaeus*), bocaccio (*S. paucispinis*) and shortbelly rockfish (*S. jordani*). Biomass in the 1750 model (juveniles: $0.207 \text{ t}\cdot\text{km}^{-2}$; adults: $2.086 \text{ t}\cdot\text{km}^{-2}$) was estimated by Ecopath in Ainsworth *et al.* (2002), as was biomass in the 1900 model (juveniles: $0.134 \text{ t}\cdot\text{km}^{-2}$; adults: $1.286 \text{ t}\cdot\text{km}^{-2}$). Juvenile biomass in 1950 ($0.189 \text{ t}\cdot\text{km}^{-2}$) is based on Ainsworth *et al.* (2002), who assumed that there were a similar proportion of juveniles to adults as in the present day. Adult biomass in 1950 ($1.213 \text{ t}\cdot\text{km}^{-2}$) is estimated by Ecopath assuming an EE of 0.9. Biomass values in 2000 (juveniles: $0.136 \text{ t}\cdot\text{km}^{-2}$; adults: $1.207 \text{ t}\cdot\text{km}^{-2}$) are based on Beattie (2001) who found yellowmouth and redstripe data in Schnute *et al.* (1999). There is a positive biomass accumulation in 1950 ($0.006 \text{ t}\cdot\text{km}^{-2}$) and a negative biomass accumulation in 2000 ($-0.027 \text{ t}\cdot\text{km}^{-2}$), which was reduced slightly for balancing from the estimates of Walters and Bonfil (1999) for yellowmouth, yellowtail and canary rockfish (1994-1996 data).

The P/B values for all time periods are from Beattie (2001) (juveniles: 0.261 yr^{-1} ; adults: 0.068 yr^{-1}). He based his estimate on total mortality data derived from otoliths (Leaman, 1986). The estimate for Q/B is based on the empirical formula of Palomares and Pauly (1998) and is calculated in Beattie (2001) (juveniles: 3.21 yr^{-1} ; adults: 2.14 yr^{-1}).

I assume there is no catch for planktivorous rockfish in 1750 or 1900. In 1950, landings are assumed to equal one half of the present day commercial landings, or $0.036 \text{ t}\cdot\text{km}^{-2}$. The 2000 figure ($0.077 \text{ t}\cdot\text{km}^{-2}$) includes groundfish trawl data from the DFO observer database reported in Beattie (2001), and recreational data from Forrest (2002). Total recreational catch for rockfish in Forrest (2002) was divided evenly among inshore rockfish and planktivorous rockfish. There are also discards in the 1950 and 2000 models ($0.06 \text{ t}\cdot\text{km}^{-2}$), which are caught by groundfish trawl (Beattie, 2001) and salmon gillnet (C. Ainsworth, *pers. obs.*).

Diet is adapted from Beattie (2001), and he based his estimate on splitnose, darkblotch, yellowtail and canary rockfishes (Brodeur and Pearcy, 1992). Forage fish prey items were divided in the northern BC models into forage fish and eulachon. Minor changes were made to the 1900 model for balancing.

Juvenile and adult turbot

Turbot, also called arrowtooth flounder (*Atheresthes stomias*), is abundant in northern BC. Biomass in 1750 was calculated by Ecopath in Ainsworth *et al.* (2002) (juveniles: $0.248 \text{ t}\cdot\text{km}^{-2}$; adults: $2.196 \text{ t}\cdot\text{km}^{-2}$). Vasconcellos and Fargo (2002) warn that the unfished equilibrium biomass of turbot is only 56,000 tonnes, or $0.8 \text{ t}\cdot\text{km}^{-2}$ – far lower than the 1750 estimate. However, equilibrium analysis using the 2000 model suggests a B_0 of at least $2.4 \text{ t}\cdot\text{km}^{-2}$, so I accept the larger pre-contact estimate. Adult biomass in 1900 is assumed to be the same as in the present day ($1.53 \text{ t}\cdot\text{km}^{-2}$), and juvenile biomass ($0.17 \text{ t}\cdot\text{km}^{-2}$) is held over from Ainsworth *et al.* (2002), who used Ecopath to estimate. Turbot biomass is assumed to have remained constant over the last fifty years based on qualitative accounts from LEK interviews (Ainsworth and Pitcher, 2005a). Biomass in 1950 and 2000 is $0.218 \text{ t}\cdot\text{km}^{-2}$ for juveniles and $1.53 \text{ t}\cdot\text{km}^{-2}$ for adults based on Beattie (2001). A small negative biomass accumulation for turbot in 2000 ($-5 \text{ kg}\cdot\text{km}^{-2}$) is used to increase EE to a preferred level (0.8).

The P/B ratio for all time periods is based on Beattie (2001) (juveniles: 0.33 yr^{-1} ; adults 0.22 yr^{-1}), except for adults in 1900 and juveniles in 1950. Beattie's (2001) adult value was from Prince William Sound (Willette, 1999), and his juvenile value assumes 150% of the adult production rate. The 1900 adult value (0.3 yr^{-1}) was increased over this estimate to facilitate historic catch rates in simulations, and the 1950 juvenile value (0.56 yr^{-1}) was estimated by Ecopath assuming an EE of 0.95 to balance the model. The Q/B ratio for all periods is based on Beattie (2001) (juveniles: 2.17 yr^{-1} ; adults 1.98 yr^{-1}) which he takes from the eastern Bering Sea (Livingston *et al.*, 1986).

I assume that there was no catch for turbot in 1750 or 1900. DFO commercial catch statistics from 1951 (DFO, 2004e) are used in the 1950 model ($3.8 \text{ kg} \cdot \text{km}^{-2}$); all fishing was done by groundfish trawlers, primarily for mink feed (Beattie, 1999). Catch in 2000 are from commercial statistics (DFO, 2004b) and IUU estimates from Ainsworth and Pitcher (2005c). In the 2000 model, turbot bycatch is $0.028 \text{ t} \cdot \text{km}^{-2}$. The figure is based on DFO observer data for groundfish trawl (Beattie, 2001), also bycatch in the shrimp trawl fishery (Hay *et al.*, 1999). A small amount of bycatch is assumed in the halibut hook and line fishery.

Turbot diet is based on Beattie (2001), who used stomach content analysis from the Gulf of Alaska (Yang, 1995). Minor modifications were made in all models for balancing.

Juvenile and adult flatfish

The flatfish group includes flounders and soles (Pleuronectidae, except arrowtooth), and sanddabs (*Citharichthys* spp.). Few species are pursued in northern BC, mainly rock sole (*Pleuronectes bilineatus*), English sole (*Pleuronectes vetulus*), Petrale sole (*Eopsetta jordani*), dover sole (*Microstomas pacificus*) and butter sole (*Isopsetta isolepis*). Biomass of flatfish for 1750 (juveniles: $2.583 \text{ t} \cdot \text{km}^{-2}$; adults: $1.765 \text{ t} \cdot \text{km}^{-2}$) and 1900 (juveniles: $1.606 \text{ t} \cdot \text{km}^{-2}$; adults: $1.014 \text{ t} \cdot \text{km}^{-2}$) was estimated by Ecopath in Ainsworth *et al.* (2002). Adult biomass in 1950 is based on catch-age analysis by Fargo (1999), and includes information on English sole, dover sole and rock sole. Juvenile biomass in 1950 is assumed to be the same as the present day, and juvenile biomass in the 2000 model is taken from Beattie (2001) ($0.259 \text{ t} \cdot \text{km}^{-2}$). Biomass for the

adult pool in 2000 is taken from Fargo (1999) ($0.236 \text{ t}\cdot\text{km}^{-2}$). There is a small negative biomass accumulation for the adult group in the 1950 model to improve fit to data ($-0.08 \text{ t}\cdot\text{km}^{-2}$), and there is a small positive biomass accumulation in the 2000 model as estimated in simulations of the 1950 model (juveniles: $0.007 \text{ t}\cdot\text{km}^{-2}$; adults: $0.006 \text{ t}\cdot\text{km}^{-2}$).

The P/B values for flatfish used for 1750 and 1900 (juveniles: 0.382 yr^{-1} ; adults: 0.257 yr^{-1}) are based on natural mortality rates calculated by Ainsworth *et al.* (2002) using the empirical formula of Pauly (1980). The 1950 P/B ratios (juveniles: 1.0 yr^{-1} ; adults: 0.45 yr^{-1}) are set approximately midway between 1900 and 2000 values. After Beattie (2001), the 2000 P/B ratio for juvenile flatfish (1.935 yr^{-1}) is based on a reported daily growth rate of 0.53%, while adult flatfish is based 0.026% per day (0.949 yr^{-1}) (Smith *et al.* 1995). Q/B rates for the 1750 and 1900 models were estimated by Ainsworth *et al.* (2002) using the empirical formula of Palomares and Pauly (1989) (juveniles: 6.31 yr^{-1} ; adults: 4.21 yr^{-1}). Adult Q/B rates for 1950 and 2000 are based on estimates from the Bering Sea (Wakabayashi, 1986), while the juvenile consumption rate is based on laboratory work (Smith *et al.*, 1995), as cited in Beattie (2001).

Catch of flatfish is assumed negligible in the 1750 and 1900 models. Landings in 1950 ($0.073 \text{ t}\cdot\text{km}^{-2}$) are based on commercial data for dover, petrale, English and rock sole from Fargo (1999), and on flounder catch from DFO (2004e). The value was increased by 10% to cover remaining flatfish species. Catch in 2000 ($0.058 \text{ t}\cdot\text{km}^{-2}$) is based on official commercial catch statistics (DFO, 2004b) for sole and flounders, and also includes IUU from Ainsworth and Pitcher (2005c).

Diet is based on Beattie (2001) for all periods. The forage fish prey item was divided between forage fish and eulachon in the northern BC models.

Juvenile and adult halibut

Halibut (*Hippoglossus stenolepis*) have been hunted for a long time on the Pacific coast. It was an important food fish for First Nations in pre-contact times (Carrothers, 1941); especially in Haida Gwaii where salmon were less available (Swan, 1884). Although no quantitative estimates were found, I assume that adult biomass in 1750 was the highest of all periods

modelled ($1.0 \text{ t}\cdot\text{km}^{-2}$), about 60% higher than at present. Ainsworth *et al.* (2002) estimated pre-contact juvenile biomass as $0.445 \text{ t}\cdot\text{km}^{-2}$ and that value is used here. Schreiber (2002) suggested that the biomass during pre-contact times was higher than at the turn of the century, so for 1900 I use a lower estimate, approximately equal to the present-day (Beattie 2001) (juveniles: $0.296 \text{ t}\cdot\text{km}^{-2}$; adults: $0.608 \text{ t}\cdot\text{km}^{-2}$). Adult biomass in the 1950 model is based on cohort analysis (Quinn *et al.*, 1985) ($0.429 \text{ t}\cdot\text{km}^{-2}$); and I assume that the study area comprises 30% of International Pacific Halibut Commission regulatory area 2B. Juvenile biomass for 1950 is taken from Ainsworth *et al.* (2002) ($0.406 \text{ t}\cdot\text{km}^{-2}$), who assumed a similar biomass to the adult pool. Adult biomass in the 2000 model is based on values in Clark and Hare (2001) corrected for area ($0.608 \text{ t}\cdot\text{km}^{-2}$), and juvenile biomass is assumed to equal adult. A small negative biomass accumulation in was set in 2000 for juveniles ($-0.011 \text{ t}\cdot\text{km}^{-2}$), based on the simulation results with the 1950 model under historic drivers.

Production rates in 1750 (juveniles: 0.099 yr^{-1} ; adults: 0.067 yr^{-1}) and 1900 (juveniles: 0.116 yr^{-1} ; adults 0.084 yr^{-1}) are assumed equal to fishing mortality plus natural mortality. Ainsworth *et al.* (2002; Appendix B Table B1) calculated fishing mortality as catch divided by biomass, and they estimated natural mortality from the empirical relationship of Pauly (1980). The P/B ratios for adults in 1950 and 2000 (0.4 yr^{-1}) are based on Beattie (2001) who cited Clark and Parma (2000). As in Beattie (2001), I assume that the juvenile production rate 50% higher than adult (0.6 yr^{-1}). The Q/B rate for 1750 and 1900 models is based on the empirical formula of Palomares and Pauly (1989), as estimated by Ainsworth *et al.* (2002: Appendix B Table B2) (juveniles: 2.556 yr^{-1} ; adults: 1.704 yr^{-1}). The consumption rate for adults in 1950 and 2000 (1.10 yr^{-1}) is taken from Beattie (2001), who assumed a daily ration that is 0.3% of body mass, while juveniles are assumed to be one third more productive than adults: 1.46 yr^{-1} .

In pre-contact times halibut were baited with cuttlefish, and caught by wooden hooks attached to lines of plaited cedar-bark or kelp, which had been treated to make it tough and pliable. (Anon, 1952). It is estimated that First Nations caught as much as 1.4 thousand tonnes of halibut per year prior to the commercial fisheries (Carrothers, 1941). This figure is used in 1750 assuming an even split between juveniles and adults (juveniles/adults: $0.01 \text{ t}\cdot\text{km}^{-2}$). Commercial fishing for Pacific halibut began in the 1880s and by 1909 fishermen already noticed that most of the

formerly productive inshore areas had been depleted, and they began actively searching for previously unfished offshore grounds (Schreiber 2002). Catch in 1900 is based on Bell (1981); it has been standardized to the model area (adults: $0.018 \text{ t}\cdot\text{km}^{-2}$). Catch in 1950 (juveniles: $0.7 \text{ kg}\cdot\text{km}^{-2}$; adult: $0.072 \text{ t}\cdot\text{km}^{-2}$) is also based on Bell (1981), but includes IUU catch from Ainsworth and Pitcher (2005c) and recreational catch, which is assumed to be 9% of the present day recreational catch (Forrest, 2002). Commercial catch in 1950 is divided by gear type according to ratios found in DFO (2004e) for the year 1951. Catch in 2000 (juveniles: $0.029 \text{ t}\cdot\text{km}^{-2}$; adult: $0.046 \text{ t}\cdot\text{km}^{-2}$) is based on IPHC (2003), and include IUU catch from Ainsworth and Pitcher (2005c), as well as recreational catch from Forrest (2002); it is assumed that juveniles constitute 5% of the total recreational catch in 2000.

Diet for halibut is based on Beattie's (2001) estimate from Yang (1995). The prey item forage fish is divided into forage fish and eulachon groups. Changes have been made in all models for balancing and/or fitting dynamics to data.

Juvenile and adult Pacific cod

For 1750 and 1900, biomass values for Pacific cod (*Gadus macrocephalus*) is taken from Ainsworth *et al.* (2002), who used Ecopath to estimate the figures: (juveniles: $0.464 \text{ t}\cdot\text{km}^{-2}$; adults: $2.039 \text{ t}\cdot\text{km}^{-2}$) in 1750, and (juveniles: $0.307 \text{ t}\cdot\text{km}^{-2}$; adults: $1.219 \text{ t}\cdot\text{km}^{-2}$) in 1900. Adult biomass in 1950 ($0.348 \text{ t}\cdot\text{km}^{-2}$) is taken from a delay-difference stock production model described by Sinclair *et al.* (2001). The data refers to Hecate Strait, so I assume it can be applied to the whole study area. The first data point given by those authors refers to 1956, so I assume that the value in 1950 is similar. The biomass of juveniles in 1950 ($0.185 \text{ t}\cdot\text{km}^{-2}$) was decided based on the assumption that there were, at that time, a similar proportion of juveniles to adults as in the present day - and that they constitute approximately 35% of total biomass. That proportion originates from the Gulf of Alaska (Niggol, 1982), as cited in Beattie (2001); it assumes that juveniles consist of ages 0-3 years. Adult biomass in 2000 is $0.163 \text{ t}\cdot\text{km}^{-2}$ (Sinclair *et al.*, 2001), and juvenile biomass is assumed to account for 35% of the total cod biomass ($0.089 \text{ t}\cdot\text{km}^{-2}$). A small negative biomass accumulation ($-9 \text{ kg}\cdot\text{km}^{-2}$) was accepted for juveniles in the 1950 model to improve dynamic fit of the adult group, and a small biomass accumulation was

accepted for 2000 based on simulations using the 1950 model under historic mortality and production drivers (juveniles: $5 \text{ kg} \cdot \text{km}^{-2}$; adults: $-2 \text{ kg} \cdot \text{km}^{-2}$).

The production rate in 1750 and 1900 (juveniles: 0.258 yr^{-1} ; adults: 0.174 yr^{-1}) is assumed to equal natural mortality, and is based on calculations done by Ainsworth *et al.* (2002; Appendix B Table B1) using the empirical formula of Pauly (1980). The P/B ratio in 1950 and 2000 is from Beattie (2001) (juveniles: 1.98 yr^{-1} ; adults: 1.32 yr^{-1}). The adult rate is based on total mortality estimated by Westrheim (1997); as in Beattie (2001), I assume that juveniles have a production rate 50% larger than adults. For 1750 and 1900, the consumption rate is based on calculations by Ainsworth *et al.* (2001; Appendix B Table B2), who used the empirical formula of Palomares and Pauly (1989) (juveniles: 3.43 yr^{-1} ; adults: 2.29 yr^{-1}). The Q/B ratio in 1950 and 2000 (juveniles: 7.50 yr^{-1} ; adults: 4.00 yr^{-1}) is based on Beattie (2001), who for adults used a daily ration rate of 1.1% (from Paul *et al.*, 1990), and for juveniles assumed a similar juvenile/adult consumption ratio as seen in sablefish.

Irwin (1984) reports that First Nations caught Pacific cod using lures and spears. Although no quantitative estimates for pre-contact catch were found, I assume there is a small annual aboriginal catch in the 1750 and 1900 models of $1 \text{ kg} \cdot \text{km}^{-2}$ (70 tonnes per year for the study area). Catches for 1950 ($0.056 \text{ t} \cdot \text{km}^{-2}$) and 2000 ($0.054 \text{ t} \cdot \text{km}^{-2}$) are based on the delay-difference model of Sinclair *et al.* (2001), and include IUU estimates from Ainsworth and Pitcher (2005c). Catch is apportioned across gear types according to the ratio found in catch records (DFO, 2004e).

For all models, the diets of juvenile and adult Pacific cod were obtained from Beattie (2001) and adapted for new model groupings by assuming that 1/6th of the proportion of forage fish in their diet is obtained from eulachon. His estimates were based on Bering Sea stomach content analysis in Yang (1995). Diet for adults in all periods is modified to balance the model. Data presented by Sinclair *et al.* (2001) suggested that Pacific cod went through several large spikes in abundance between 1950 and 2000, but the present day abundance is purported to have returned to the levels of 50 years ago or even lower. The model was not able to recreate this trend until the contribution of Pacific cod to the diet of seals and sea lions was increased

Juvenile and adult sablefish

Sablefish (*Anoplopoma fimbria*) biomass in 1750 (juveniles: $0.180 \text{ t}\cdot\text{km}^{-2}$; adults: $0.191 \text{ t}\cdot\text{km}^{-2}$) is estimated by Ecopath in Ainsworth *et al.* (2002), as was juvenile biomass in 1900 ($0.108 \text{ t}\cdot\text{km}^{-2}$). Adult biomass in 1900 was assumed to be $0.6 \text{ t}\cdot\text{km}^{-2}$, which is twice the present-day estimate of Walters and Bonfil (1999). I assume that adult biomass in 1950 was similar to 1900, ($0.6 \text{ t}\cdot\text{km}^{-2}$), while juvenile biomass in 1950 ($0.238 \text{ t}\cdot\text{km}^{-2}$) is from Ainsworth *et al.* (2002), who assumed that was a similar ratio of juveniles to adults as in the present day. Juvenile biomass in 2000 ($0.119 \text{ t}\cdot\text{km}^{-2}$) is from Beattie (2001), who assumed that 30% of the stock consists of juveniles by weight based on a qualitative account by Haist *et al.* (1999), while the estimate of adult biomass ($0.269 \text{ t}\cdot\text{km}^{-2}$) is revised from Haist *et al.* (2001).

The P/B estimate for 1750 and 1900 (juveniles: 0.273 yr^{-1} ; adults: 0.174 yr^{-1}) is based on natural mortality calculated in Ainsworth *et al.* (2004) using the empirical relationship of Pauly (1980). The adult P/B estimate for 1950 and 2000 (0.276 yr^{-1}) was taken from Beattie (2001), and is based on total mortality: the sum of fishing mortality (Haist *et al.*, 1999) and natural mortality (assumed 0.08 yr^{-1}). Juvenile P/B in 1950 and 2000 (0.6 yr^{-1}) is based on Beattie (2001), and represents the value reported in McFarlane and Beamish (1983). Consumption rates are from Beattie (2001). For all periods, juvenile Q/B (3.73 yr^{-1}) is from McFarlane and Beamish (1983); for all periods, adult Q/B (7 yr^{-1}) is from Livingston (1996).

I assume that there is no catch for sablefish in 1750 or 1900. Landings in 1950 ($0.006 \text{ t}\cdot\text{km}^{-2}$) are taken from DFO (2004e) for the year 1951; the total value is partitioned across gear types according to the historical record, and it also includes a small amount of IUU catch from Ainsworth and Pitcher (2005c). I assume a similar amount of discards in 1950 as in the present day, about $3 \text{ kg}\cdot\text{km}^{-2}$ as estimated from unpublished DFO observer data reported in Beattie (2001). Catch in 2000 ($0.041 \text{ t}\cdot\text{km}^{-2}$) is taken from DFO (1999c), and similarly includes IUU estimates and discards (same data sources as 1950).

Diet for all periods is taken from Beattie (2001), which he based on stomach content analysis from the west coast of Vancouver Island (Tanisichuk, 1997). Diet remains unchanged from

Beattie's (2001) estimate, except for minor modifications to the adult matrix in 1750, which were made for the purpose of balancing.

Juvenile and adult lingcod

Lingcod populations in Hecate Strait have been reduced greatly by fishing over the last 50 years (Martell, 1999; Beattie 1999), although the depletion is far worse in southern BC (Martell *et al.*, 2000). Pre-contact biomass is taken from Ainsworth *et al.* (2002), who used Ecopath to estimate the values (juveniles: $0.006 \text{ t}\cdot\text{km}^{-2}$; adults: $0.148 \text{ t}\cdot\text{km}^{-2}$), as were 1900 biomass values (juveniles: $0.005 \text{ t}\cdot\text{km}^{-2}$; adults: $0.119 \text{ t}\cdot\text{km}^{-2}$). Juvenile biomass in 1950 ($0.078 \text{ t}\cdot\text{km}^{-2}$) is based on Ainsworth *et al.* (2002), who assumed a similar proportion to adults as in the present day. Adult biomass in 1950 ($0.104 \text{ t}\cdot\text{km}^{-2}$) is based on the stock reduction analysis in Martell (1999); it is assumed that his value for 1955 can be applied to 1950. Biomass in 2000 (juveniles: $0.031 \text{ t}\cdot\text{km}^{-2}$; adults: $0.039 \text{ t}\cdot\text{km}^{-2}$) is also based on Martell (1999). A small biomass accumulation was accepted for 1950 to assist dynamic fitting (juveniles: $-0.016 \text{ t}\cdot\text{km}^{-2}$; adults: $0.001 \text{ t}\cdot\text{km}^{-2}$), while 2000 biomass accumulation is based on the results of forward simulations using the 1950 model with historic mortality and production drivers (juveniles: $0.002 \text{ t}\cdot\text{km}^{-2}$; adults: $-0.001 \text{ t}\cdot\text{km}^{-2}$).

The P/B ratios in 1750 (juveniles: 0.389 yr^{-1} ; adults: 0.262 yr^{-1}) are based on total mortality as calculated in Ainsworth *et al.* (2002; Appendix B Table B1). Juvenile P/B in 1900 is assumed to equal the pre-contact rate, but adult P/B was increased to 0.3 yr^{-1} to facilitate catch and biomass dynamics similar to the historical record (catch: Cass *et al.*, 1990; biomass: Martell, 1999). For adults in 1950 and 2000, I use the P/B value of Beattie (2001) (0.8 yr^{-1}), who consulted a tagging study based in the Strait of Georgia (Smith *et al.*, 1990). The production rate of juveniles in 1950 is set high (1.4 yr^{-1}) to achieve a better dynamic fit to data in the adult group, while juveniles in 2000 are assumed to be 50% more productive than adults (1.2 yr^{-1}). The Q/B rate in 1750 and 1900 (juveniles: 3.94 yr^{-1} ; adults: 2.80 yr^{-1}) is based on Ainsworth *et al.* (2002; Appendix B Table B2), who used the empirical formula of Palomares and Pauly (1989). The Q/B rates in 1950 for juveniles and adults were increased and decreased respectively from the present-day estimate of Beattie (2001) to help fit the model to data (juveniles: 3.5 yr^{-1} ; adults: 3.0 yr^{-1}). For 2000, I use the estimate of Beattie (2001) for juveniles and adults (3.3 yr^{-1}).

First Nations people caught lingcod with wooden gorges in pre-contact times, but the species was of minor importance (Vasconcellos and Pitcher, 2002h). I assume a small catch of $0.5 \text{ kg}\cdot\text{km}^{-2}$ for the 1750 model. Cass *et al.* (1990) provide a catch estimate for 1900 for the whole BC coast. I scaled their time series to agree with the one provided by King and Surry (2001), who gave data for the correct area (but began in 1927). Using the scaled estimate, catch in 1900 is $5 \text{ kg}\cdot\text{km}^{-2}$. In the 1950 model, groundfish trawl, groundfish hook and line and recreational fisheries together catch $6 \text{ kg}\cdot\text{km}^{-2}$ of adult lingcod based on King and Surry (2001). Catch is apportioned across gear types according to the ratios found in Cass *et al.* (1990), and includes a small recreational catch (5%) for juveniles and adults. Lingcod catch in 2000 ($0.02 \text{ t}\cdot\text{km}^{-2}$) is based on King and Surry (2001) and includes a small recreational catch for both juveniles and adults from Cass *et al.* (2001). Adult lingcod is discarded by groundfish trawl in the 1950 and 2000 models (Beattie 2001) ($1 \text{ kg}\cdot\text{km}^{-2}$).

The diet of juvenile lingcod was adapted from the text of Cass *et al.* (1990), who suggested that juvenile lingcod feed on herring, forage fish, juvenile flatfish, Pacific cod, shrimp and invertebrates. I assume that each prey item constitutes 20% of the diet as in Beattie (2001), and divide among appropriate functional groups for the northern BC models. Diet for adults is based on Cass *et al.* (1986) as cited in Beattie (2001).

Shallow water benthic fish

This group includes sculpins (Cottidae), blennies (Bleniidae), poachers (Agonidae), gobies (Gobiidae), greenlings (Hexagrammidae, except lingcod), eelpouts (Zoarcidae), northern clingfish (*Gobiesox maeandricus*), red Irish lord (*Hemilepidotus hemilepidotus*), cabezon (*Scorpaenichthys marmoratus*), snowy snailfish (*Liparis pulchrettus*), cutthroat trout (*Oncorhynchus clarki clarki*) and white sturgeon (*Acipenser transmontanus*). Biomass for 1750 and 1900 was estimated by Ecopath in Ainsworth *et al.* (2002) ($7.506 \text{ t}\cdot\text{km}^{-2}$ and $4.465 \text{ t}\cdot\text{km}^{-2}$, respectively). Biomass for 1950 and 2000 is based on Beattie's (2001) present-day estimate of $0.509 \text{ t}\cdot\text{km}^{-2}$, which he obtained from an eastern Bering Sea survey (Wakabayashi, 1986).

Ecopath is used to estimate the P/B rate of shallow water benthic fish in 1750 and 1900 (0.266 yr^{-1}) in Ainsworth *et al.* (2002). The production rate for 1950 and 2000 is assumed equal to the

natural rate of mortality (1.5 yr^{-1}) based on Beattie's (2001) estimate that used the empirical relationship of Pauly (1980). The Q/B rate for 1750 and 1900 (2.1 yr^{-1}) is based on the empirical formula of Palomares and Pauly (1989), and is calculated in Ainsworth *et al.* (2002; Appendix B Table B2). The Q/B rate for 1950 and 2000 (5.26 yr^{-1}) is based on the mean consumption rate of poachers, eelpouts and sculpins given in Wakabayashi (1986) and cited by Beattie (2001).

In the 2000 model, a small amount of shallow water benthic fish is caught and discarded by groundfish trawlers (Beattie, 2001) and shrimp trawlers (Hay *et al.*, 1999) totaling $1.24 \text{ kg} \cdot \text{km}^{-2}$.

Diet for this group is modified from Beattie (2001), to accommodate the northern BC functional groups and to permit balancing (especially for 1900). The source information is from stomach content analysis in the eastern Bering Sea (Wakabayashi, 1986).

Skates

This compartment consists mostly of skates, although the few stingrays and sharks that are present in the system are also included. The skates include the big skate (*Raja binoculata*), longnose skate (*R. rhina*), starry skate (*R. stellulata*), black skate (*Bathyraja interrupta*) and the deep-sea skate (*B. abyssicola*) (Beattie 2001), while sharks include the tope shark (*Galeorhinus galeus*), great white shark (*Carcharodon carcharias*), broadnose sevengill shark (*Notorynchus cepedianus*), bluntnose sixgill shark (*Hexanchus griseus*), blue shark (*Prionace glauca*) and basking shark (*Cetorhinus maximus*). Stingrays include the diamond stingray (*Dasyatis dipterura*) and pelagic stingray (*Pteroplatytrygon violacea*). Biomass for 1750 and 1900 is estimated by Ecopath in Ainsworth *et al.* (2002) as $0.239 \text{ t} \cdot \text{km}^{-2}$ and $0.167 \text{ t} \cdot \text{km}^{-2}$, respectively. Biomass in 2000 ($0.335 \text{ t} \cdot \text{km}^{-2}$) is taken from Beattie (1999), who cites Fargo *et al.* (1990). Biomass in 1950 was reduced from this estimate to $0.3 \text{ t} \cdot \text{km}^{-2}$ because the LEK trend suggested that biomass has increased over the last 50 years (Ainsworth and Pitcher, 2005a).

The P/B ratio used in the 1750 and 1900 models is based on natural mortality (0.15 yr^{-1}) as estimated by Ainsworth *et al.* (2002; Appendix B Table B1) using the empirical formula of Pauly (1980). The P/B ratio used in the 1950 and 2000 models (0.31 yr^{-1}) is taken from Beattie (1999).

The Q/B rate used in the 1750 and 1900 models (1.2 yr^{-1}) is reduced slightly from the estimate of Beattie (1999). The 1950 and 2000 models use his value directly (1.24 yr^{-1}).

I assume there is no catch for skates in 1750 or 1900. A very small catch of skate was included in the 1950 model, $0.09 \text{ kg} \cdot \text{km}^{-2}$ (DFO, 2004e). Only half this amount was indicated by the DFO catch records for groundfish trawl in 1951, but an equal value was arbitrarily assigned to longline in order to account for some level of bycatch. Catch in 2000 ($0.029 \text{ t} \cdot \text{km}^{-2}$) is based on observer data for groundfish trawl (Beattie, 2001).

The diet of skates is represented by the Atlantic species thorny skate (*Amblyraja radiata*) from Robichaud *et al.* (1991) as cited in Beattie (2001). Diet is adapted for the northern BC model groupings: forage fish are divided into forage fish and eulachon, and the proportion of the diet attributed to benthic invertebrates is divided into infaunal carnivorous invertebrates and infaunal invertebrate detritivores.

Small and large crabs

Crabs are divided into large crabs with a carapace length of more than 120 mm, and small crabs with carapace length less than 120 mm. The large crabs include mostly dungeness crab (*Cancer magister*), but also the red rock crab (*C. productus*), tanner crab (*Chionoecetes bairdi*) and king crab (*Paralithodes* spp.), while the small crabs include the juveniles ($< 120 \text{ mm}$ carapace length) and other small crabs like kelp crab (*Pugettia producta*) (Beattie 2001). Biomass of small and large crabs in 1750 (small: $2.407 \text{ t} \cdot \text{km}^{-2}$; large: $0.652 \text{ t} \cdot \text{km}^{-2}$) is estimated by Ecopath in Ainsworth *et al.* (2002), as was biomass in 1900 (small: $1.458 \text{ t} \cdot \text{km}^{-2}$; large: $0.388 \text{ t} \cdot \text{km}^{-2}$), 1950 (large: $0.506 \text{ t} \cdot \text{km}^{-2}$) and 2000 (small: $0.650 \text{ t} \cdot \text{km}^{-2}$; large: $0.456 \text{ t} \cdot \text{km}^{-2}$). Ecopath estimated the biomass of small crabs in 1950 as $0.599 \text{ t} \cdot \text{km}^{-2}$ by assuming an EE of 0.95. A small biomass accumulation is used in the 1950 model (large: $-0.05 \text{ t} \cdot \text{km}^{-2}$) to balance the model, and a small biomass accumulation is used in 2000 (small: $-0.01 \text{ t} \cdot \text{km}^{-2}$; large: $0.002 \text{ t} \cdot \text{km}^{-2}$) based on dynamic simulations of the 1950 model, driven by historic mortality and production rates.

The P/B used for small crabs in all models (3.5 yr^{-1}) is based on Beattie (2001); he assumed a production rate three times greater than large crabs. Beattie (2001) approximated the P/B for

large crabs as 1.5 yr^{-1} based on total mortality rates from Clayoquat Sound (Smith and Jamieson, 1991). For all periods, the Q/B for small and large crabs is from Beattie (2001) (small: 14 yr^{-1} ; large: 5 yr^{-1}). He estimated the value for small crabs by assuming a P/Q ratio of 0.2, and he based the value for large crabs on red king and tanner crab data from Alaskan waters (Wakabayashi, 1986).

Catch of large crabs by traps and trawls in 1950 ($0.005 \text{ t}\cdot\text{km}^{-2}$) is based on historical records from 1951 (DFO 1995), and includes a recreational estimate from Forrest (2002), who assumed 9% of the current sports catch. Catch in 2000 ($0.027 \text{ t}\cdot\text{km}^{-2}$) is based on DFO 2004b, and includes a recreational estimate from Forrest (2002). In the 2000 model groundfish trawl is said to discard $0.037 \text{ kg}\cdot\text{km}^{-2}$ of small crabs and $0.225 \text{ kg}\cdot\text{km}^{-2}$ of large crabs based on observer data (Beattie, 2001).

Diet of small crabs is based on juvenile dungeness crab, and the diet of large crabs is based on dungeness crab (Bernard 1981); the 1750 and 2000 models have been modified for balancing.

Commercial shrimp

This group includes prawn and shrimp (Pandalidae): smooth shrimp (*Pandalus jordani*), spiny shrimp (*P. borealis*), pink shrimp (*P. montagui*), coonstripe (*P. danae*), humpback shrimp (*P. hypsinotus*), sidestripe (*Pandalopsis disbar*) and prawn (*P. platycerus*) (Beattie, 2001). The biomass of shrimp in 1750 and 1900 ($0.07 \text{ t}\cdot\text{km}^{-2}$ and $0.047 \text{ t}\cdot\text{km}^{-2}$, respectively) was taken from Ainsworth *et al.* (2001); they used Ecopath to estimate the values. The 2000 value ($0.2 \text{ t}\cdot\text{km}^{-2}$) is based on pink shrimp and sidestripe shrimp biomass estimates given by Rutherford *et al.* (2004), which have been scaled for area. Since the revised 2000 value is much larger than the estimate made by Ainsworth *et al.* (2001), I also increased the 1950 estimate to $0.15 \text{ t}\cdot\text{km}^{-2}$. I therefore assume a 33% increase over the last 50 years, which is commensurate with the trend suggested in Ainsworth *et al.* (2002). The 2000 model uses a small biomass accumulation ($2 \text{ kg}\cdot\text{km}^{-2}$) based on simulations using the 1950 model, driven by historic production and mortality rates.

Beattie (2001) calculated the production rate of shrimp as 11.48 yr^{-1} based on data collected of southwestern Vancouver Island by Jarre-Teichmann and Gu  nette (1996). This value was used

in 1950 and 2000, while the 1750 and 1900 production rate was assumed to be 50% less (5.7 yr^{-1}) to represent an unexploited population. Consumption rates for 1750 and 1900 (22.8 yr^{-1}) and 1950 and 2000 (45.9 yr^{-1}) are taken from Ainsworth *et al.* (2002); they assumed a P/Q ratio of 0.25.

I assume there was no catch for shrimp in 1750 or 1900. Catch in 1950 is based on shrimp trawl landings in DFO (2004e), and includes a small recreational catch from Forrest (2002) equal to 9% of the 2000 sport estimate. I have also assumed a small catch using traps to provide a total catch of $1.65 \text{ kg} \cdot \text{km}^{-2}$ in 1950. Catch in 2000 is based on Beattie (2001), who used 1996-1998 values from the shrimp trawl fishery (DFO 1999d) and prawn trap fishery (DFO 1999e). I also use recreational catch from Forrest (2002) to provide a total present-day catch estimate of $3.67 \text{ kg} \cdot \text{km}^{-2}$.

Beattie (2001) cites qualitative diet information in Bundy *et al.* (2000); I use a variant of his data matrix. I have assumed that zooplankton includes copepods as well as euphausiids, and I also assume a small (10%) invertebrate diet component (i.e., infaunal detritivorous invertebrates).

Epifaunal and infaunal invertebrates

Epifaunal invertebrates include Echinodermata, Mollusca, Cnidaria and Amphipoda, while infaunal carnivorous invertebrates include mostly Annelida (polychaetes). Infaunal detritivorous invertebrates include Nemertea, Gastropoda, Pelecypoda, Scaphopoda, Ostracoda, Cumacea, Isopoda, Amphipoda, Decapoda, Sipunculida, Ophiuroidea, Echinoidea, and Holothuroidea that feed on detritus. Biomass of epifaunal invertebrates for all periods was estimated by Ecopath in Ainsworth *et al.* (2002) (1750: $42.8 \text{ t} \cdot \text{km}^{-2}$; 1900: $28.6 \text{ t} \cdot \text{km}^{-2}$; 1950: $11.2 \text{ t} \cdot \text{km}^{-2}$; 2000: $13.4 \text{ t} \cdot \text{km}^{-2}$). Biomass estimates for carnivorous and detritivorous infaunal invertebrates for 1750 (carnivorous: $8.2 \text{ t} \cdot \text{km}^{-2}$; detritivorous: $39.3 \text{ t} \cdot \text{km}^{-2}$) were estimated by Ecopath in Ainsworth *et al.* (2002), while biomass in both groups is assumed to have remained constant since 1900. Biomass of these groups in 1900, 1950 and 2000 is based on the functional group 'benthic infauna' used in Beattie (2001). The biomass of polychaetes was extracted from his group and is used to represent carnivorous infauna in the northern BC model ($13.2 \text{ t} \cdot \text{km}^{-2}$); the remainder represents detritivorous infauna ($34.3 \text{ t} \cdot \text{km}^{-2}$). A large negative biomass accumulation is

accepted in the 1950 model for epifaunal invertebrates ($-0.224 \text{ t}\cdot\text{km}^{-2}$). This value in the base year allows the population density to remain stable under historic mortality and production drivers; otherwise, an erroneous population increase is seen (Ainsworth and Pitcher, 2005a). A large negative biomass accumulation is accepted for epifaunal invertebrates in the 2000 model (-0.194), based on simulations of the 1950 model under historic mortality and production drivers. Carnivorous and detritivorous infaunal invertebrates also receive biomass accumulations in the 2000 model based on simulations ($0.270 \text{ t}\cdot\text{km}^{-2}$ and $-0.377 \text{ t}\cdot\text{km}^{-2}$, respectively).

For all periods, the P/B rate for epifaunal invertebrates (1.448 yr^{-1}) and detritivorous infaunal invertebrates (1.349 yr^{-1}) is based on Beattie (2001); he used an empirical formula from Anon (1993) to estimate the figures. The P/B rate for carnivorous infaunal invertebrates (2 yr^{-1}) is based on Jarre-Teichmann and Gu  nette (1996). As in Beattie (2001), the Q/B ratio for all invertebrate groups was based on the assumption that P/Q equals 0.09.

Vasconcellos and Pitcher (2002h) suggest that aboriginal fisheries for invertebrates have always existed, but with no estimate of catch I assume $0.5 \text{ kg}\cdot\text{km}^{-2}$ each for epifaunal invertebrates and infaunal detritivores in the 1750 model. I assume a smaller catch of $0.1 \text{ kg}\cdot\text{km}^{-2}$ each in the 1900 model, as there was a large reduction in First Nations people from the pre-contact period. Historic catch records (DFO, 2004e) reveal that the commercial harvest of epifaunal invertebrates in 1950 was approximately 37.7% of the present day; this guideline is used to calculate the 1950 catch estimate, $0.0294 \text{ t}\cdot\text{km}^{-2}$. That amount accounts for butter clams primarily and includes some recreational catch (assumed 9% of the present-day based on Forrest, 2002). In the 2000 model, a small amount of epifaunal invertebrates are caught by groundfish trawlers ($0.08 \text{ kg}\cdot\text{km}^{-2}$) from observer records in Beattie (2001). The largest directed fishery for epifaunal invertebrates ($0.078 \text{ t}\cdot\text{km}^{-2}$) is for sea urchins, *Stronglyocentrotus* spp., and sea cucumbers, primarily *Parastichopus californicus* (Beattie 2001). Forrest (2002) cites an unpublished DFO survey that identifies a small recreational catch composed of clams, oysters and other shellfish. Epifaunal invertebrates ($0.002 \text{ t}\cdot\text{km}^{-2}$) and detritivorous infaunal invertebrates ($0.003 \text{ kg}\cdot\text{km}^{-2}$) are caught and discarded by the groundfish trawl fishery in the 1950 and 2000 models based on observer records in Beattie (2001).

Carnivorous infaunal invertebrates are said to consume detritus and other invertebrates. I assume that epifaunal invertebrates consume mainly detritus, and some amount of infaunal invertebrates. Epifaunal invertebrates are a populous and controlling group in the northern BC models; their diet was modified to balance the Ecopath models, and to adjust dynamic Ecosim behaviour in the 1950 model.

Carnivorous jellyfish

The biomass of jellyfish in 1750 ($4.625 \text{ t}\cdot\text{km}^{-2}$) and 1900 ($3.363 \text{ t}\cdot\text{km}^{-2}$) was estimated by Ecopath in Ainsworth *et al.* (2002). The 1950 and 2000 biomass estimate ($3.0 \text{ t}\cdot\text{km}^{-2}$) was obtained from Beattie (2001), who halved the value of Mackas (1992) to account for seasonal presence. The P/B ratio for jellyfish (18 yr^{-1}) was obtained from Beattie (2001); it represents an average value from Larson (1986). The Q/B for jellyfish (60 yr^{-1}) is set by assuming a P/Q ratio of 0.3.

I assume there is no directed catch for jellyfish in any period. However, jellyfish are caught as bycatch and discarded ($0.134 \text{ kg}\cdot\text{km}^{-2}$) by the groundfish trawl fishery (Beattie 2001) and by salmon gillnets (C. Ainsworth, *pers. obs.*). For the 1950 model, I assume a bycatch from only salmon gillnets ($0.1 \text{ kg}\cdot\text{km}^{-2}$). I assume that jellyfish consume mainly suspended detritus, zooplankton and other jellies.

Euphausiids and copepods

Ninety percent of euphausiid biomass consists of three species: *Thysanoessa spinifera*, *T. longipes* and *Euphausia pacifica* (Beattie, 2001). Copepods include *Pseudocalanus* spp., *Oithona* spp. and *Acartia* spp. (Beattie 2001). Biomass of euphausiids in 1750 and 1900 was estimated by Ecopath in Ainsworth *et al.* (2002) as $22.7 \text{ t}\cdot\text{km}^{-2}$ and $12.6 \text{ t}\cdot\text{km}^{-2}$, respectively; biomass of copepods was similarly estimated as $13.1 \text{ t}\cdot\text{km}^{-2}$ in 1750 and $8.7 \text{ t}\cdot\text{km}^{-2}$ in 1900. The biomass estimates of euphausiids ($8.70 \text{ t}\cdot\text{km}^{-2}$) and copepods ($4.7 \text{ t}\cdot\text{km}^{-2}$) in the 1950 and 2000 models were obtained from Beattie (2001). The P/B ratio for euphausiids (6 yr^{-1}) is based on *E. pacifica* (Iguchi and Ikeda, 1999) as cited in Beattie (2001); he found an estimate for the copepod production rate (27 yr^{-1}) in Robinson and Ware (1994). Consumption rate for euphausiids and

copepods is based on the assumption that P/Q equals 0.3. Euphausiids are assumed to eat copepods and phytoplankton; copepods are assumed to eat phytoplankton.

Corals and sponges

Coral and sponge biomass is calculated based on data in Conway (2002). Areas that have not been affected by trawl damage or other stressors are thought to have approximately $300 \text{ t}\cdot\text{km}^{-2}$ of sponge reef biomass, and there are approximately 700 km^2 of sponge reef in the study area. This provides a pristine estimate of $3.2 \text{ t}\cdot\text{km}^{-2}$ for the 1750 and 1900 models. Assuming that 30-50% of coral and sponge biomass has been removed by fishing effects and anthropogenic damage, the resulting biomass for 1950 and 2000 is $1.9 \text{ t}\cdot\text{km}^{-2}$. Conway (2002) suggested a P/B ratio of 0.01 yr^{-1} , which I used for all models. I assume a Q/B rate of 2.0 yr^{-1} . Corals and sponges filter detritus from the water column in all models.

Phytoplankton and macrophytes

Beattie (2001) calculates phytoplankton biomass ($15.4 \text{ t}\cdot\text{km}^{-2}$) based on the mean annual density from Robinson *et al.* (1999). All models use this value. Macrophyte biomass is assumed to be $5.3 \text{ t}\cdot\text{km}^{-2}$ in 1900, 1950 and 2000 based on Beattie (2001). Pre-contact macrophyte biomass was assumed to be two times higher ($10.6 \text{ t}\cdot\text{km}^{-2}$), since there were more sea otters to keep grazing populations of urchins suppressed (see Steneck *et al.*, 2002). Beattie's (2001) P/B rate for phytoplankton (178.5 yr^{-1}) was calculated using information in Ware and McFarlane (1989), while the P/B rate for macrophytes (5.26 yr^{-1}) was based on data for *Macrocystis pyrifera* from Lobban and Harrison (1994).

Discards

This functional group represents fishery discards, which are fed upon by birds and other scavengers. The discard pool is assumed to be zero in the 1750 model. The discard pool in 1900 is assumed to be small ($0.001 \text{ t}\cdot\text{km}^{-2}$), while the pool for 1950 and 2000 ($0.072 \text{ t}\cdot\text{km}^{-2}$) is based on evidence from groundfish trawl observer data (Beattie, 2001).

5.2 Fisheries

Present-day

The fleet structure in the 2000 model contains 17 gear types: groundfish trawl, groundfish hook and line, halibut hook and line, sablefish trap, salmon gillnet, salmon seine, salmon troll, salmon troll freezer, longline, herring gillnet, herring seine, crab trap, shrimp/prawn trap, shrimp trawl, eulachon, other invertebrates, and recreational. The gear structure is based on Beattie (2001), which was later modified by Pitcher *et al.* (2002c) using expert opinion. Appendix Table A5.1.4 gives landings data for all models; Appendix Table A5.1.5 gives discards data, which is based on observer records in Beattie (2001) and estimates of IUU made in Chapter 4 (see functional group descriptions for landings and discard data sources).

Market prices for the present-day model, reported in Appendix Table A5.1.6, were assembled by Beattie (2001) based on various governmental reports and subsequently adjusted by this author and by S. Heymans to reflect gear-specific prices (*Unpublished manuscript*. UBC Fisheries Centre. Contact: c.ainsworth@fisheries.ubc.ca). For example, troll-caught salmon were assigned a higher value than salmon caught with gillnets or seine nets; groundfish and rockfish caught with hook and line were assigned a higher value than groundfish and rockfish caught using trawl; and trap-caught shrimp were assigned a higher value than trawl-caught shrimp. These gear-specific adjustments were made approximately; they were not based rigorously on data and could be improved upon. Note that the prices for the *lost valley* fleet version of the northern BC models, upon which the forecasts made in this volume rely, are improved upon and reported in Chapter 6 (see section 6.2: The *lost valley* fleet). Operating costs for all gear types was assumed to equal 60% of catch value based on Anon. (1994), and an additional 20% sailing cost was included to bring net profit to 20%, as in Beattie (2001). For trap fisheries, sailing-related costs are reduced by 10% and effort costs are increased by 10%.

Historical

The fleet structure in the 1750 model includes fisheries for sea otters, halibut hook and line, salmon seine, eulachon, herring, whaling, other demersals (cod, etc.) and invertebrates. The 1900 model omits the fishery for sea otter, and adds a setline fishery for demersals. Both 1750 and 1900 fleet structures are based on expert opinion summarized in Pitcher *et al.* (2002c). The 1950 model contains the same fleet structure as the present-day model, with the addition of sea lion shooting, and another seine fishery for miscellaneous catches of forage fish and squid (Pitcher *et al.*, 2002c). Landings data appear in Appendix Table A5.1.4.

5.3 Ecosim parameterization

Stage linking

Trophic ontogeny is represented using 11 juvenile/adult split-pool functional groups (Appendix Table A5.2.1). All values are taken from Fishbase (Froese and Pauly, 2005) or left at Ecosim defaults. A recent addition to Ecosim has succeeded the juvenile/adult group linking routine; the new multi-stanza routine (Christensen and Walters, 2004a) considers a more detailed age structure based on life-stage transition parameters, and state parameters of a 'leading' stanza group. However, the routine was not available at the time of this study.

Feeding parameters

Appendix Table A5.2.2 provides feeding parameters used in the 1950 model. These were altered from default values to improve biomass fit to data. Most assignments are made arbitrarily to affect the generated time series of predation mortality rates. Feeding time adjustment rates for juveniles are set equal to or greater than the adult rates; feeding time adjustment rates are set to zero for some invertebrate groups, which are assumed not vary their feeding time in relation to predator abundance (e.g., sessile filter feeders). All feeding parameters in the 1750, 1900 and 2000 models remain as default.

Trophic flow parameters

The main parameters governing ecosystem behaviour in temporal simulations are the predator-prey vulnerability settings, entered as a matrix in Ecosim. Each predator-prey trophic interaction is assigned a vulnerability coefficient, from one to infinity. The figure is unitless and it describes the maximum increase in predation mortality allowable on that feeding interaction. By assigning a low value, we imply a donor driven density-dependant interaction. In foraging arena theory (Walters and Juanes, 1993; Walters and Korman, 1999; Walters and Martell 2004), the prey can remain hidden or defended during periods of high predator abundance. Predators are never satiated, and handling time or physiological constraints do not limit predation mortality (Essington *et al.*, 2000). By assigning a high value, we imply a predator driven density-independent interaction, in which predation mortality is proportional to the product of prey and predator abundance (i.e., Lotka-Volterra). This implies a high flux rate for prey species in and out of vulnerable biomass pools.

Strict bottom-up control in Ecosim tends to produce unrealistically smooth changes in prey and predator biomass that fail to propagate through the food web (Christensen *et al.*, 2004aa), and can impart an unrealistic degree of resilience to the effects of fishing (Martell *et al.* 2002). Strict top-down control may cause rapid oscillations in biomass and unpredictable simulation behaviour (Christensen *et al.*, 2004a; Mackinson, 2002), and will often produce a complex response surface that is difficult to work with under policy optimizations (Cheung *et al.*, 2002; Ainsworth, C. *Unpublished manuscript*). Mid-range vulnerabilities may offer an adequate solution to temper the dynamics (Okey and Wright, 2004; Ainsworth and Pitcher, *in press*) and most users do assume mixed trophic control in the absence of better information (e.g., default EwE setting = 2). Ideally, parameters should be set independently for each trophic interaction (C. Walters, UBC Fisheries Centre, *pers. comm.*), because the specific vulnerabilities will depend on the particular mode of attack and defense, which is determined by evolutionary ecology.

Although empirical evidence is not usually available to help parameterize vulnerabilities, we can use the dynamic behaviour of Ecosim to determine reasonable settings. By comparing dynamic output of the simulator with observed time series for catch, fishing mortality, biomass and

relative biomass, the user can adjust vulnerabilities to minimize residuals between predicted dynamics and observed time series. A routine for optimizing vulnerability parameters is automated in Ecosim. Manual adjustment of key values can also bring results, especially if the improved dynamics are not earmarked by a reduction in residuals, but by a subjective improvement in catch or biomass forecasts.

Vulnerability parameterization

The automated procedure to fit vulnerability parameters in Ecosim was used initially to establish vulnerability settings for all group interactions in the model. Subsequently, a smaller set of vulnerabilities was optimized, between 10 and 15 key interactions, with manual adjustments being made to achieve specific effects (see next section: Tuning the model). With each revision to Ecopath and Ecosim parameters, further improvements were made to the vulnerability matrix to approach the final solution iteratively.

The fitted vulnerabilities for 1950 were next extended to the other time periods to reflect relative differences in predation mortality during those periods (C. Walters. University of British Columbia, *pers. comm.*). For example, if the predation mortality rate was higher in the past, then the vulnerability parameter, which represents the maximum increase in predation mortality as compared to model baseline, should be proportionately reduced. Therefore, for each trophic interaction, the product of the vulnerability rate and the predation mortality rate is conserved between periods. This approach assumes stationarity in the density-dependant foraging tactics of species.

Where the calculated vulnerability parameters for the 1750, 1900 and 2000 models are less than 1, the value 1.001 is used (a value >1 must be entered). Predator-prey vulnerabilities determined for the 1950 model are listed in Appendix Table A5.2.3.

Tuning the model

Time series patterns of historical fishing mortalities were used to drive fisheries in the model. Coarse corrections were first made to basic Ecopath parameters, primarily the diet matrix. Biomass accumulations were used widely to balance baseline production versus mortality. The relationship of these two variables at time zero is critical in determining a functional group's response to fishing and predation. In the absence of biomass time series with which to tune the model, many EwE modelers assign zero biomass accumulations under baseline conditions. However, this involves a weighty and often overlooked assumption - that baseline exploitation rates are sustainable over the long term, and catch rates perfectly balance with surplus production in all groups. I suggest that a steady-state condition is not a good default assumption. It will usually be incorrect for all ecosystems except unexploited systems near virgin biomass and systems that are nearly collapsed from overfishing. Moreover, assuming that baseline fisheries are sustainable, when they are not, will cause the model to overestimate safe yields under policy forecasts.

Certain commercial groups, which are greatly influenced by fishing, could be fitted to data quite well using only historic fishing mortalities in the 1950 northern BC model. Other groups were coerced to follow time series by finessing predation mortality trends through the diet matrix and vulnerability parameters. The greatest improvement in model fits to data was achieved through a process of 'trading off' predation mortalities among prey items. Any predator group that sees an increase in biomass over the simulated period exerts an increasing amount of predation mortality on its prey. Such a group can be used as a tool to reduce the biomass of a prey group towards the end of a simulation in order to achieve better agreement with time series data. Similarly, a predator that undergoes a reduction in biomass throughout the simulation exerts a diminishing level of predation mortality. By increasing the relative proportion of predation mortality due to that predator under baseline conditions, a prey group can be made to increase in biomass towards the end of the simulation. Through the diet matrix, the relative proportion of mortality suffered by a given predator can be adjusted.

The available array of predators for any given prey group offers a palette of mortality trends that can be applied in varying degrees and overlapped to produce specific temporal changes in prey

abundance. That palette can be extended somewhat through manual adjustment of trophic flow parameters. When vulnerabilities are set high, the predation mortality trends very closely reflect predator biomass. By lowering the vulnerability for a particular interaction, the shape of the predation mortality trend can be altered to reflect the feeding time of the prey, giving the modeler a wider range of options when deliberately imposing mortality trends. Further, the feeding time of the prey can be adjusted through Ecosim feeding parameters. There is far more potential to affect precise changes in ecosystem dynamics through the diet matrix than by altering vulnerabilities, and the technique is more defensible than arbitrary production forcing. The diet composition for predators needs only to remain reasonable for this to be a powerful fitting technique.

Using these general procedures, redistribution of mortality sources, and reshaping of mortality trends, fair improvement can be made to system dynamics. However, the procedure works best for prey groups that are not heavily fished and have no other major sources of mortality, and for groups that have high ecotrophic efficiency (i.e., dynamics are dictated by local mortality sources). This kind of manipulation should be reserved for high trophic level prey groups that are expected to be subject to top-down controlling factors. However, functional group structure in EwE models is often heavily aggregated at the lower end of the food web, (this is true of ecosystem models in general; Hall and Raffaelli, 1993), and so the majority of functional groups can benefit from this kind of adjustment.

Forced catch routine

A recent development in Ecosim (V5.1) allows the user to force model dynamics using a catch series. The simulator removes the expected catch at each time step directly, rather than to infer the amount of catch based on biomass and fishing mortality. The forced catch routine is a useful technique where fishing mortality trends are uncertain, and it can be a valuable diagnostic tool for evaluating historic models because it quickly reveals whether functional groups are capable of sustaining required levels of production. However, it does not use a back-calculation method like stock reduction analysis (an initial confusion); it is mechanistically similar to a previous routine used to force biomass trends.

Unfortunately, few complete time series of commercial fishing mortalities were available for northern BC stocks. As a default, fishing mortality was assumed to equal catch divided by biomass (Appendix Tables A5.3.5 and A5.3.6). Where this assumption is in place, there is little advantage to apply the forced catch routine over the previously available method to drive dynamics by fishing mortality, since our estimates of fishing mortality are exactly proportional to catch when corrected for biomass. In effect, the models are being driven by catch already under the assumed F_s . Under the new forced-catch option, any failure to recreate observed system dynamics will reveal itself in a poor fit to expected catch values, while under the forced- F option, failure to recreate observed dynamics will reveal itself in a poor fit to both biomass and catch time series.

Incorporating the effects of climate

Primary production anomaly

An automated procedure in Ecosim was used to create a forcing pattern to represent primary production anomalies, predicted by the model to minimize discrepancies between model dynamics and biomass time series across all functional groups (EwE production forcing: Walters *et al.* 2004). This assumes that a spike in primary production will cascade up the food web, increasing the average abundance of high order species (Beamish, 1995; McFarlane *et al.*, 2000). A forcing factor was introduced for each year between 1950 and 2000. The sum of squares versus biomass time series (Appendix Table A5.3.2) for all functional groups was reduced 2% by applying the primary production anomaly pattern after initial fitting through diet matrix and vulnerability adjustments. A two-tailed Spearman's rank sums test suggests significant negative correlation with sea surface temperature ($r_s = -0.57$; $\alpha < 0.001$) (Fig. 5.1). Generally, there is less primary production than expected in the second half of the simulation ~1975-2000; this period corresponds to a warming trend, and possibly a regime shift around 1977 (Lluch-Belda *et al.*, 2001; Parrish *et al.* 2000; Hare and Mantua, 2000; Steele, 1998; Francis and Hare, 1994). The apparent negative correlation with temperature then may be more closely connected to changes in the nutrient supply and other factors associated with the regime shift, for example - as may be related to disruptions in the thermocline or upwelling patterns (Wong *et al.*, 2004).

Table 5.1 describes data sources for NE Pacific environmental indices tested in this chapter. Fig. 5.2 demonstrates that the predicted production anomaly pattern also correlates with northern oscillation index (NOI) and the Pacific decadal oscillation (PDO) index (although Guénette, 2005 established no such link to PDO in southeast Alaska).

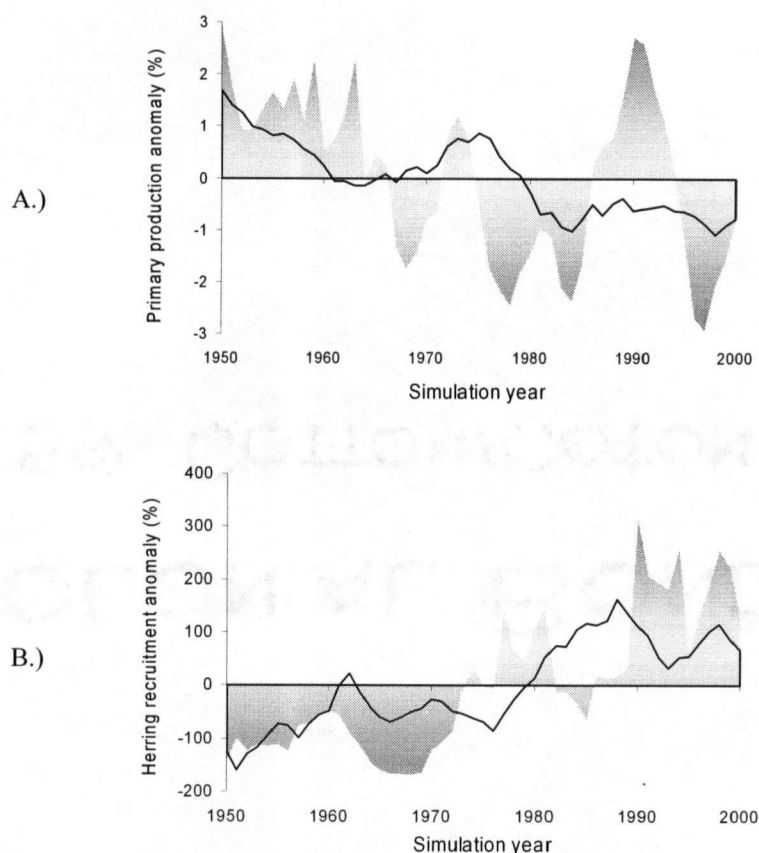


Figure 5.1 EwE's predicted climate anomalies versus their strongest correlating environmental indices. Anomaly (shaded area); environmental index (line). A.) Primary production anomaly versus sea surface temperature (inverted). B.) Herring recruitment anomaly versus PDO. Environmental indices are scaled to minimize discrepancies with predicted anomaly patterns.

Table 5.1 Data sources for NE Pacific environmental indices.

Index	Abbreviation	Reference
Sea surface temperature ¹	SST	DFO (2005a)
Pacific decadal oscillation index	PDO	Mantua <i>et al.</i> (1997)
Upwelling index (Hecate St.)	UI	NOAA (2005) ²
Northern Pacific index	NPI	Trenberth and Hurrell (1994)
Southern oscillation index	SOI	IOS (2005)
Northern oscillation index	NOI	Schwing <i>et al.</i> (2002)

¹ Includes observations from six BC lighthouses (Preikshot, 2005).

² Average values for central and northern BC with smoothing factor (Preikshot, 2005).

Herring recruitment anomaly

As herring are suspected of being a controlling factor in the ecosystem, I have introduced a forcing pattern to account for anomalies in their recruitment, or egg production. Forcing patterns for egg production are used less frequently in EwE than primary production; other examples include Baltic Sea cod (Harvey *et al.*, 2003) and E. Pacific oviparous piscivores (Watters *et al.* 2003). The forcing pattern determined by Ecosim is designed to reduce residuals versus herring biomass in Appendix Table A5.3.2. Fig. 5.2 compares the recruitment anomaly with the strongest correlating environmental index, Pacific Decadal Oscillation (PDO). Fig. 5.3 demonstrates the effect of climate forcing patterns on herring recruitment in the model.

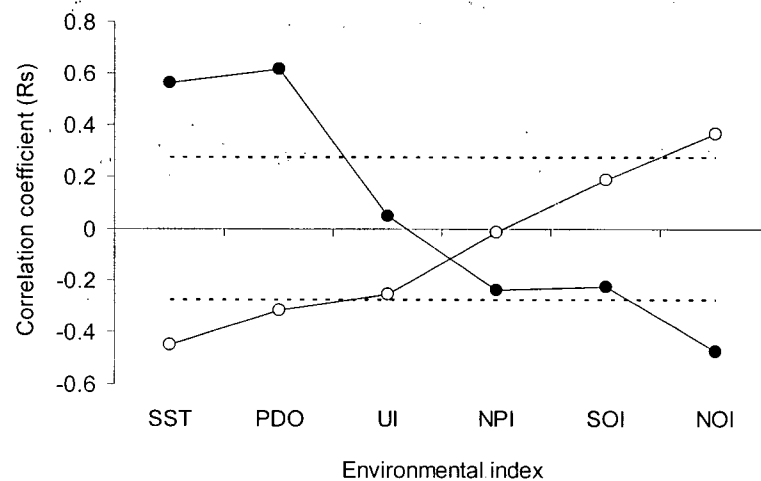


Figure 5.2 Correlation of primary production and herring recruitment anomalies with environmental indices. Open circles show time series correlation of the predicted primary production anomaly versus known environmental indices, closed circles show correlation of the predicted herring recruitment anomaly. Dotted line indicates significant correlations at $\alpha = 0.05$. SST = Sea surface temperature; PDO = Pacific decadal oscillation index; UI = Upwelling index (Hecate St.); NPI = North Pacific index; SOI = Southern oscillation index; NOI = Northern oscillation index.

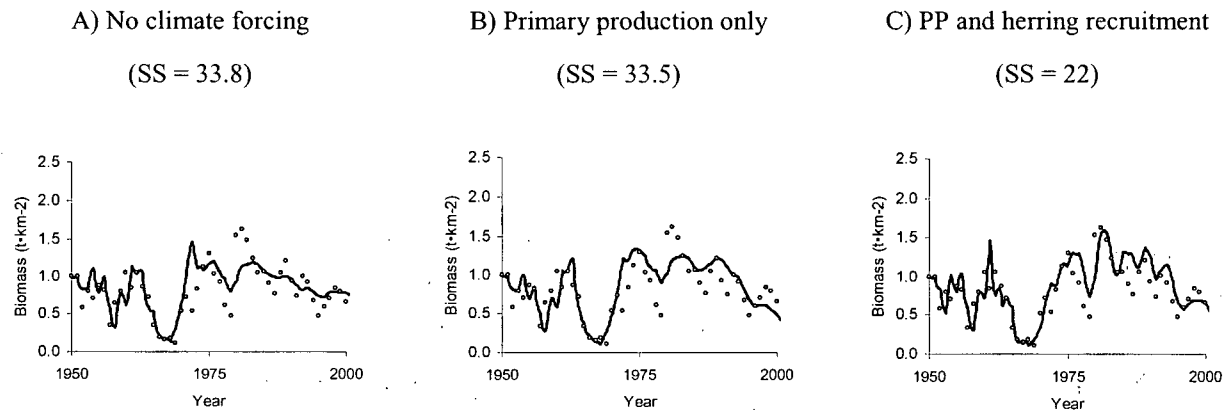


Figure 5.3 Predicted and observed herring trend (1950–2000) under three conditions of climate forcing. Open circles show observed biomass trend from stock assessment, line shows biomass trend predicted by EwE model. All simulations are driven by historic fishing mortalities. A) Predicted biomass dynamics without climate forcing; B) with primary production forcing only; C) with primary production and herring recruitment forcing. Addition of climate forcing patterns reduces sum of squares (SS) between the observed and predicted biomass series.

Although numerous studies have examined the relationship of herring recruitment with environmental proxies e.g., salinity, river discharge, wind transport, upwelling (Beamish *et al.*, 1994; Schweigert, 1995; Williams and Quinn, 2000; Zebdi and Collie, 1995), Williams and Quinn (2000) point out that the only impacting index people tend to agree on is sea surface temperature. The herring recruitment anomaly defined here does correlate strongly with temperature ($r_s = 0.566$; $\alpha < 0.001$) (Fig. 5.2). However, it is a positive correlation, which contradicts Williams and Quinn (2000). They suggested that variations in herring recruitment tend to correlate negatively with SST in BC waters, even though it is a positive correlation elsewhere in the north Pacific.

Matching system variability to data

Initial tests of the model using the primary production and herring recruitment forcing patterns achieved a good fit to data. However, for most functional groups the biomass trajectory predicted by the model was less variable than the time series information, when driven by historical mortalities and climate forcing. This was particularly noticeable for planktonic groups that show a high inter-annual degree of variability. In order to account for stochastic processes

and increase the overall variability of functional group biomass, the forcing pattern for primary production was scaled to match the observed annual variance of phytoplankton abundance (from Gulf of Alaska; Preikshot, 2005). To do this, the initial primary production anomaly was used to generate a time series of predicted biomass for the phytoplankton group between 1950 and 2000. The biomass was scaled and reentered as a direct driver of phytoplankton biomass in the CSV input file (i.e., using data code -1; see Walters *et al.*, 2004).

Scaling phytoplankton variance to data allows Ecosim to predict the correct variability of functional groups biomass within an order of magnitude throughout all levels of the food web (Fig. 5.4). The automated routine in Ecosim used to generate a primary production anomaly pattern cannot be trusted to produce accurate variability in biomass predictions, even for low trophic levels, if the least squares criterion is better satisfied using flat biomass trajectories. For most EwE models there will likely be a cost associated with scaling the variability of biomass predictions; for the 1950

northern BC model, the sum of squares was increased by about 9% in order to accommodate the scaling factor. However, the fit to data was good to begin with, and this procedure should further improve predictions under policy scenarios that estimate extinction risk based on historical climate variability (T.J. Pitcher, *in review*. Conference

proceedings: 2005 ICES Annual Science Conference, Aberdeen, UK). The vulnerability of the

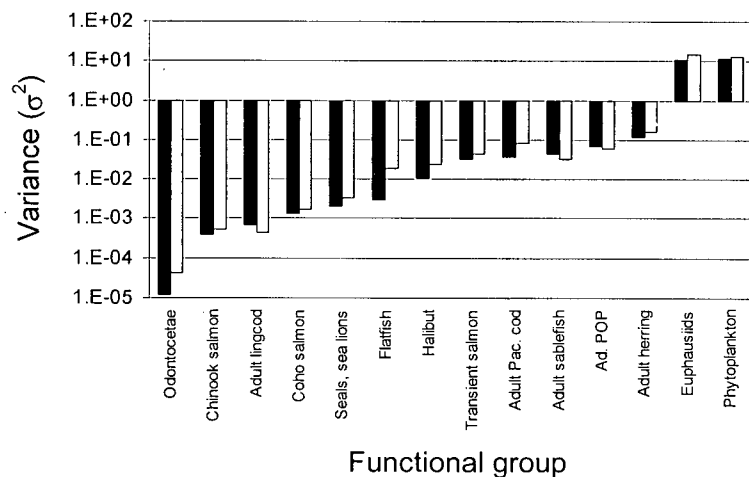


Figure 5.4 Predicted and observed variance of group biomass trajectories (1950-2000). Black bars show annual biomass variance predicted by EwE; white bars show variance observed in stock assessment records. Variability of phytoplankton biomass has been fixed at the observed level and re-entered into Ecosim as a biomass driver. The realistic level of phytoplankton variability is propagated throughout the food web so that predicted variances of other groups are within an order of magnitude of observed trends.

phytoplankton group to euphausiid predation was increased *ad hoc* to help propagate production variability throughout the food web.

5.4 Assembling time series data

Time series for biomass trends, catch and fishing mortality were developed for Northern British Columbia by several previous authors: M. Vasconcellos (*Unpublished manuscript*. Departamento de Oceanografia, Universidade do Rio Grande, Caixa Postal 474, 96201-900, Rio Grande, RS, Brazil), Preikshot (2005) and S. Heymans (*Unpublished manuscript*. UBC Fisheries Centre. Contact: s.heymans@fisheries.ubc.ca). The data time series compiled here utilize some of their sources, add newer information, and include biomass trends based on LEK information (Chapter 3; Ainsworth and Pitcher, 2005a). Also included in the assembled time series are estimates for illegal, unreported and unregulated catch (Chapter 4; Ainsworth and Pitcher, 2005c) for salmon and groundfish functional groups.

Appendix 5.3 summarizes time series data for the period 1900 to 2000. Appendix Tables A5.3.1 and A5.3.2 show 50-year biomass series beginning in 1900 and 1950, respectively, Appendix Tables A5.3.3 and A5.3.4 show catch, and Appendix Tables A5.3.5 and A5.3.6 show fishing mortality. All tables in Appendix 5.3 provide an approximate ranking of data quality. A dark cell colour indicates high quality data and a light cell colour indicates poor quality data. References are listed in Appendix 9.5.3. All estimates of catch and biomass are presented in $t \cdot km^2$, and are calculated assuming the whole shelf area of BC is 113,000 km^2 (D. Preikshot, *unpublished manuscript*. Contact: d.preikshot@fisheries.ubc.ca) and the whole area of northern BC is 70,000 km^2 (Chapter 1).

Local environmental knowledge

For data-poor functional groups, I use time series estimates of relative abundance determined from LEK information in Chapter 3. For certain functional groups, Ainsworth and Pitcher (2005a) discovered discrepancies in the period 1950-2000, between biomass trends suggested by stock assessment and biomass trends suggested by interviews (Chapter 3). Therefore, in order to use the LEK information to validate model dynamics, several key assumptions are made.

Scientific information is always considered preferable to interview information. LEK data is never used for groups where stock assessment is available, and so primarily non-commercial groups are guided by LEK. The relative LEK abundance trends, which are categorized in yearly time steps between 1950 and 2000, are assumed proportional to functional group biomass; and the start and end points are fixed to equal the biomass estimates of the 1950 and 2000 models, respectively. As the LEK data cannot be used to infer an absolute quantity of biomass, this method allows me to verify at least the relative trends. In the absence of better information, I believe the assumption is reasonable, although there is a drawback. For completely data-deficient groups, the assumption made most often in creating the models is that the biomass level has not changed between 1950 and 2000, or has changed very little. Therefore, start and end points for these groups were similar, resulting in a flat line estimate of biomass change for all but the most variable LEK trends. Groups prone to this were forage fish, eulachon, infaunal detritivorous invertebrates and epifaunal invertebrates. Having scaled the information to agree at its endpoints with model estimates, I enter the LEK time series as an absolute trend of abundance in the EwE input file (data type 1), for all Ecosim simulations.

Unfortunately, LEK cannot help validate predicted variability of biomass fluctuations; instead it provides relative estimates that should be viewed as decadal averages of biomass change. I therefore disregarded all LEK trends from the quantitative fitting procedure, but I have included the information for the purpose of comparison in Appendix Fig. A5.4.1. Some of the LEK trends extend back in time beyond 1950, but they are available for only a small number of functional groups and are based on information from only a few respondents. LEK data prior to 1950 was therefore omitted.

Appendix 5.4 shows the model's dynamic fit to data. Appendix Fig. A5.4.1 shows predicted and observed biomass trajectories when driving the 1950 model forward fifty years under historical trends of fishing mortality and climate forcing. Climate forcing factors are presented for primary production and herring recruitment. Biomass error bars reflect the quality of data used in the 2000 models; they show default confidence intervals employed by the data pedigree routine (Christensen and Walters, 2004a). Appendix Fig. A5.4.2 shows predicted and observed catch series (1950-2000).

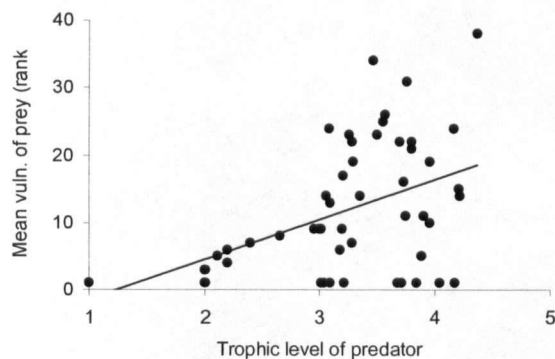
5.5 Analysis of fitted vulnerabilities

I found a positive, significant correlation of the fitted vulnerabilities with predator trophic level ($p < 0.01$; Spearman's rank correlation test), as did Lozano, H. (*in prep.* Ph.D. thesis. Department of Zoology. University of British Columbia) for the northern Gulf of California. This finding (Fig. 5.5a) indicates that top predators like salmon exert the most control over prey populations (e.g., forage fish), while mid-range predators like herring are subject to more bottom up control from their prey (e.g., zooplankton), likely because of climate effects. It is difficult to generalize any further because there is so much variability in the mode of attack and defense.

Although the correlation is weak ($p = 0.16$), average prey vulnerability also tends to increase with prey trophic level (Fig. 5.5b), which supports a similar conclusion. Predators of low trophic level creatures are more often subject to environmental control than are apex predators. This was confirmed also by Ainsworth and Pitcher (2004a). Variability of the fitted vulnerability parameters increases with TL of both prey and predator (by prey: $p < 0.01$; by predator: $p < 0.01$). This is a consequence of the fact that high TL predators feed on a greater variety of species than low TL predators, although the result is also determined by model structure. Fig. 5.6 shows the log-scale relationship between vulnerability parameters and the trophic level of predator and prey.

A.) Rank vulnerability compared with predator TL.

($p < 0.01$)



B.) Rank vulnerability compared with prey TL.

($p = 0.16$)

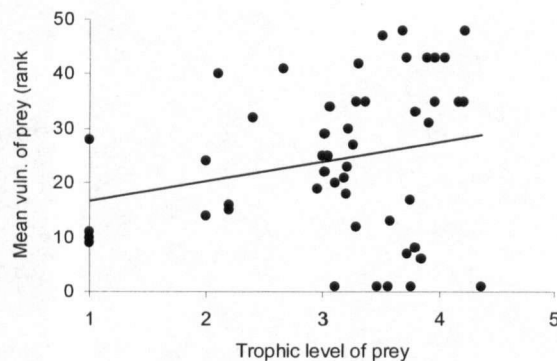


Figure 5.5 Rank order of vulnerabilities in the fitted 1950 model versus predator and prey trophic level. Trophic level of predator (A); prey (B). Fitted vulnerability parameters are positively correlated with trophic level of predator and prey; Spearman's rank correlation test finds the former significant. Trend line is shown.

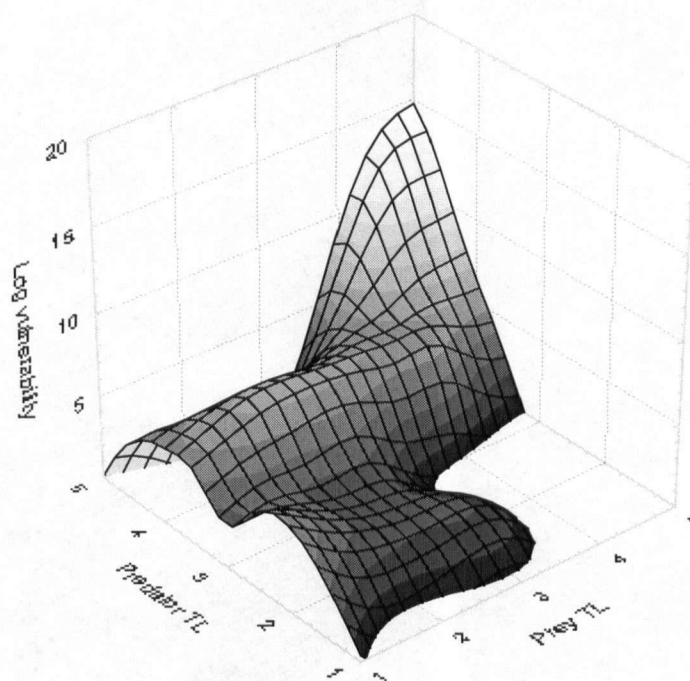


Figure 5.6 Log vulnerabilities in fitted 1950 model versus predator and prey trophic level. Vulnerability is positively correlated with trophic level for both predator and prey. A high vulnerability indicates top-down control of trophic interaction.

Testing vulnerability shortcuts

Until recently, Ecosim has allowed users to search for only up to 15 predator-prey vulnerability interactions. As a result, most authors have assumed that the same vulnerability setting can be applied to each prey sought by a predator, and optimized vulnerability parameters uniformly by columns. Another approach used is to search by rows, so that a particular prey is assumed equally vulnerable to all its predators. Newer versions of Ecosim (beginning V.5.1 Oct, 2004) have allowed users to search for additional vulnerability parameters, including every predator-prey interaction in the model.

It is important to consider the policy impacts of various assumptions on trophic flow. Ainsworth and Pitcher (2004a) examined several assumptions on vulnerabilities, and determined which 'short-cuts' allow the best predictions of ecosystem dynamics. They tested historic models of northern BC (1950 model; Ainsworth *et al.*, 2002), Bay of Biscay (1970 model; Ainsworth *et al.*, 2001), Strait of Georgia (1950 model; Dalsgaard *et al.*, 1998) and the English Channel (1973 model; Stanford, 2002). Ainsworth and Pitcher (2004a) suggested scaling vulnerabilities proportionately to prey trophic level (prey-control), as this provided a better fit to data for three out of four models tested. The exception, northern BC, showed a better fit to data under the assumption of predator-control. However, after revisions made to the 1950 northern BC model for the current volume, results now agree with the other tested systems: scaling vulnerabilities proportionately to prey trophic level provides a better fit to data than scaling by predator trophic level (Fig. 5.7). Results from 1900 northern BC model also confirm that this is a suitable assumption (analysis for both models is based on minimizing residuals versus time series data in Appendix Table A5.3.1).

Results for 1900 and 1950 models in Fig. 5.7 show remarkable consistency, demonstrating that individual parameterization using search methods provides the best fit to data. However, short-cut methods can provide a reasonable representation of system dynamics. Global vulnerabilities provide a good fit to data in both the 1900 and 1950 models, particularly when using low vulnerability settings. However, low vulnerabilities are expected to provide a good fit to data

only in lightly exploited systems like the ones represented by the 1900 and 1950 models (W. Cheung, UBC Fisheries Centre, *pers. comm.*), but when dynamics are changing fast, low vulnerabilities will under estimate ecosystem response (Martell *et al.*, 2002).

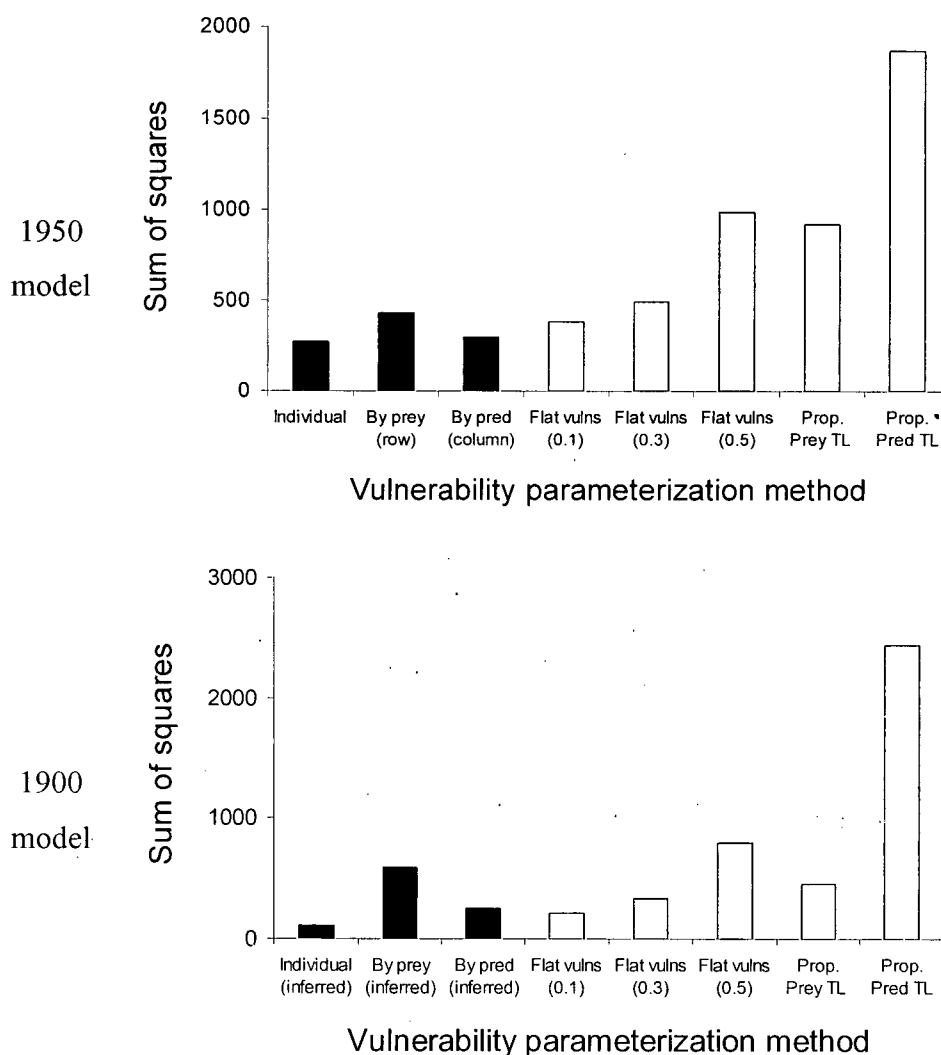


Figure 5.7 Evaluation of short-cut methods used to parameterize Ecosim vulnerabilities. Common parameterization methods are compared in their ability to recreate observed time series (1950 model and 1900 model; 50 year simulations). A low sum of squares indicates the model's predictions agree closely with observed biomass trends. Black bars use customized vulnerabilities fitted with a non-linear search routine; white bars show generic parameterizations using short-cut methods. Fitted vulnerabilities outperform short-cut methods; short-cut methods that employ low global vulnerabilities create conservative dynamics, and can outperform vulnerability values assigned proportionately to trophic level (TL), at least for un-degraded ecosystems.

5.6 Validation of dynamic function

Equilibrium analysis

As a test of ecosystem predictions of the 2000 model, the analysis presented in Appendix Fig. A5.5.1 reduces EwE to a single species model. Increasing fishing mortality stepwise from zero to several times the baseline value, the automated equilibrium routine in Ecosim calculates the equilibrium biomass established for the subject functional group under that level of fishing mortality (Equilibrium routine: Christensen *et al.* 2004). For this example, the biomass of other functional groups is held constant to remove confounding effects from trophic interactions. At their left-most extent, the biomass equilibrium curves tell us what biomass level the group assumes under zero fishing mortality (B_0). The catch equilibrium curves are essentially single-species surplus production curves; the maximum height of the curve shows the maximum sustainable yield (MSY) of the stock and the fishing mortality at which that occurs, the F_{MSY} . The dotted vertical line shows the baseline (current) level of fishing mortality. In a properly parameterized model, the baseline fishing mortality of underexploited groups should generally fall to the left of F_{MSY} , and to the right for overexploited stocks. How a functional group behaves under dynamic simulation will be greatly influenced by the initial relative level of exploitation represented in the basic Ecopath model.

Recreating the present-day system from 1950 model

Driving the 1950 model forward 50 years should produce a new ecosystem structure that is very similar to the 2000 model, when dynamics are driven by historical fishing mortalities and climate forcing. Dynamic predictions of the 1950 model under these conditions are shown in Appendix 5.4. For most functional groups, the end-state biomass (in year 2000) falls within the confidence interval suggested by the Ecopath pedigree ranking of data quality (Walters *et al.*, 2004) for the 2000 model. Confidence intervals associated with each degree of data quality are based on the default settings used by Ecosim's Monte Carlo routine.

Appendix Table A5.6.1 compares the end-state of the 1950 model simulation (called the 'derived' 2000 model), versus the 2000 model that is based on current scientific data (called the 'proper' 2000 model). Biomass trends for seven functional groups out of 51 fall outside of the confidence intervals set for the year 2000 data points. The biomass of flatfish, herring and Pacific cod (juv/adult) is overestimated in the dynamics, and the biomass of seals and sea lions and halibut (juv/adult) falls short of observed levels. The time series fit to data is actually quite good for flatfish, halibut, herring and Pacific cod (see Appendix Fig. A5.4.1), but the confidence intervals used are extremely narrow due to high data quality. Only one group, seals and sea lions, is of concern. The predicted biomass trend could not be made to recreate observed dynamics despite all efforts to tune the model. Preikshot (2005) encountered a similar result using the same data series for observed biomass in BC, which suggests this may be data problem. The data represents an average of biomass change throughout the whole BC coast (harbour seals: Olesiuk, 1999; Sea lions: Bigg, 1985), so one likely explanation is that the population increase in northern BC has been more modest than in southern BC (*vis.* Strait of Georgia: Olesiuk, 1999). Otherwise, there may still be population dynamics concerning pinnepeds that are poorly represented. For a discussion regarding the difficulties of modeling marine mammal populations in Ecosim, see Gu  nette (2005).

Dynamic tests of the derived 2000 model

A new 2000 Ecopath model was created based on the end-state of the 1950 dynamic simulation, using the EII export/import procedure described in Walters *et al.* (2004). The derived 2000 model was subjected to tests to determine dynamic responses of the ecosystem, and to compare responses with predictions made using the proper 2000 model. Figs. 5.8 shows the equilibrium ecosystem condition in 2050 predicted by both versions of the 2000 model, after shutting off fishing for 50 years. The long simulation length allows the system to reach equilibrium. The behaviour of functional groups is largely consistent between the two models except for infaunal invertebrates. Figs. 5.9 and 5.10 compare the resulting ecosystem structure for commercial functional groups in both the derived and proper 2000 models, after 50 years without fishing. Predictions are similar between the models in relative change of functional groups.

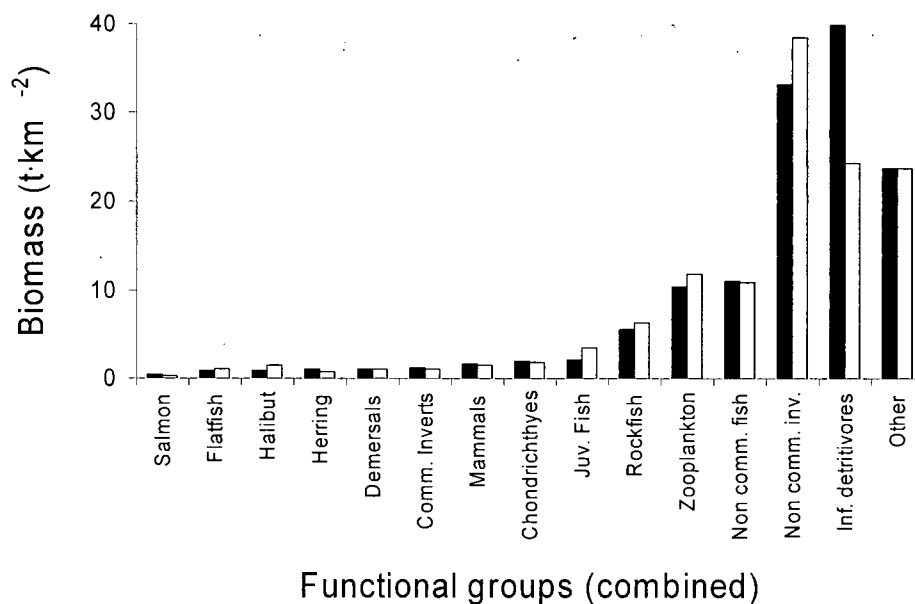


Figure 5.8 Group biomass predicted in 2050 by derived and proper 2000 models after fishing release. Black bars are based on the derived 2000 model (end state of a 50-year forward simulation using the 1950 model); white bars are based on the proper 2000 model (constructed using current scientific data). X-axis combines functional groups.

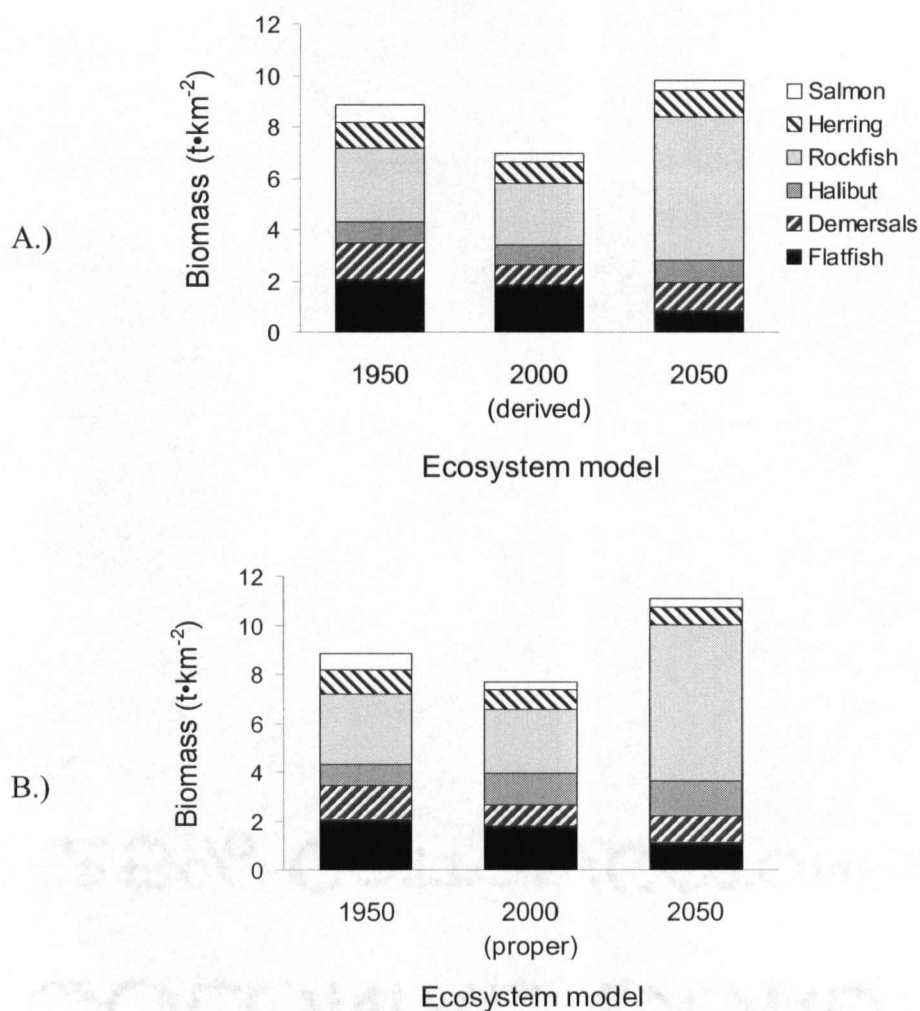


Figure 5.9 Biomass change predicted by the derived and proper 2000 models after fishing release. Biomass of commercial groups shown. (A) Biomass predictions of the derived 2000 model (end state of a 50-year forward simulation using the 1950 model). (B) Biomass predictions of the proper 2000 model (constructed using current scientific data). 1950 model is provided for comparison. The proper and derived 2000 models predict a similar ecosystem response when fishing pressure is removed.

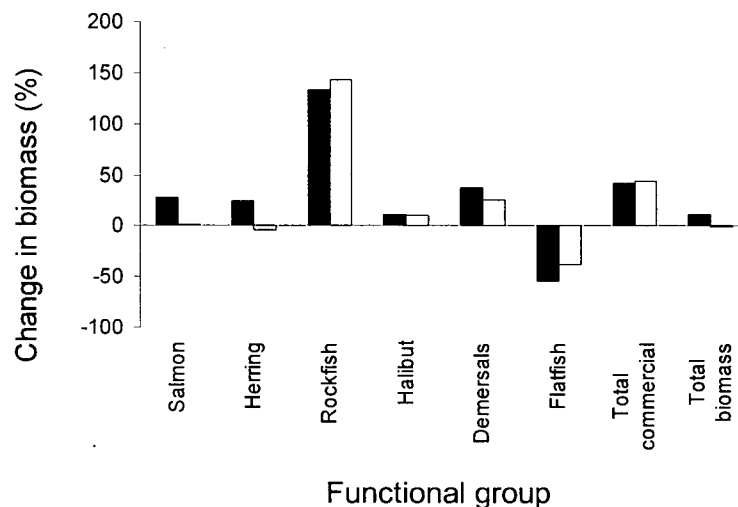


Figure 5.10 Direction of biomass change predicted by proper and derived 2000 models after fishing release. Black bars show group biomass predictions in 2050 after 50 years of fishing release based on the derived 2000 model; white bars show biomass predictions of the proper 2000 model.

5.7 Discussion

EwE models

This chapter introduced the revised northern BC Ecopath and Ecosim models, and established that dynamics of the 1950 model are functioning well enough to predict observed trends in the ecosystem over the last 50 years, providing one plausible explanation for observations. Predicting the abundances of commercial groups is relatively easy, since a large part of their mortality is caused by the direct impacts of fisheries. The observed mortality rate can be approached closely using only catch or fishing mortality drivers. Predicting abundances in non-commercial groups presents more of a challenge since their population dynamics are determined by the sum action of many semi-independent mortality sources and generally, there is no single overwhelming influence. The modeler must determine a combination of diets and mortalities that permits the observed dynamics, while staying within the realm of what is ecologically reasonable. Although data on non-commercial groups is scarce, the model shows agreement with proxy abundance information provided by LEK interviews.

There are many ways to recreate observed dynamics to any given level of precision, and so there are unavoidably subjective parts to the modeling process. Chapter 8 will address this. Nevertheless, it is an achievable goal to represent broad ecosystem responses, and this chapter offers several hints that real ecological processes are being reflected in the population dynamics.

The models for northern BC await their next revision. Thanks to the efforts of previous authors, and the contributions in this volume, the 1950 and 2000 models especially represent useful potential tools for Canadian fishery policy analysis. Only one application of the models is presented in this report, under the auspice of the BTF project - the development of whole ecosystem restoration strategies. However, the flexibility of EwE encourages diverse uses, and the northern BC models are unique in that so many individual authors have contribute to them. I invite future researchers to carry on the lineage.

Climate factors

The primary production and herring recruitment anomalies both correlate with environmental time series in a consistent way. Both series concur that SST, PDO and NOI are closely linked to system productivity. However, the reciprocal relationship in Fig. 5.2 seems unlikely to have emerged by chance. It may be only an artifact of the modeling. Primary production forcing was added first, and so the herring recruitment trend could be correcting for damages done to the herring fit. However, Fig. 5.3 indicates that is not the case, the sum of squares was reduced in herring by primary production forcing. At any rate, the recruitment anomaly has an overwhelming effect on herring productivity, so it cannot simply be reacting to the primary production trend. Another possibility is that the series are revealing actual ecology. An apparent shift in productivity is predicted by both series to have occurred around the mid 1970s. The hypothesized regime shift in that period (Ebbesmeyer *et al.* 1991) lends credence to the suggestion that the anomaly patterns are meaningful, and have independently arrived at the same conclusions regarding mesoscale climate variation. In this case, the primary production anomaly reads evidence of climate variation hidden in system-scale dynamics, while the herring recruitment trend sees the effect of the same climate factors on herring alone. Although, the

correlation could break down at longer timescales if the environmental proxies themselves only indirectly relate to the real causes of environmental change (Holm *et al.*, 2001).

Fitting procedure

The fitted vulnerabilities may also reveal real ecosystem dynamics. There is a correlation with trophic level: low trophic level dynamics are dominated by bottom-up controls, while high trophic level dynamics also include top-down interactions. This indicates a wasp-waist ecosystem (Cury *et al.*, 2000). Strong predator control does not cascade down to phytoplankton, possibly due to compensatory mechanisms operating in the ecosystem and revealed in the time series data (Vander Zanden *et al.*, 2005; McQueen *et al.*, 1989). Certainly, this can be one hypothesis to explain the observed dynamics, but it is difficult to rule out the possibility that the relationship is spurious or only a modeling effect. Since trophic dynamics of ecosystems are not well understood, and since any result can be particular to the ecosystem in question, there is currently no way to validate the result. Ongoing attempts to fit historical models to data in other ecosystems may demonstrate a link to trophic level or some other measurable quantity, and ground the vulnerability parameter more firmly to empirical evidence. Considering the results in Fig 5.5 are noisy, and considering prior EwE investigations have sometimes failed to detect a relationship with trophic level, it may take much work before we can draw generalizations between trophic flow dynamics and the inherent oceanographic or biological properties of ecosystems.

Only the 1900 and 1950 models can be fitted to data. Precise biomass dynamics in pre-contact times can never be known, and we will have to wait some time before the 2000 model can be validated with data. Since the current ecosystem will be the starting point for any future restoration policies, it is critical that the dynamics are well represented in the 2000 model. The key assumption I have made, that the fitted vulnerabilities in the 1950 model can be applied to other time periods, must be supported if predictions of the 2000 model are to be trusted. Of course, an assumption must be made in any case regarding vulnerabilities for a present-day model that cannot be fitted to data. However, Fig. 5.7 demonstrates that transferring vulnerabilities to other time periods from a fitted model is a sound approach, and is better than

other typical short-cuts used to parameterize present-day models. Although the method assumes stationarity in the vulnerability parameters, the fact that the 1900 model performs well under the 1950 vulnerability scheme indicates that the modelled vulnerabilities are appropriate for the system and that the true vulnerabilities governing trophic interactions have not changed much in the last century. The decision was made to use the 'proper' 2000 model for policy explorations (Chapter 7) rather than the 'derived' 2000 model to avoid any errors accumulated in forward simulations of the 1950 model. The proper 2000 model represents the best guess of current ecosystem structure; and with the vulnerability matrix based on fitted parameters, the dynamics should perform adequately.

Having designed and tested suitable models for 1750, 1900, 1950 and 2000, the next chapter will introduce a new conceptual goal for ecosystem restoration based on the historic systems. It will evaluate the potential harvest benefits from the restored historic states, and make trade-offs explicit that are inherent in the choice of restoration goal.

6 EVALUATING RESTORATION GOALS

A goal is not always meant to be reached, it often serves simply as something to aim at.

Bruce Lee

Tao of Jeet Kune Do (1975)

6.1 Introduction

In this chapter, I design whole-ecosystem restoration goals for northern BC. The goals are based on descriptions of four historic ecosystems developed in Chapter 5; 1750, 1900, 1950 and 2000 AD, but the goals represent modified forms of these systems. Using patterns of fishing mortality determined by a policy optimization routine, the historic ecosystems are fished under dynamic simulation. This restructures them over time into more productive forms, tailored to deliver harvest benefits specified by imaginary resource users. The exercise is analogous to the changes caused by the uncontrolled experiment witnessed in the real-world history of exploitation. Except, the fishing rates used to harvest the ecosystem are optimal for the specified goal, the fishing fleet is designed to minimize environmental impact, and the selective pattern of exploitation leaves the ecosystem not degraded and devalued, but in a healthy and maximally productive state. I put forward these idealized ecosystems as potential goals for restoration.

Candidate restoration goals based on the four historical periods are compared, and a variety of optimization objective functions are used to reveal the full range of sustainable (i.e., equilibrium-level) benefits available from each historic system. Five optimization objective functions are tested that, together, span the spectrum of human use versus conservation. A continuum of possible restoration targets is identified from each historic system. At one extreme, the historic ecosystem is manipulated to support maximum catch rates or fishery's economic value. At the other, biodiversity or ecosystem maturity is preserved or augmented using a harvest agenda orientated towards conservation. Between these extremes, potential harvest benefits trade-off

between socioeconomic and ecological returns. The trade-off range inherent to each period is made explicit using existing ecosystem indicators and new ones developed in Chapter 2.

Comparing sustainable benefits predicted from several historic ecosystems allows us to estimate the relative worth of these periods in both monetary and non-monetary terms. This serves two purposes. First, it demonstrates what potential harvest benefits have been lost through shortsighted fishing practices in northern BC. It is an admonishment, but it should also help combat the shifting baseline syndrome (Pauly, 1995) in a site-specific and historical context. Second, evaluating fishery benefits of these periods also demonstrates what a restored ecosystem may be worth to stakeholders. Knowing this could help us justify the initial economic and social costs of ecosystem restoration.

The expected economic, social and ecological benefits from each historical period are quantified in this chapter under a range of policy objectives, and a few candidate goals for ecosystem restoration are considered in detail: Chapter 7 will look at ways of approaching these restoration goals through restorative fishing policies. The precise choice of ecosystem structure that we might adopt for a restoration goal is left open to policy makers, to the public, and to the reader - but characterization of benefits here, and further cost-benefit analysis of restoration strategies in Chapter 7 should help to inform the choice. This chapter builds on the work of Ainsworth and Pitcher (2005b) and Pitcher *et al.* (2005), but all models have been improved and further validated against data since those preliminary efforts.

Optimal restorable biomass

Any amount of fishing will disturb the biomass configuration of a pristine ecosystem. Pursuing a pristine historic state as a restoration goal, such as the pre-European contact period in northern BC, could only be done at a huge expense to industrial fishing. Moreover, single species science has long understood that the most productive state of a stock is not in its unfished condition, but when older, less productive individuals have been removed from the population. No rational restoration policy would therefore seek to restore the pristine state, just to allow fisheries to reduce stock populations to more productive levels. Instead, we should restore that maximally

productive state directly. In other words, the goal for ecosystem restoration should be the biomass equilibrium that naturally results from the historic ecosystem, once subjected to a responsible fishing regime and optimized in biomass structure to deliver maximum equilibrium-level benefits. Fig. 6.1 illustrates this conceptual goal for restoration, which is called the optimal restorable biomass, or ORB.

The ORB ecosystem configuration I advocate as a restoration goal is a theoretical one. It does not represent a particular period in the real-world evolution of the northern BC ecosystem. It represents how the ecosystem might have looked today if past generations had preserved and cultivated the productive potential. However, there is no ideal solution. The specific ORB configuration that we prefer to restore will depend on what harvest benefits we wish to maximize.

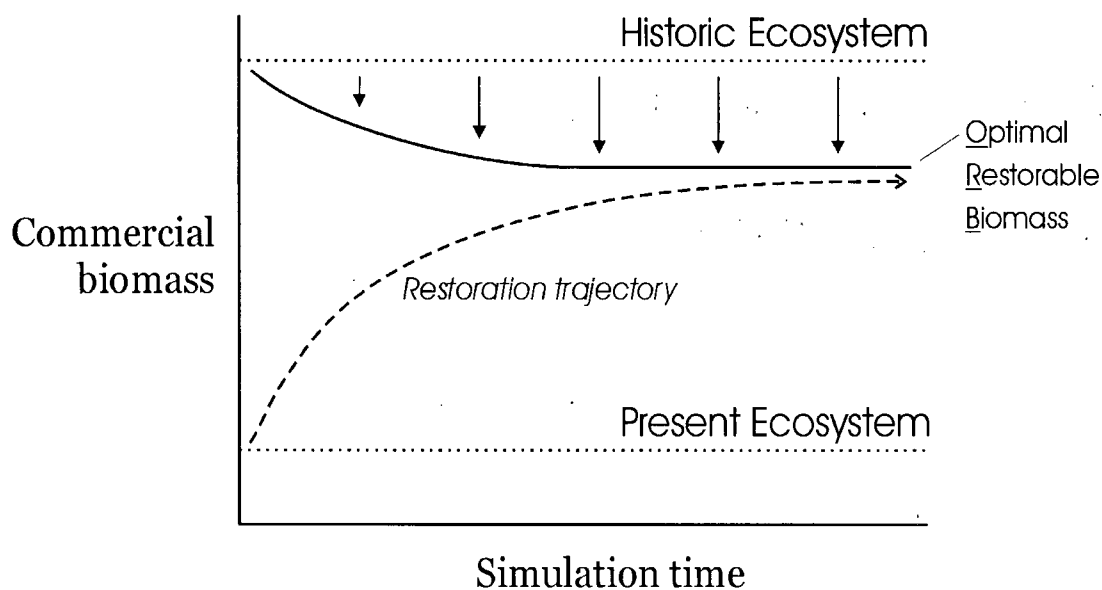


Figure 6.1 Optimal Restorable Biomass (ORB) concept. ORB is the theoretical ecosystem biomass equilibrium that would result after long-term optimal harvesting of the historic ecosystem (downward arrows). A possible restoration trajectory is shown (broken line) that would see the present ecosystem changed to resemble the ORB state. Simultaneity is not implied between the ORB determination, which is theoretical, and restoration, which is practical.

Where stocks interact through predation or competition, it may be impossible to achieve the biomass that permits the single species maximum sustainable yield (B_{MSY}) simultaneously for multiple stocks (Larkin, 1977; 1996; Walters *et al.* 2005). From a whole-ecosystem perspective, it becomes necessary to choose between stocks, holding the biomass of some close to their optimal levels while sacrificing the productivity of others. According to our choice, total catch from the ecosystem can be maximized; the desired modality may be profit, or biodiversity, or any other practical measure of socioeconomic or ecological utility. If our management goal is simply to maximize catch, for example, B_{MSY} should be sought only for the most productive and massive stocks, while maximum productivity of low-volume fisheries may need to be sacrificed. Similarly, a plan to maximize total profit would see biomass of the most profitable stocks held close to the level permitting maximum economic yield (B_{MEY}), while the biomass of less profitable stocks may need to lie further away from their B_{MEY} . An alternative goal may maximize system biodiversity, in which case the biomass of ecosystem components would be altered from the historic state to maximize species evenness or richness as required. In most cases, a practical policy goal for restoration will contain a balance between socioeconomic and ecological priorities.

By use of ecosystem models, we can calculate the specific biomass configuration that will yield maximum harvest benefits, though the optimal design may be constrained by additional caveats (e.g., a minimum species biomass threshold). If we structure the ecosystem to deliver maximum catch, then ORB becomes a whole-ecosystem analogy to B_{MSY} ; if we structure the ecosystem to provide maximum profit, then ORB becomes an analogy to B_{MEY} . ORB calculation based on historic systems therefore satisfies two requirements: it increases the production rate or harvest utility of key groups by changing their equilibrium biomass level and the biomass of supporting/detracting groups, and it trades off optimality between groups in order to provide maximum net benefit from the ecosystem as a whole.

Response surface analysis

In this work, the ORB system configuration is calculated by the policy search routine in Ecosim (Christensen and Walters, 2004b) through iterative harvest simulations of the historic models.

An optimal fleet-effort solution is determined (i.e., a set of equilibrium fishing mortalities per gear type) that will manipulate the historic ecosystem into a maximally beneficial form. The policy search routine probes an n -dimensional parameter space (a response surface) for zones that yield increased fishery benefits, where n is equal to the number of gear types or fishery sectors in the model. However, the response surface may be non-linear, and the starting point of the optimization procedure can have a large impact on the recommended solution. If the initial fleet-effort configuration used by the policy search routine is far from the optimal solution, then a sub-optimal peak on the response surface may be identified by the search algorithm (Ainsworth and Pitcher, 2005b). Note that this problem is exacerbated in models that apply high trophic flow vulnerabilities (Cheung *et al.*, 2002). In these models, the response surface is typically convoluted and this leads to inconsistent optimization results that are highly dependant on the initializing parameters. Because of this danger, many optimizations are conducted in this chapter for each candidate restoration goal. Each optimization begins from a random location on the response surface (i.e., the fishing mortality exerted by each gear type is set randomly in year zero) so we can be sure to have located the best peak on the response surface. The macrostructure (geometry) of the response surface is also revealed by repeated optimizations, and policy implications can be considered in addition to the rank benefit of the harvest plan.

Often, optimal policies will cluster around the same peak indicating one indisputable fleet-effort configuration that maximizes fishery benefit over the simulation time horizon. If a recommended harvest policy resides on a narrow peak (i.e., a global maximum), then any variation from the optimal fishing pattern may result in sub-optimal harvests. However, if multiple local maxima are present then random-F initializations will indicate two or more discrete, tight clusters of solutions. For example, multiple policy avenues may exist if the structure of the ecosystem can be manipulated to permit great landings on target group A or B, but not both. If these target groups are associated with a contradictory set of supporters (e.g., prey) and detractors (e.g., predators, competitors), then the policy search must decide between mutually incompatible solutions that yield comparable benefits.

If the identified maximum resides on a broad peak (i.e., a plateau), then the solutions will form a continuous loose cluster where minor variations in the fleet-effort configuration will not

significantly affect the net harvest benefits received. In this case, the precise structure of the ORB ecosystem is less important; near maximum benefits can be obtained even without precise application of fishing effort. A restoration policy seeking this goal will be forgiving of management and implementation errors. Response surface geometries are illustrated in this chapter for candidate restoration goals based on objective criteria.

Addressing parameter uncertainty

Ecosystem models always have data deficiencies, and the problem is amplified when we try to quantify historic systems. Models of the distant past must rely on anecdotal information and use unconventional data sources (Chapter 3; Ainsworth and Pitcher, 2005a). EwE has several capacities to deal with data limitation (Ainsworth and Pitcher, 2005a); nevertheless, data points used to construct historic models typically have wide confidence intervals. Unfortunately, multispecies models can be sensitive to initializing parameters (Hollowed *et al.* 2000; Fulton *et al.* 2003), and even broad policy outcomes in EwE are highly dependant on the underlying Ecopath model. Uncertainty surrounding input data can carry major implications for our harvest policy recommendations, and so in this chapter I predict a range of possible outcomes from harvest simulations of the historic ecosystems using an automated Monte Carlo procedure in Ecosim.

Biomass and production rates contained in the baseline model are randomly varied under a static fishing plan to address the economic and ecological consequences of the optimal fishing vector in the face of parameter uncertainty. From this, we can evaluate the optimality of the ORB ecosystem configuration under different assumptions of historical ecosystem structure, and we can set confidence limits on the estimates of harvest benefits from the optimized systems. Model specification uncertainty is not covered by this procedure.

The *lost valley* fleet

It may be unwise to restore the marine ecosystem through a long and costly process of conservation and rehabilitation, only to unleash the current fleet upon it (Pitcher *et al.*, 2004).

Fishing has been linked to some troubling changes in the ecology of marine systems (Pauly *et al.*, 1998; Jackson *et al.*, 2001; Myers and Worm, 2003), and many of the criticisms of fisheries are related to issues of overcapacity (Mace, 1997; Gréboval and Munro, 1999; Ward *et al.*, 2001). Pitcher (2001a) realized that overcapacity would work against the aims of ecosystem restoration. The issue of fleet overcapacity is assumed to be resolved under the current methodology, since optimal fishing effort will be decided by the policy search routine in calculating ORB ecosystems. Also, I have assumed completely malleable fishing capital, in that there is no cost or penalty associated with reducing or increasing fishing effort in any gear type⁷.

Some suggest that the fishing methods themselves may be partly to blame for current troubles. That is, the gear types and fishing techniques currently in use may be the product of a 'bad evolution' of management decisions that placed politics ahead of efficiency or stewardship (Haggan *et al. in press*). Pitcher *et al.* (2001; 2004) designed a responsible fishing fleet that could be used to sustainably fish a restored historic system, and preserve much of the health and biodiversity while providing a reasonable source of jobs and income to resource users. In their interpretation, the pristine ecosystem represents a blank slate; a newly discovered '*lost valley*' awaiting responsible, sustainable use by humans. Those authors envisaged a fleet designed to minimize collateral damage to the ecosystem. A similar *lost valley* fleet was created for northern BC based on community input in Pitcher *et al.* (2002b). The *lost valley* fleet is used in this chapter to harvest the historic ecosystems and to determine various ORB ecosystem configurations. The hypothetical fleet structure is not based on past or present fisheries. It is designed according to responsible criteria (Table 6.1); discards and habitat damage are minimized within achievable technological limits. The criteria also satisfy the FAO Code of Conduct for Responsible Fisheries (FAO, 1995).

⁷ Modifications are currently being made to EwE optimization source code to allow consideration of fleet buyback schemes (W. Cheung, *pers. comm.* UBC Fisheries Centre).

Table 6.1 List of nine criteria for sustainable and responsible ‘lost valley’ fisheries. Reproduced from Pitcher *et al.* (2004).

	Criteria for sustainable fisheries	Notes
1	Minimal by-catch and discards	Technological modifications to gear
2	No damage to habitat by gear	Technological modifications to gear
3	Includes aboriginal fisheries	Customary rights recognized
4	Includes traditional target species	Except where #1 and #2 would bar
5	Minimizes risk to charismatic species	Except as under #3 and #7
6	Excludes fisheries on juveniles	Except where minimal impact is proven
7	Participatory vetting of fisheries	By management agency, local community and public
8	Simulations show fishery sustainable	100-year simulations are satisfactory
9	Adaptive management plan in place	Adaptive changes to the unexpected (e.g., climate change)

The procedure introduced in this chapter uses the fishing fleet as a tool to modify the historic ecosystems, so the gear structure affects the optimal ecosystem design. The degree of precision to which the fleet can simultaneously affect the biomass of functional groups will put strict limits on the sustainable harvest benefits delivered by ORB ecosystems. The issue of fleet performance in manipulating the ecosystem will be revisited in Chapter 7, when we develop fishing strategies to achieve restoration and compare the effectiveness of the *lost valley* fleet with other fleet structures.

6.2 Methods

EwE models

Ecopath and Ecosim models of northern BC for four historical time periods are described in Chapter 5 (1750, 1900, 1950 and 2000). The only modification made for Chapter 6 was to remove the historical fishing fleets characteristic to each real-world period, and replace them with the hypothetical *lost valley* fishing fleet.

None of the historical models were assumed in Chapter 5 to be steady-state. Instead, each model was parameterized so that the instantaneous biomass flux of every functional group matched reality, or the best guess *in lieu* of an estimate. The biomass accumulation rates used in the historic Ecopath models are indeed representative of real-world dynamics during the baseline year, but they are only appropriate to use in the model if the historical rates of fishing mortalities are also in place. If we remove sources of mortality from the baseline model caused by fishing, and we do not permit a corresponding increase in the biomass accumulation rate, then we have disturbed the initial balance of production and mortality in the model. This will lead us to overestimate biological production rates. Conversely, if fishing mortalities are increased by the fleet substitution, a corresponding reduction in the biomass accumulation rates is required to preserve the initial production-mortality ratio.

Therefore, in substituting the *lost valley* fleet for the historic fleet I did not preserve the historic biomass accumulation rates, and I did not adjust the diet matrix to rebalance the models. Instead, I carried over the EEs of the historic models into the *lost valley* versions, and adjusted the biomass accumulation rates to preserve mass-balance. The baseline *lost valley* models therefore do not represent any real-world condition. However, model dynamics should faithfully recreate actual ecosystem performance under the imposed fleet structure.

The *lost valley* fleet

The *lost valley* fleet for northern BC was designed by Ainsworth *et al.* (2004), and used in policy explorations by Ainsworth and Pitcher (2005b) and Pitcher *et al.* (2005). The same fleet design

is applied here. It includes groundfish trawl, shrimp trawl, shrimp trap, herring seine, halibut longline, salmon freezer troll, salmon wheel, live rockfish, crab trap, clam dredge, aboriginal and recreational fisheries. Directed catch and retained bycatch is shown in Table 6.2 (reproduced from Ainsworth *et al.*, 2004); retained bycatch occurs in all fleets except salmon wheel, live rockfish, clam dredge and aboriginal fisheries. Discards were assumed minimal, only groundfish trawl, shrimp trawl, salmon freezer troll and clam dredge produce discards (Table 6.3; reproduced from Ainsworth *et al.*, 2004). Directed fisheries were set to catch 2.5% of the total biomass of their target groups annually under baseline conditions, and 0.5% or 0.25% of retained bycatch groups. Major discards were set at 1.25% of group biomass, while minor discards were set to 0.25% or 0.025% of group biomass. The catch / bycatch ratios chosen for this hypothetical fleet represent what may be an achievable reduction in non-targeted interceptions based on expert opinion (T. Pitcher, UBC Fisheries Centre. *Pers. comm.*). Catches and discards vary between time periods in proportion to functional group biomass. The fishing policy optimizations used in this chapter are free to vary catch in year zero⁸, so absolute catches represented under baseline conditions are not important. Only the relative proportions of directed catch, bycatch and discards are important.

The northern BC models do not consider the problems of trawl damage, ghost fishing, or any other deleterious (non-trophic) gear effect. In simulations, there is no ecological or economic benefit associated with preserving habitat, and nothing is to be gained by restricting damaging fisheries, except perhaps a coincidental reduction in discards. Similarly, ecologically responsible fleets that omit damaging gear types will not be credited with their full ecological benefit in the simulations. I will address the prospect of modeling non-trophic gear effects in the discussion.

⁸ This represents a special modification to the policy search routine made by this author. The default setting in EwE uses baseline F_s in the first year of the simulation as a numerical convenience for calculating the proportional increase in harvest benefits. However, this would not be appropriate for the present application because the initial exploitation rate of the *lost valley* fleet does not relate to any real-world condition, and so a relative increase over the baseline value is meaningless. Modified EwE executable file is available from author (c.ainsworth@fisheries.ubc.ca).

Table 6.2 *Lost valley* fleet catch. Values in percentage (%) of target group biomass per year. Directed catch under model baseline conditions represent 2.5% of target species biomass; retained bycatch is 0.5% or 0.25%. The ratios between catch and retained bycatch used for this hypothetical fleet are meant to represent a realistic reduction in non-targeted interceptions that is achievable through modifications to fishing gear and techniques.

Target group	Gear type											
	Groundfish trawl	Shrimp trawl	Shrimp trap	Herring seine	Halibut longline	Salmon freezer troll	Salmon wheel	Rockfish live	Crab trap	Clam dredge	Aboriginal	Recreational
Transient salmon						2.5	2.5				2.5	
Coho salmon						2.5					2.5	2.5
Chinook salmon						2.5					2.5	2.5
Ratfish	0.25	0.25										
Dogfish	0.25	0.25				0.25						
Pollock	0.25											
Eulachon		2.5									2.5	
J. herring				2.5								
A. herring				2.5								
A. POP	2.5											
Inshore rockfish	2.5				0.25	0.25		2.5				0.25
A. pisc. rockfish	2.5					0.25						0.25
A. plank. rockfish	2.5					0.25						
J. turbot					0.25							
A. turbot	0.25	0.25			2.5							
J. flatfish					0.25							
A. flatfish	2.5	0.5			0.25							
J. halibut					2.5							0.25
A. halibut					2.5						2.5	0.25
A. Pacific cod	2.5				0.25							
A. sablefish	0.25				0.25							
A. lingcod	0.25				0.25			2.5				2.5
S. benthic fish		0.25	0.25	0.25								
Skates	0.25	0.25			2.5							
Large crabs	0.25								2.5			
Small crabs									0.25			
Comm. shrimp		2.5	2.5									
Epifaunal inv.										2.5		

Table 6.3 *Lost valley* fleet discards. Values are in percentage (%) of target group biomass per year. The discard rates employed are meant to represent a reduction over current levels that is realistically achievable through modifications to fishing gear and techniques.

Group Name	Gear type			
	Groundfish trawl	Shrimp trawl	Salmon freezer troll	Clam dredge
Seabirds	-	-	0.025	-
Small crabs	1.25	1.25	-	0.25
Epifaunal inv.	1.25	1.25	-	0.25
Inf. carn. inv.	1.25	1.25	-	0.25
Inf. det. inv.	1.25	1.25	-	0.25
Corals and sponges	1.25	1.25	-	0.25

Market prices for the *lost valley* fleet are improved over the northern BC historical estimates for the 2000 model (see section 5.2: Fisheries). The prices used are important because they determine which species and gear types are favoured in economic and social optimizations. The *lost valley* prices were assembled by Buchary and Sumaila (2002) based on DFO records for landed value available online [www.dfo-mpo.gc.ca/communic/statistics/commercial/landings/sum_e.htm]. The *lost valley* fleet prices are similar overall to the historical estimates for 2000. The only major modification to the historical price estimates was made for the recreational sector; salmon, halibut, lingcod and rockfish were assumed to be worth 20 times more when landed by the recreational fleet than the commercial fleet. The prices are meant to reflect additional revenues enjoyed by supporting industries, such as hotels and fishing charter companies. A proper calculation of recreational value would require study across multiple service industries. This approximation therefore represents a speculative estimate, however since the *lost valley* fleet is hypothetical, I am assuming in effect that a well-marketed and well-

attended recreational industry could add value to choice stocks pursued by anglers. There is also a new specialty gear-type added for the *lost valley* fleet, live rockfish capture; the value of rockfish caught with this gear type is assumed to be worth 10 times more than other commercial gear types. Market prices for the *lost valley* fleet are reported in Appendix Table A6.1.1.

As with the historical 2000 fleet, operating costs for all gear types was assumed to equal 60% of catch value based on Anon. (1994), and an additional 20% sailing cost was included to bring net profit to 20%, as in Beattie (2001). For trap fisheries, sailing-related costs are reduced by 10% and effort costs increased by 10%.

Policy search routine

The policy search routine in Ecosim is used to identify the optimal fleet-effort configuration that maximizes harvest benefits from the historic systems over a given time horizon. The simulation length used to calculate ORBs is not critical provided that it is long enough to allow the ecosystem to adjust to its new long-term equilibrium. All simulations tested in this chapter apply a 50-year time horizon. In most cases, this is a long enough simulation time to result in steady-state dynamics, or at least eliminate non-repetitive dynamics. To verify that each ORB solution represents a stable system, I quantify in this chapter the residual biomass dynamics present at the simulation end-state. The stability of the ecosystem is judged based on biomass fluctuations in the last 5 years of the simulation, and is reported as an average coefficient of variance across functional groups. An ORB restoration goal should only be considered if it can be demonstrated as a stable equilibrium. Any severe biomass fluctuations present will invalidate the ORB solution as a policy goal. I did not encounter chaotic behaviour in any of the simulations.

Using a nonlinear optimization procedure called the Fletcher-Powell (FP) method⁹, the search routine iteratively adjusts fishing mortality on each gear type in the model. It runs the harvest simulation and evaluates socio-economic and ecological success measures until the optimal vector of fishing mortalities is discovered that maximizes harvest benefit. As programmed by Christensen *et al.* (2004a), harvest benefit is defined by a multi-criterion objective function (OBJ) that can accommodate economic, social or ecological harvest priorities (eq. 6.1).

$$\text{OBJ} = W_{\text{ECON}} \cdot \sum \text{NPV}_{ij} + W_{\text{SOC}} \cdot \sum \text{Jobs}_{jt} + W_{\text{ECOL}} \cdot \sum \text{B/P}_{it} + W_{\text{MR}} \cdot \sum \text{MR}_{it} \quad \text{Equation 6.1}$$

W_{ECON} , W_{SOC} , W_{ECOL} and W_{MR} are, respectively, relative weighting factors applied for economic, social, ecological and mandated ecosystem rebuilding criteria. The summed terms evaluate socio-economic and ecological benefits of the harvest plan across each functional group (i), gear type (j) and simulation time step (t), the latter is set by default at monthly intervals. I have modified the ecological and MR criteria as described below.

In this chapter, each of the four historical periods is optimized under five harvest objectives, creating 20 ORBs for review as candidate restoration goals. The five harvest objectives tested are: an economic objective maximizing fishery profit; one maximizing social utility in the form of direct employment in the fisheries; two ecological objectives (ecosystem maturity and biodiversity); and a mixed objective function that balances economic and ecological (maturity) objectives. The policy objectives are explained in detail below.

⁹ The Fletcher-Powell (Fletcher and Powell, 1963) algorithm uses a conjugate gradient search method. Like steepest descent (SD) methods, the gradient vector along the response surface (approximated locally as a quadratic function) is taken as the direction of the maximum rate of change of the function; this vector provides the starting direction for the iteration. Unlike SD, conjugate methods also consider the history of gradients to move more efficiently towards the optimum. NB: An alternate optimization procedure is available in Ecosim based on the Davidon-Fletcher-Powell method (Davidon, 1959).

Economic criterion

By default in Ecosim, the NPV is the metric for assessing economic benefits of a harvest plan. NPV condenses the expected economic benefit of all forecasted years into a single term. In calculations, the discount rate used represents an assumed human preference for early benefits. The form of the discounting equation can be made to represent either conventional discounting or the intergenerational form introduced by Sumaila (2001) (also see Sumaila, 2004; Sumaila and Walters, 2005). Intergenerational discounting is used by default in Ecosim¹⁰, where the standard discount rate (δ) is 4% and the rate for future generations (δ_{fg}) is 10%. The properties and applications of intergenerational discounting are discussed in Chapter 2 and elsewhere (e.g., Sumaila, 2001, 2004; Sumaila and Walters, 2005; Ainsworth and Sumaila, 2003; 2005). In short, bequest value is included in the present value term. Optimizing for intergenerational NPV will return more conservative harvest policies than conventional NPV.

However, for the current exercise, we are interested only in maximizing the equilibrium-level profit of the ORB system – that is, the end year of the theoretical 50-year simulation. The NPV of the policy is not relevant because the “fishing down” of the historic system is done only in simulation to determine the ORB configuration; it is never meant to be recreated in reality by the BTF approach. (NB: The practical harvest plan will come when we rebuild the current ecosystem to the ORB condition in Chapter 7). To have the economic objective maximize end-state profit a very low discount rate, approaching zero, is used in the analysis¹¹. Ecosim calculates catch value in each simulation year based on market prices for the *lost valley* fleet (Appendix Table A6.1.1); costs of fishing are assumed to be 60% of landed value based on Anon. (1994).

¹⁰ The conventional form of the discounting equation can be accessed by setting the intergenerational discount rate = 20 (C. Walters, *pers. comm.* UBC Fisheries Centre). This number is not arbitrary; only a rate of 20 (i.e., $d_{fg} = 2000\%$) will deactivate intergenerational discounting due to the specific code structure used in Ecosim. The generation length is assumed to be 20 years in the intergenerational form.

¹¹ A non-zero value is required because of the specific coding used in the economic evaluation routine (C. Walters, *pers. comm.*).

Social criterion

Under the social criterion, harvest benefits are assessed as the total number of jobs directly produced by the harvest plan, summed across each gear type and simulation year. The number of jobs is calculated as the sum product of catch value (calculated internally in Ecosim) and jobs-per-unit-catch-value, as input on the policy search form. Jobs-per-unit-catch-value for all gear types in the *lost valley* fleet was assumed to equal 1, so that total employment is proportional to catch value. Future efforts should use better estimates of employment rates per gear type, although to determine appropriate values would be a substantial task and would likely require interviews to consider fisheries and supporting industries. For now, a simple weighting scheme may be appropriate, since EwE handles the employment estimates in a coarse way and provides only a rough estimate of employment figures. In Chapter 8 (section 8.2: Social values reconsidered), I comment on the limitations of the current EwE code for determining social harvest benefits, and suggest alternate employment models that would improve estimates of employment rates.

Ecological criterion

For the ecological criterion, Christensen *et al.* (2004a) expressed harvest benefit according to a proxy for ecosystem maturity, after Christensen (1995). Biomass over production (B/P) is summed across functional groups and simulation years. B/P ratios for functional groups calculated from the northern BC models are in Appendix Table A6.1.2. Use of the index was inspired by E.P. Odum's description of mature ecosystems (Odum, 1969). The optimization criterion is used in practice to reduce and reverse simplification of ecosystems attributable to fishing.

A recent addition by this author allows an alternate function to be used for the ecological criterion - biodiversity based on the Q-90 index. But in this chapter, the biodiversity objective function utilizes the existing Q-75 code in EwE (Christensen *et al.*, 2004a), changing only the quartile boundaries to 10-percentiles. As explained in Chapter 2, this is not precisely the same form of Kempton's index as was used by Ainsworth and Pitcher (*in press*). However, the two correlate adequately well under most circumstances.

Balanced multi-criterion objective function

Combining economic and ecological criteria in the objective function causes the search routine to maximize the weighted sum of fisheries profit (dollars) and B/P per functional group. Optimizations performed using the multi-criterion objective function may represent more a likely management approach to ecosystem restoration than purely economic, social or ecological approaches.

Balancing the two criteria for a combination run is not straightforward. Placing equal weightings on both criteria rarely results in an equal improvement in each field. This is entirely due to a mismatch in the units measuring economic and ecological utility. Since there is no intrinsic comparability between the objective function units (dollars and B/P), then the relative weightings used to parameterize the policy search are meaningless - only the relative improvement in each field over baseline is significant. In particular, the ecological return from the system (average system B/P) remains virtually unchanged unless it receives a heavy weighting in the policy search relative to the economic criterion.

Mackinson (2002) tried a multi-criterion objective function on a model of the North Sea. He found that the relative improvement in ecosystem maturity (B/P) consistently failed to match the relative improvement of social and economic criteria, and the ratio barely improved using a higher relative weight for ecology in the objective function. However, he used a relatively small weighting for ecology; the largest weighting he applied was 1:1:10 for economic, social and ecological criteria. Zeller and Freire (2002) likewise found that the relative improvement in ecology was insensitive to the ecological weighting factor, and Buchary *et al.* (2002) found that a 1:1:1 mixed search for economics, social and ecological benefit results in an optimal policy that was very similar to their social optimization. However Ainsworth *et al.* (2004) found that a ratio of 1:1:100 for economic, social and ecological criteria provided equal increase in harvest benefits versus the model baseline for northern BC models (Ainsworth *et al.*, 2002) and Newfoundland models (Pitcher *et al.*, 2002a). As there is no comparability between harvest benefits, there is no right or wrong weighting to use. But to see an effective increase in average system B/P over the course of a 50 year restoration plan, a high ratio is needed in the ecological objective. A ratio of

1:100 for economics and ecology is therefore used for all 'mixed' objective optimizations in this chapter.

Where I have standardized the input to the multi-criterion objective function at a 1:1:100 ratio, other authors have standardized the output so that the proportional increase in all fields is equal (e.g. Pitcher *et al.*, 2004). However, I maintain that neither method is more appropriate since there is no inherent comparability between the economic, social and ecological harvest benefits. An equivalent increase in each criterion (e.g. economics, social and ecology) is a meaningless artifact of the units used to express benefits. For example, a harvest plan that increases the B/P of the system by a given amount, say 10%, may produce a very different result if we measure ecosystem maturity instead in terms of trophic connectivity, prevalence of specialists, or average food chain length. Equating a dollar value to any of these ecological changes is arbitrary. It is only the pattern of trade offs that is interesting, and several ecological factors should be considered simultaneously.

Preventing depletions

In preliminary work determining ORB restoration goals from historic systems, Ainsworth and Pitcher (2005b) used Ecosim's 'mandated rebuilding' option *ad hoc* to prevent functional group depletions beyond a certain biomass threshold. By iteratively increasing the weighting of MR in the overall objective function, those authors were able to preserve the biomass of species that would have otherwise been depleted, especially by the exploitative economic and social optimizations. However, the method was subjective. A relative weighting for the MR function was required that was large enough to prevent depletions, but small enough so that the principle policy objective (e.g., economic return) was not impaired.

With the addition of a new check box in the policy search form by this author (see Fig. 7.1), the *ad hoc* technique is obsolete. When the 'No Extinctions' box is marked, the routine will return only fishing policies that maintain functional group biomass above the selected threshold. The default threshold value is used for all simulations in this chapter, so that no functional group biomass in the ORB configuration is allowed to fall below 5% of the historic level. This procedure has the effect that the ORB goals for restoration must somewhat resemble the parent

historical period and the policy search routine is not completely free to restructure the historic ecosystem - extirpations are prohibited.

Response surface analysis

This chapter introduces a new objective methodology to classify the response surface into one of three categories: a global maximum, multiple local maxima or a plateau surface. Two statistical techniques formalize the classification. A two-way analysis of variance test (ANOVA) measures how disperse the optimal solutions are in absolute terms (with a tight clustering, $p < 0.05$, indicating a global maximum). Hierarchical cluster analysis differentiates whether multiple local maxima are present, or whether a plateau surface is present based on the pattern of clustering. If there is a small number of clusters, then multiple local maxima are said to be present, but if there is a large number of clusters (indicating a continuous spread of points on the response surface) a plateau surface is said to be present. An arbitrary number of clusters is set as a threshold, below which the former condition applies and above which the latter condition applies. The threshold was chosen to produce an equal number of cases among the two possibilities (threshold is 10 clusters, based on the squared Euclidian distance separating optimal F vectors).

Multidimensional scaling (MDS) is also used in this chapter to illustrate the response surface in two dimensions for visualization purposes, although the technique did not contribute to any quantitative statistical application. Multivariate statistical analyses were performed using SPSS v.10.0 software package.

Addressing parameter uncertainty

An automated Monte Carlo routine in Ecosim is used to test the consequences of uncertain parameters on the fisheries optimization procedure, and quantify uncertainty surrounding the optimized ORB biomass goals for functional groups. Input Ecopath biomass and production rate (P/B) values are varied for all functional groups assuming a uniform probability distribution. The routine re-samples the Ecopath model using randomly generated data until it finds a combination of new data points that produces a balanced model. The coefficient of variation

(c.v.) for biomass is taken as 10%; c.v. for production rate is 5%. A recent contribution by this author provides an output form for the Monte Carlo routine to summarize variance in key parameters (biomass, catch, P/B, Q/B, M) and estimate depletion risk.

At present, the Monte Carlo routine has a limit of 2000 iterations. After that point, it uses the final iteration whether the solution produced a balanced model or not. I therefore used low c.v.'s to ensure that the large majority of samples resulted in balanced models. The variations surrounding ORB value and biomass structure estimated in this chapter may therefore be a minimum estimate. Ideally, we should use a wide confidence interval for unsure data points, such as biomass and production variables used in models of the ancient past.

However, this would require substantial modifications to the existing Monte Carlo routine. Simply increasing the maximum number of iterations is not a sufficient remedy. Although it would ensure that all iterations result in balanced solutions, it would still exclude extreme combinations of initial data values because they tend to result in unbalanced models. Ideally, the automatic mass-balance routine of Kavanagh *et al.* (2004) should be used at each iteration so that any reasonable combination of Ecopath data values could be considered. There is an option in the Monte Carlo routine to use coefficients of variation based on the data pedigree from Ecopath, but experience suggests that the option is not generally usable: confidence intervals provided by default are typically too large for a balanced solution to be located in only 2000 iterations. Since the proportion of unbalanced runs is not reported (presently) to the user, the procedure could provide misleading results.

6.3 Results

Appendix Fig. A6.2.1 shows the results of all random-F initializations ($n = 25$) for each of the 20 ORB ecosystems investigated (i.e., four historical periods optimized under five harvest objectives). The total observed range of values is presented, along with the mean and standard deviation of the random-F solutions. Equilibrium harvest benefits are shown for each ORB in a variety of terms: catch, trophic level of catch, standing commercial biomass, catch value and biodiversity using Q-90 and the Shannon-Weaver index (see Chapter 2 for methods). Table 6.4

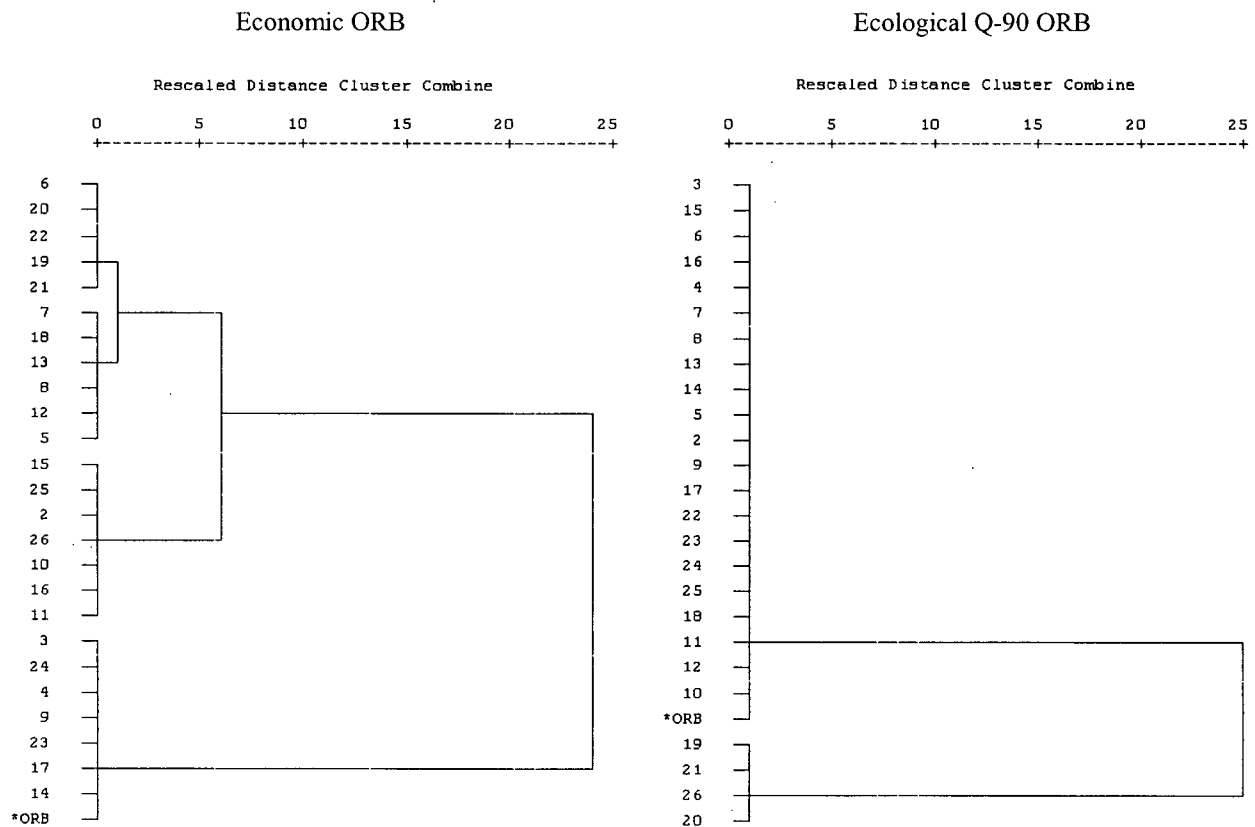


Figure 6.2 Cluster analysis of group biomass configurations for two example ORB ecosystems. ORB ecosystems are based on the 1950 period. Left figure shows economic optimum maximizing profit; right figure shows ecological optimum maximizing Q-90 biodiversity. 25 solutions are presented for each ORB based on random-F initializations of the policy search routine. Optimal solutions marked with an asterisk represent recovery targets pursued in Chapter 7. Similarity is based on squared Euclidean distance between functional group biomass vectors.

evaluates the rank order of harvest benefits for each period and objective function. Table 6.5 describes the average fleet-effort configuration associated with each ORB and reports whether residual biomass dynamics confound a clear equilibrium solution. From among the random-F repeats, an ORB should be accepted for a policy goal only if a steady-state solution is available. Table 6.5 also categorizes the shape of the response surface using objective criteria.

For each ORB, an example ecosystem is selected from among the 25 random-F repeats, and the optimal biomass values are presented in Appendix Table A6.2.1 with measurements of uncertainty determined through Monte Carlo resampling of Ecopath input data. Appendix Table A6.2.2 presents the landings by gear type for each of these example ORB solutions, with

confidence intervals. The example ORB ecosystems were selected objectively to represent the peak on the response surface that was identified most often. The 'peak' is a clustering of similar ORB solutions identified using hierarchical analysis of the type presented in Fig. 6.2. Shown in Fig. 6.2 are ORB solutions derived from the 1950 model (economic and Q-90 biodiversity harvest objectives). I will present these two ORB solutions several times throughout this results section and analyze them special detail because draft restoration scenarios (Chapter 7) will attempt to restore the present-day ecosystem into these particular forms.

Economic benefits of ORB ecosystems

Fig. 6.3 shows the fishery value of 20 ORB ecosystems. The data in Fig. 6.3 show the means of 25 random-F initializations (NB: the complete range of observations is presented in Appendix Fig. A6.2.1).

The greatest end-state value is offered by ORB ecosystems based on 1750 and optimized for social and economic benefits. BC fisheries had approximately a \$360 million value in 2004, (this figure includes revenue from the Straight of Georgia; DFO, 2004d). By comparison, the 1750 ecosystem of northern BC could sustainably deliver about \$1073 million annually. The 1750 system produces the most valuable harvest profiles under all harvest objectives, owing to its large abundance of valuable commercial species. The 1950 system offers the second best alternative followed by 1900 and

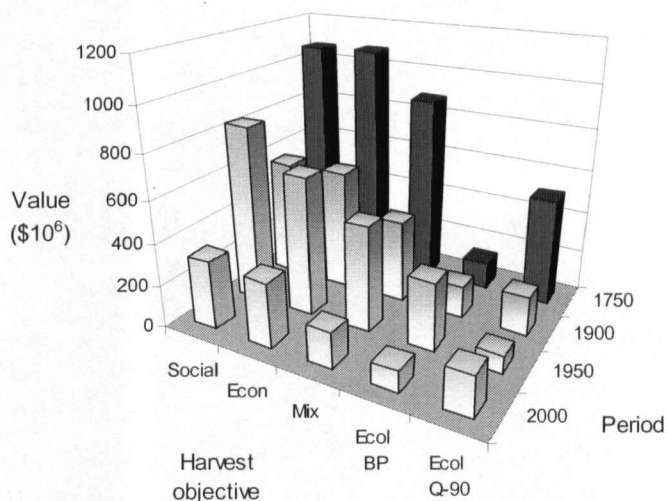


Figure 6.3 Value equilibriums for ORB ecosystems based on various historical periods. ORB ecosystems optimized to deliver social and economic harvest objectives generate the greatest harvest value. The pre-contact 1750 system shows the greatest potential for sustainable harvest value, and the 1950 system provides a much higher value than 2000. Values represent mean of 25 optimizations using random-F initializations.

2000. Kendall's concordance indicates that the rank order is on the margin of significance ($\chi^2 = 7.80$; $\chi^2_{(0.05,3)} = 7.82$) (Table 6.4). One might expect that 1900 would outperform 1950, since it possesses a greater resource density. The 1900 model can probably be blamed for this discrepancy, and it may reflect an incomplete knowledge of system dynamics occurring during that period (see discussion).

Overall, ORBs based on the 1750 system could provide about 5.5 times the value of current northern BC fisheries (social objective). This is a downgraded estimate from Ainsworth and Pitcher (2005b), who projected the potential value of the 1750 ecosystem at around 10 times the current value, based on the social optimum. The present Monte Carlo analysis establishes a 95% confidence interval between 3.5 - 8.0 times the real-world fishery value in 2000, and so the high estimate of Ainsworth and Pitcher (2005b) clearly falls outside of the likely range identified here (see Appendix Table A6.2.2 for range and standard deviation of Monte Carlo solutions). The difference is attributable to improved dynamic predictions resulting from the fitting procedure used in Chapter 5. In any case, such large economic benefits of the exploitative plans would only come at the expense of system health (see tradeoffs below), as social and economic optimizations tend to sacrifice a large amount of biodiversity from the historic system.

Fig. 6.3 also demonstrates the effect of the harvest objective. The greatest catch value is delivered by the social and economic objectives, the least value is delivered by the ecological objectives, and the mixed objective provides intermediate value (Table 6.4). The rank order is highly significant ($\chi^2 = 13.80$; $\chi^2_{(0.05,3)} = 7.82$).

To maximize total employment, the social objective increases effort by fishing uneconomically, disregarding the costs of fishing. The smallest exploitation rates are advocated by the ecological objective function for system maturity. The biodiversity objective, Q-90, tends to employ higher fishing mortalities than expected and creates a substantial catch value. This results stems from the fact that the index is based on the existing Q-75 code in EwE, which does not include reference to species richness, only species evenness (see Chapter 2). As a result, biodiversity can be increased by reducing population size. Under the social optimization, the policy search tends to identify fleet configurations that support a large harvest of invertebrates, and also a large

recreational sector (see Fig. 6.4). This amounts to reducing fishing in commercial gear types that compete with these sectors. Examining mixed trophic impacts for the *lost valley* fleet¹², the invertebrate fisheries operate largely independently of other sectors. In the case of ORB systems based on the ancient models, 1750 and 1900, it is primarily halibut longline and aboriginal terminal fisheries that compete with sport fisheries. With ORB models based on the recent past, 1950 and 2000, all fleets conflict with recreational fisheries to some degree because the baseline abundance of target species is lower.

Value per gear type

Fig. 6.4 shows the catch value per gear type at ORB equilibrium associated with each harvest objective (Y-axis) and period (X-axis). The economic potential of the 1750 system, as discerned by the social and economic objectives, lies in its large harvestable biomass of epifaunal invertebrates. The invertebrate fisheries are responsible for more than 75% of the revenue made available by this restoration goal. This may even be an underestimate, since the pre-contact ecosystem contains a higher relative proportion of valuable Northern abalone (*Haliotis kamtschatkana*) than the recent past models, although the species is not explicitly represented in the model. The ecological solutions for system maturity and biodiversity advocate a proportionately larger recreational sector than do the other objectives. The optimizations presented in Fig. 6.4 are key examples selected from among random-F initializations, representing the most common peaks located by the policy search routine. Details on these solutions are provided in Appendix Tables A6.2.1 and A6.2.2 (biomass and landings, respectively).

¹² Mixed trophic impacts is a form of sensitivity analysis that summarizes the net impact of functional groups and fisheries on each other, considering direct and indirect trophic interactions caused by predation and competition. The routine is based on the Leontief matrix (Leontief, 1951), and was applied to Ecosim by Ulanowicz and Puccia (1990). Christensen *et al.* (2004a) provide more detail.

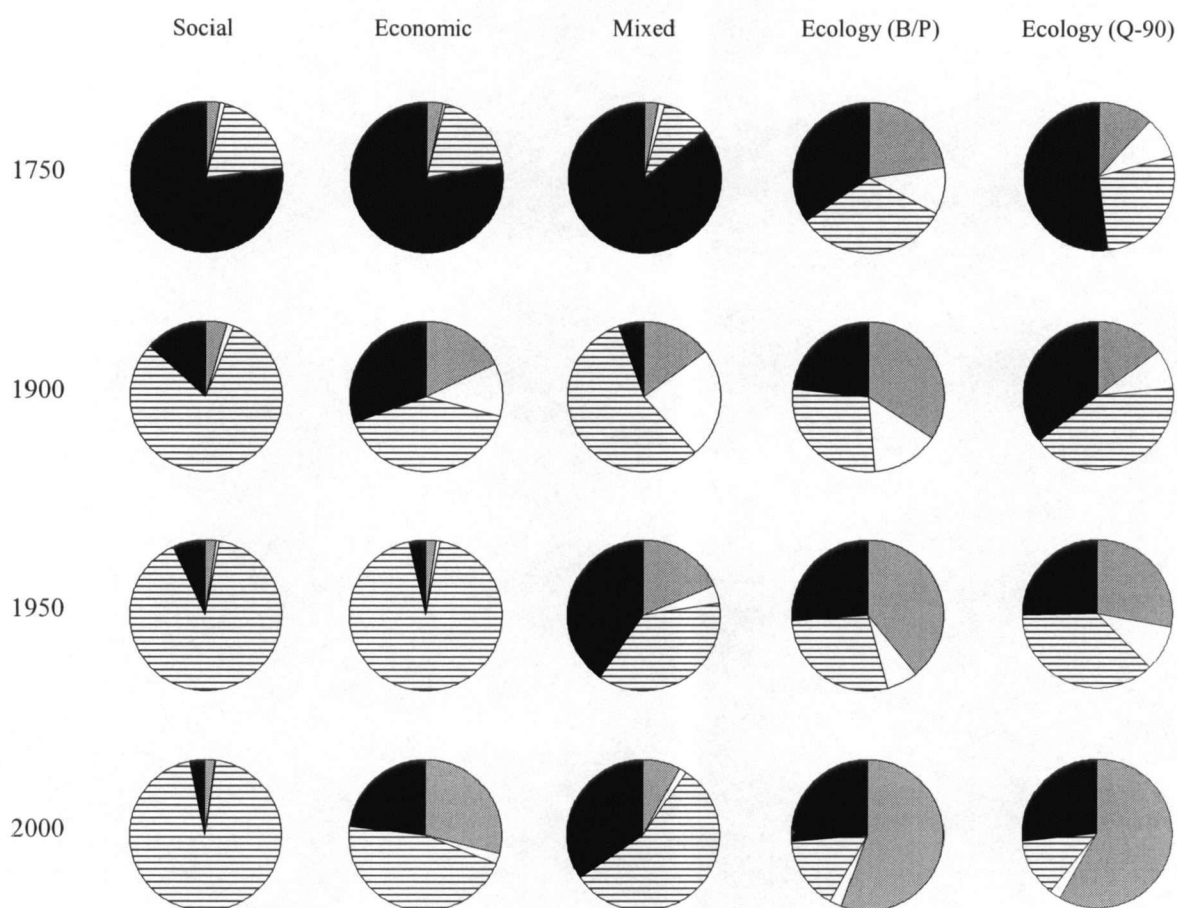


Figure 6.4 ORB equilibrium catch value per gear type. X-axis compares harvest objective; Y-axis compares restoration period. The solutions presented represent the response surface peaks most commonly located by the policy search routine. Black shows invertebrate fisheries, grey shows demersal fisheries, white shows pelagic fisheries, and horizontal bands show recreational fisheries. The pre-contact system provides its greatest social and economic benefit through invertebrate fisheries. The recent past models (1950 and 2000) rely on the recreational sector to generate wealth and jobs, and tend to spread fishing pressure out evenly among gear sectors in solutions maintaining ecological health.

Value per functional group

Fig. 6.5 shows equilibrium profits per functional group made available by the 1750 ORB system and the 2000 ORB system under the economic harvest objective, the mixed objective, and the ecosystem maturity objective. Fishery value is presented for highly commercial groups as a fraction of the current real-world profit from northern BC. The profits detailed in Fig. 6.5 represent the optimum profits per functional group necessary to maximize benefit from the

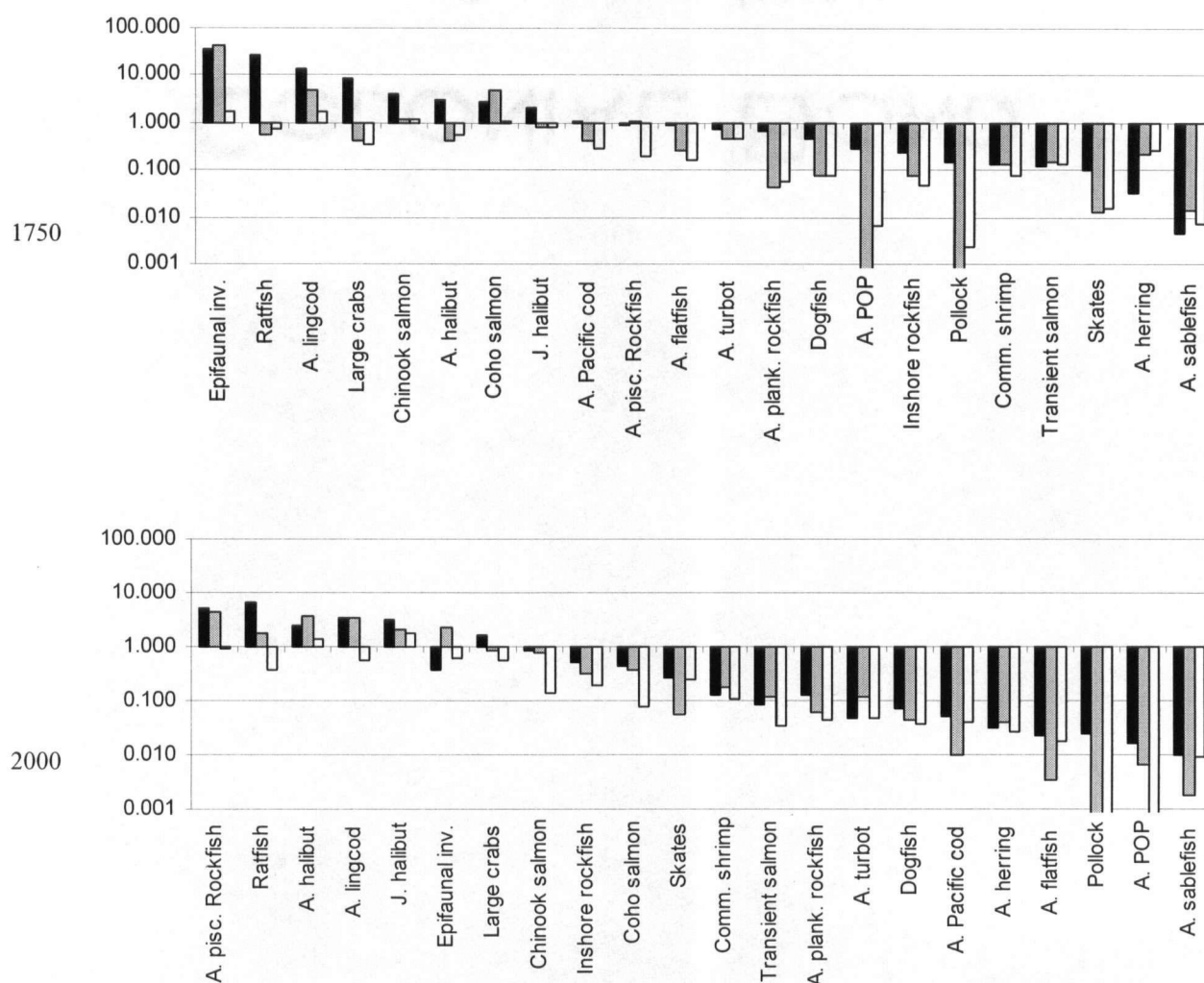


Figure 6.5 ORB equilibrium value by group under various harvest objectives. Black bars show economic objective, grey bars show mixed objective, white bars show ecological objective. ORB ecosystems are derived from 1750 and 2000 periods. Catch value of these optimal solutions is presented as a fraction of today's real value.

system as a whole, and not the maximum profit available from each individual group. The absolute levels of harvest, therefore, do not necessarily reflect the available production per group, but the optimum combination of harvests that will maximize system benefit.

Under the most exploitative solution, the ORB ecosystem of 1750 is able to provide more than 35 times the current value of today's epifaunal invertebrate fisheries (shellfish), and it is sustainable. The ORB lingcod fishery also sustains about 30 times the value of the contemporary

lingcod fishery. Mostly, this is a sad comment on the state of the stocks in northern BC. There are ratfish to spare, and crab fisheries also do well thanks to the pre-contact abundance. However, even if we were to restore northern BC's marine ecosystem to the abundance and diversity of the ORB goal based on the pre-contact period, current real-world harvests would still exceed the optimal rate for many functional groups. This suggests that present real-world profits for these groups may be unsustainable, or conflict with the optimal harvest profile through higher order interactions.

Even the 2000 ecosystem, after 50 years of restructuring into an ORB configuration, is able to deliver greater value in several fisheries – at least under the economic and social harvest objectives. The current pittance we receive for lingcod is easily improved upon by the optimal solution by a factor of 3.5 times the current value. The value for piscivorous rockfish increases greatly (5.3X, social objective) as does halibut (3.7X, economic objective). The improved catch rate is a product of 50 years of growth; the ORB solution based on the 2000 model increases the biomass of these groups by 90% and 15%, respectively. It is interesting that neither period, 1750 or 2000, maintains the current real value of the sablefish industry. The 1750 biomass estimate for adult sablefish ($0.191 \text{ t}\cdot\text{km}^{-2}$) was made by mass-balance in Ecopath, and it is uncertain (Appendix Table A5.1.2). If it were underestimated, then the restored value of the fishery could also be underestimated. However, the pre-contact biomass assumed for 1750 is already larger than the unfished vulnerable biomass used by Haist *et al.* (2004) ($0.168 \text{ t}\cdot\text{km}^{-2}$; assumes BC area is $113,000 \text{ km}^2$; Chapter 5) so this is probably not the case. Current catch rates for sablefish appear to be unsustainable or at least suboptimal under a variety of whole-ecosystem objectives.

Generally, ORB goals based on the 1750 ecosystem use higher fishing mortalities than goals based on the 2000 system. The optimal harvest strategies advocated by the policy search routine tend to deplete the system to a more productive level, but harvest strategies based on the 2000 system use lower harvest rates and allow the ecosystem to build up to ORB biomass levels. That is why there is a profit to be made from the 2000 ecosystem on currently depleted groups; equilibrium profits are higher than current levels because some rebuilding has occurred up to ORB biomass levels. Fifty years is enough time to thoroughly restructure the 2000 ecosystem, but the decision on what groups to restore depends largely on the price matrix in use.

Social benefits of ORB ecosystems

Jobs created

Fig. 6.6 (left) demonstrates the level of employment supported by ORB ecosystems. The social harvest objective generates the most jobs annually followed by the economic, mixed and ecological objectives. The rank order is almost significant ($\chi^2 = 7.80$; $\chi^2_{(0.05,3)} = 7.82$) (Table 6.4). As with the economic valuation, the 1750 system offers the best potential for employment followed by 1950 then 1900 and 2000.

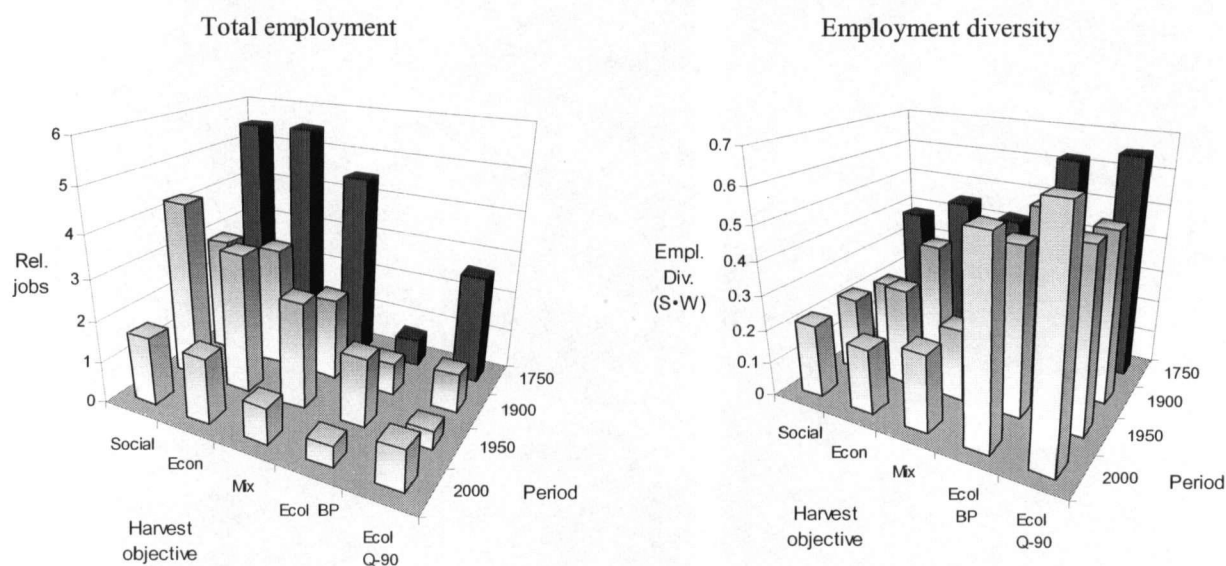


Figure 6.6 Social utility provided by ORB ecosystems based on various historical periods. Total employment (left) shows the relative number of jobs sustained at ORB equilibrium; employment diversity (right) reveals the concentration of fishing effort among *lost valley* gear types based on the Shannon-Weaver entropy function (Shannon and Weaver, 1949). ORB ecosystems optimized to deliver social and economic benefits tend to concentrate fishing in a few profitable gear types, while ecological runs spread effort across more sectors. Values represent the mean of 25 optimizations using random-F initializations.

Employment diversity

As a second social indicator, we have used the methodology of Attaran (1986) to measure the employment diversity (D) of the optimal harvest simulations, based on Shannon's entropy function (Shannon and Weaver, 1946) (Fig. 6.6; right). Chapter 2 describes how this function was applied to evaluate BTF restoration scenarios (also see Ainsworth and Sumaila, 2004b).

ORB solutions optimized for ecological harvest objectives tend to use lower exploitation rates than economic and social objectives, and spread out fishing more evenly across sectors. The rank order between harvest objectives is highly significant ($\chi^2 = 13.00$; $\chi^2_{(0.05,3)} = 7.82$) (Table 6.4); the ecological objectives provide a much more even distribution of fishing effort than the more exploitative harvest plans. The rank order between periods is not significant. However, the results suggest that 1750, at least, can maintain high catch rates simultaneously among several gear types, whereas the recent past must concentrate fishing in fewer sectors (i.e., exploiting the strongest stocks) in order to maintain high levels of employment.

Ecological benefits of ORB ecosystems

Biodiversity

Fig. 6.7 shows the biodiversity of ORB ecosystems. 1750 always outperforms the more depleted systems, maintaining a higher biodiversity equilibrium under harvests. 1900, 1950 and 2000 follow. It is predictable that the exploitative social and economic optimizations result in poor biodiversity, since the Q-90 measure considers group biomass. The ecosystem maturity (B/P) objective function settles on a higher optimum biodiversity than the Q-90 objective function. This is due to the fact that the policy search routine has maximized for a different version of the Q-90 index than the one used to compute results in Fig. 6.7. The optimization maximizes the Q-75 index of Christensen *et al.* (2004a) (modified to 10-percentiles rather than quartiles; see Chapter 2), while the analysis in Fig. 6.7 uses the method of Ainsworth and Pitcher (*in press.*).

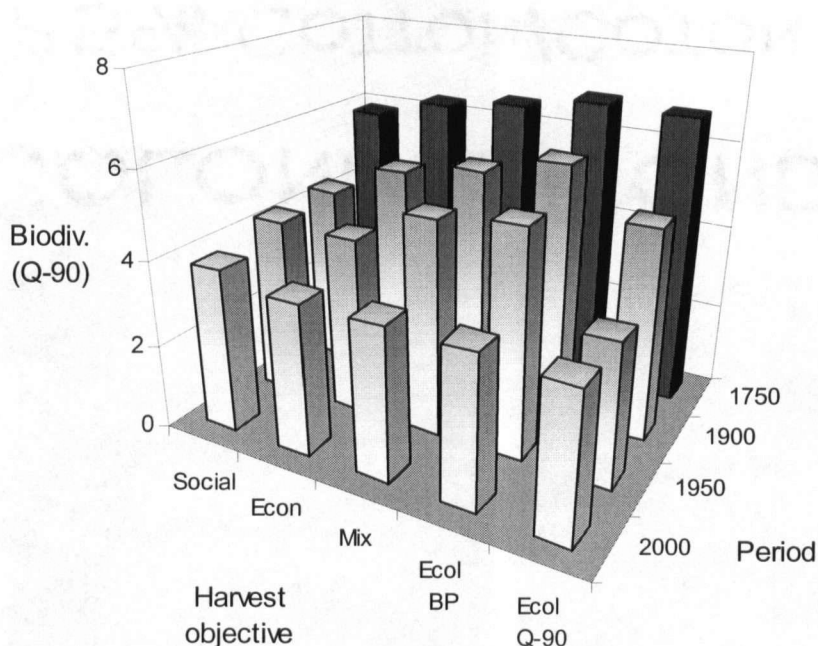


Figure 6.7 Biodiversity of ORB ecosystems based on various historical periods. ORB ecosystems optimized under social and economic criteria for harvest benefits tend to have a lower equilibrium-level biodiversity than systems optimized for ecosystem maturity. The biodiversity equilibrium is highest for ORBs based on ancient periods. Values represent the mean of 25 optimizations using random-F initializations.

Fig. 6.8 shows dynamic biodiversity trajectories for some example ORB simulations. These scenarios represent the most commonly identified fleet-effort pattern among the random-F initializations. There is a fair bit of variation in the specific biodiversity trajectories, but some generalizations can be made. ORBs calculated based on 1750 always lose biodiversity from the pre-contact level, which implies that there would be a heavy cost to maintain that high degree of biodiversity. All other periods exhibit optimal fishing solutions that increase or decrease the initial level of biodiversity (depending on the harvest objective). For all periods, the highest level of biodiversity is maintained by one of the ecological objectives, either ecosystem maturity (B/P) or Q-90 biodiversity. The 1950 system is especially responsive in simulations (see discussion). It can assume a wide variety of equilibrium positions, and optimal fishing plans are able to increase biodiversity from the depauperate state, even while satisfying social and

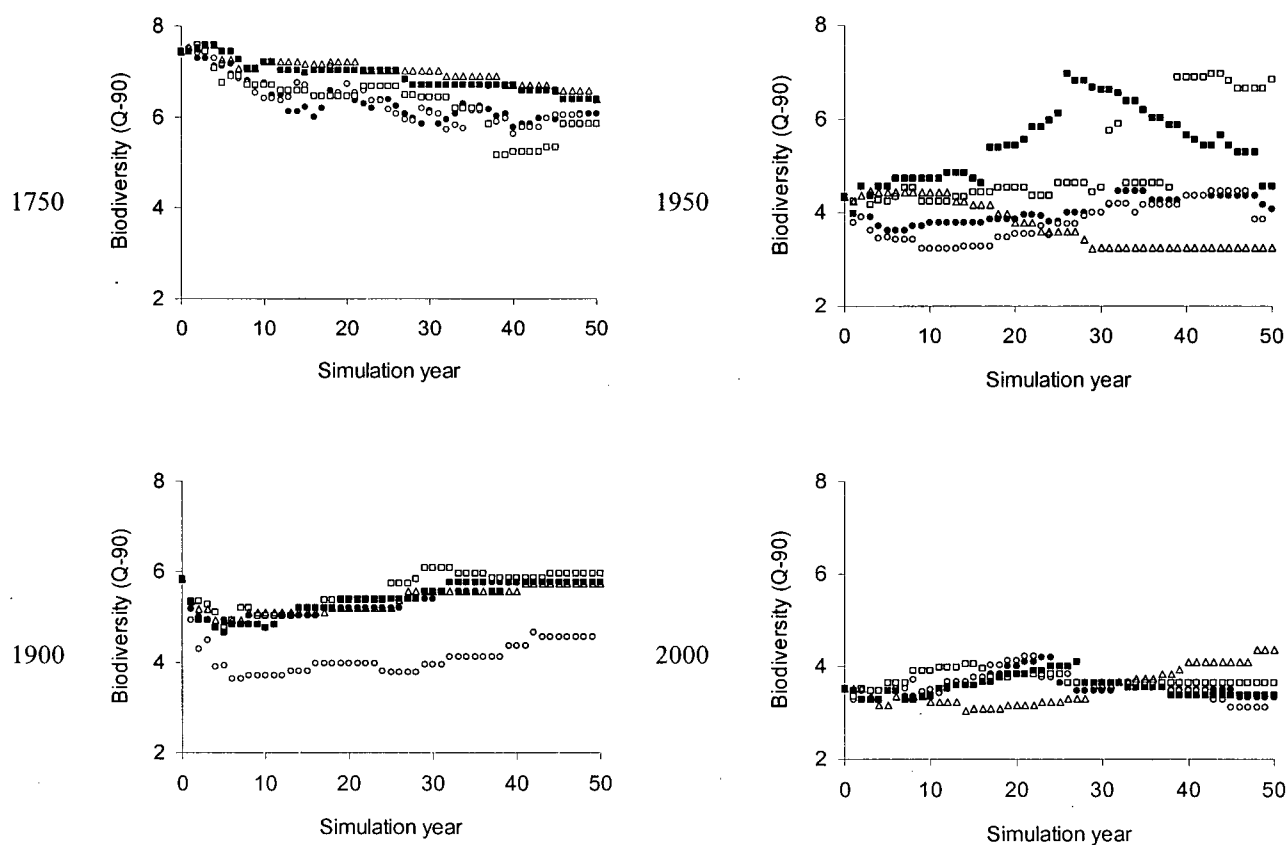


Figure 6.8 Biodiversity of historic ecosystems under optimal fishing policies using *lost valley* fleet structure. Simulation end-states represent ORB ecosystem configurations. The system is assumed to be at equilibrium after 50 years of optimal fishing. Closed circles show economic objective, open circles show social objective, closed squares show mixed objective, open squares show ecological objective (ecosystem maturity) and triangles show an alternate ecological objective (biodiversity).

economic demands. The 1950 ORB system, optimized for ecosystem maturity, achieves a particularly favorable increase in biodiversity compared to the initial system configuration.

Effect of period and objective function on harvest benefits

Table 6.4 shows the rank order of each period and objective function in its ability to produce ORB solutions that maximize economic, social and ecological gains. Kendall's concordance coefficient (W) shows when the rank order is significant (Kendall, 1962) and is calculated to compare benefits based on period and harvest objective. A variety of evaluation measures is

Table 6.4 Rank order of ORB ecosystem performance in various evaluation fields. A high value indicates that the period or harvest objective supports increased utility. Kendall concordance coefficient (W) determines if rank order is significant (corresponding chi-squared value is presented based on Zar, 1996); bold values show significant effects of period or harvest objective. Employment diversity is based on the Shannon-Weaver entropy function (after Attaran, 1986). Biodiversity is based on the Q-90 statistic and Shannon-Weaver function (see Chapter 2).

		Value	Catch	Jobs	Employment diversity	Biodiversity (Q-90)	Biodiversity (S-W)
Period	1750	1	1	1	4	1	1
	1900	3	3	3	2	2	2
	1950	2	2	2	1	3	4
	2000	4	4	4	3	4	3
	W	0.52	0.65	0.52	0.49	0.94	0.94
	χ^2	7.80	9.72	7.80	7.32	14.04	14.04
	$\chi^2_{(0.05,3)}$	7.82					
Objective	Social	1	1	1	5	3	2
	Economic	1	2	1	3	4	5
	Mix	2	4	2	4	2	4
	Ecology (B/P)	4	5	4	2	1	3
	Ecology (Q-90)	3	3	3	1	5	1
	W	0.86	0.79	0.86	0.81	0.46	0.19
	χ^2	13.80	12.60	13.80	13.00	7.40	3.00
	$\chi^2_{(0.05,4)}$	9.49					

considered: dollar value, relative number of jobs created, employment diversity, etc.. To assess the significance of the association represented by the Kendall coefficient, an equivalent chi-square value χ^2 was determined based on Zar (1996).

The 1750 period produces the most attractive ORB solutions from the perspective of gross equilibrium benefits. ORBs based on the pre-contact ecosystem are able to sustain large catch rates, generating wealth and jobs, while preserving the biodiversity of the system at a higher level than the other periods (Fig. 6.8).

Social and economic optimizations tend to concentrate fishing effort in fewer gear types than the ecological objectives. The option to 'specialize' fisheries on fewer target species is supported by all ecosystems as a means of generating wealth. In the case of the ancient system (1750), effort tends to focus on the invertebrate fisheries; with the more recent systems (1900, 1950 and 2000),

effort concentrates in the recreational sector (Fig. 6.4). The effect of period on employment diversity is not significant, but 1950 shows the greatest tendency to support diverse fisheries. This may be due to the fact that 1950 is generally depleted, and no single commercial group (or set of groups) can support intensive harvesting. 2000 is also depleted, but it has a prodigious biomass of halibut (Appendix Table A5.1.2) which enables a greater proportional take by the recreational sector. For example, when optimized for social utility, 90% of the catch supplied by the 2000 ecosystem comes from the recreational sector (79% of that amount is halibut). In the case of 1950, the recreational sector contributes less to overall landings (85%), and no single functional group is responsible for more than 35% of recreational catch (the main contributors are salmon groups).

The resulting ORB biodiversity is more dependent on the historical period used than the harvest objective. The Shannon-Weaver index considers species evenness alone, and so exploitative harvest plans do not necessarily result in reduced biodiversity. They can maintain the relative proportion of group biomass despite allowing serious depletions from the historic level.

Examining trade-offs

Fig. 6.9 shows the inherent trade off between exploitation and conservation contained in the ORB solution for each historical period. Along the X-axis are various ORB solutions determined by the policy search routine, varying the harvest objective and the fishing mortality vector used to initialize the search. At the left extreme of the X-axis are harvest plans that generate large revenues from the restored system at the expense of biodiversity (typically social and economic optimizations), while on the right are plans that preserve biodiversity but return only modest harvest value (ecological objectives). Between these extremes lie mixed objective runs, which provide an intermediate trade-off. All systems except 2000 exhibit a clear decline in ORB system biodiversity with increasing profits: there is little variation in the 2000 output, regardless of the objective function in place. This may indicate that there is little scope to increase profits sustainably beyond the 2000 baseline, and that biodiversity cannot be reduced any further from the baseline level without impacting socioeconomic benefits. In other words,

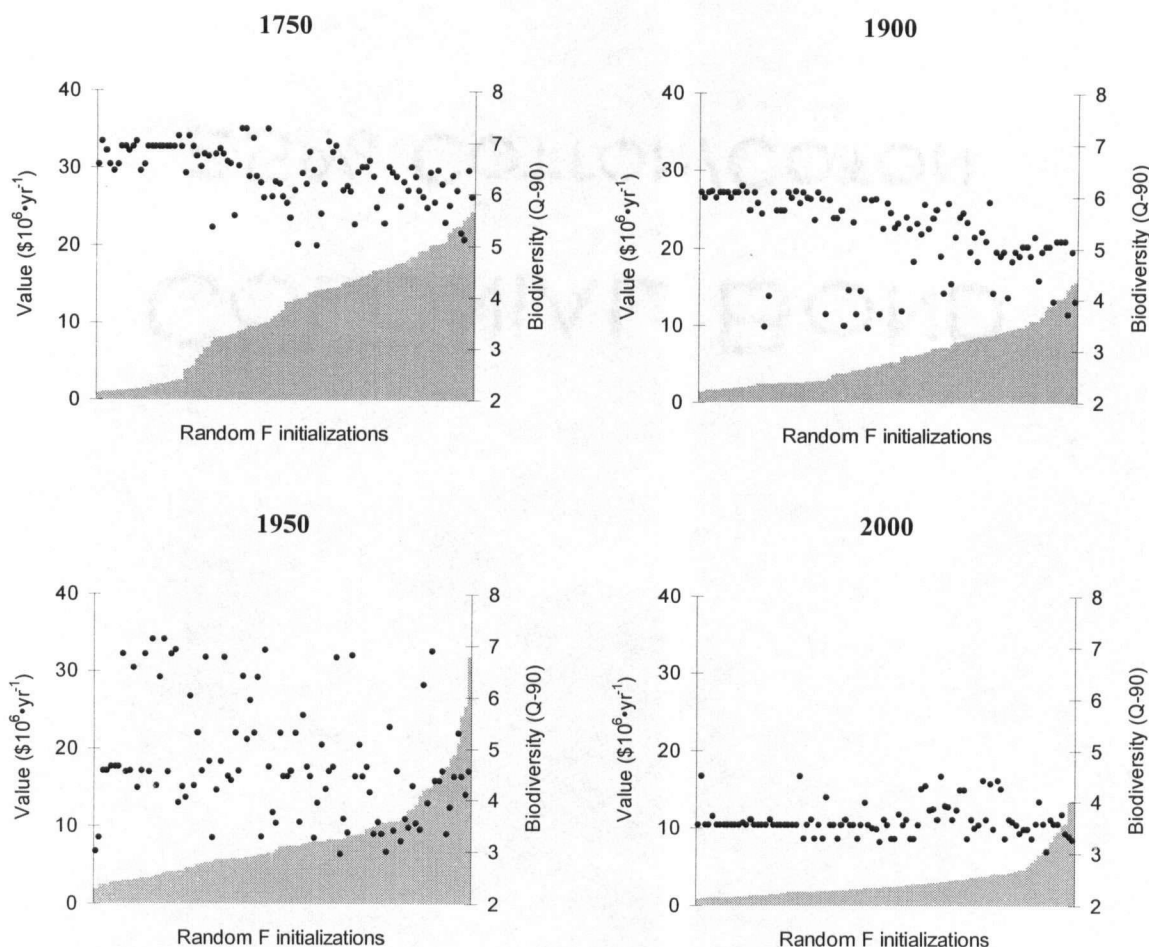


Figure 6.9 Profit and biodiversity of ORB equilibria based on 1750, 1900, 1950 and 2000 periods. Closed circles show biodiversity (Q-90) and shaded area shows harvest value. X-axis shows various ORB ecosystems determined using random-F initializations of the policy search routine, and increasing (from left to right) the weighting of socioeconomic benefits in the objective function relative to ecological benefits. A continuous trade-off between socioeconomic and ecological utility is presented for each period; ORB ecosystems optimized to deliver economic benefits tend to sacrifice system biodiversity, and vice versa. A candidate ORB goal for restoration should be selected from some region along the X-axis according to contemporary social priorities.

ORBs geared for socioeconomic and ecological returns converge on the same optimal policy solution for this depleted system.

The absolute level of profit and biodiversity achieved by ORB solutions is less for models representing more recent ecosystems. For its most lucrative ORB solution (determined under the

economic harvest objective), the 1750 period generates a greater profit than the other historical periods, about \$22,500 per km², almost \$1.6 billion per year for the study area. This enormous value was located only once by random-F solutions, but the mean value of 1750 economic runs is still \$15,400 per km²; this is significant when compared to the current real-world value of \$2,790 per km². For that level of performance, biodiversity of the 1750 state is sacrificed. On average, the Q-90 value of the resulting ORB is reduced from 7.4 (the historic value) to 6.3.

The maximum profit available from ORBs based on the 1900 period is about \$8,200 per square kilometer, or about \$574 million for the entire area. That amount corresponds to a smaller loss of biodiversity, from 5.5 (historic Q-90) to 5.2 (average ORB for economic objective). The 1950 period can produce \$9,380 per square kilometer, or \$656 million for the total area under the economic objective. That corresponds to a still smaller proportional loss of biodiversity - from 4.4 to 4.2. ORBs based on the 2000 system could produce \$4,400 per square kilometer, almost \$308 million annually. That represents a 57% increase over the current real-world fishery value. Importantly, this economic optimization actually results in a slight increase in biodiversity (3.53 to 3.55). With depleted systems then, we risk far less to maximize the value of the fishing industry. In fact, given the currently depleted state of fisheries in northern BC, an economic solution demands that we protect system biodiversity and even increase it.

Fig. 6.10 evaluates the social utility of ORB solutions for each period. The figure compares total employment, considered proportional to catch, with employment diversity. A similar trade-off emerges as in Fig. 6.9. The 1750 ecosystem promises a greater number of jobs than the more recent periods. The great abundance of commercial species in 1750 permits 'specialized' harvest policies that concentrate fishing effort on a relatively small proportion of species in order to provide more jobs overall. These solutions correspond to the right side of the graphs in Fig. 6.10. Effectively, the 1750 ecosystem is restructured by the ORB solution to augment production in the invertebrate groups. This ancient system exhibits the clearest tradeoff between total employment and employment diversity, because many harvesting options exist due to the ample biomass of target species. With the more recent ecosystems, 1900, 1950 and 2000, there are less available jobs. Highly exploitative strategies are rarely optimal, and even scenarios designed to maximize fishery production (and therefore jobs) tend to use conservative

exploitation rates spread out evenly among target species. Fishing options have been foreclosed with the more recent ecosystems due to less available commercial biomass, and only a diminished level of employment can be sustained compared to the ancient system.

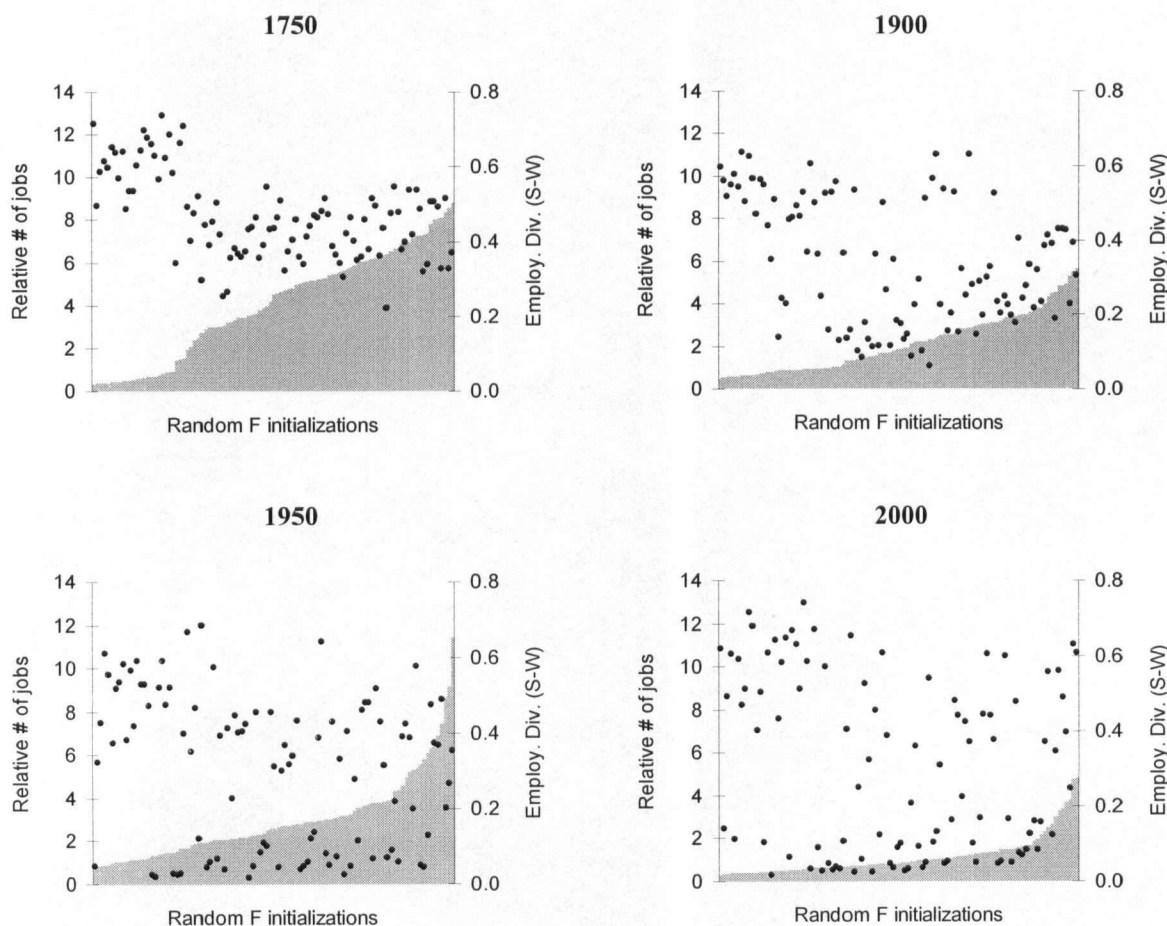


Figure 6.10 Social utility provided by ORB ecosystems based on 1750, 1900, 1950 and 2000 periods. Closed circles show employment diversity (based on Shannon-Weaver entropy function) and shaded area shows the relative number of jobs sustained at equilibrium (where 1 corresponds to the total employment offered by northern BC fisheries in 2000). X-axis shows various ORB ecosystems determined using random-F initializations of the policy search routine, and increasing (from left to right) the weighting of ecological benefits in the objective function relative to socioeconomic benefits. There is a continuous trade-off between total employment and employment diversity. Optimal solutions favouring the ecology of the system (left most solutions along X-axis) tend to have light exploitation rates, employ fewer people, and spread out exploitation across gear types. Solutions favouring socioeconomic returns tend to concentrate fishing effort in a few number of profitable sectors, most notably in the ancient systems. A candidate ORB goal for restoration should be selected from some region along the X-axis according to contemporary social priorities.

Response surface analysis

Fig. 6.11 examines the response surface geometry of three example ORB scenario categories: a global maximum, multiple local maxima, and a plateau surface. MDS reduces the 12-dimensional gear-space to two dimensions (i.e., 12 gear types of independently varying effort are used in the *lost valley* fleet). A two-way ANOVA tests the similarity of the resulting optimal fleet-effort configurations. A global maximum occurs when there is a low absolute diversity between solutions (high p value); the presence of multiple local maxima will generate two or more tight clusters that are dissimilar to each other (low p value; few clusters); a plateau will generate a loose cluster containing dissimilar solutions (low p value; many clusters). Only scenarios that most clearly demonstrate these surface geometries are presented. Table 6.5 presents the full results for ANOVA and cluster analyses, and reports on the stability of the ORB solution by quantifying residual biomass dynamics at simulation end-state.

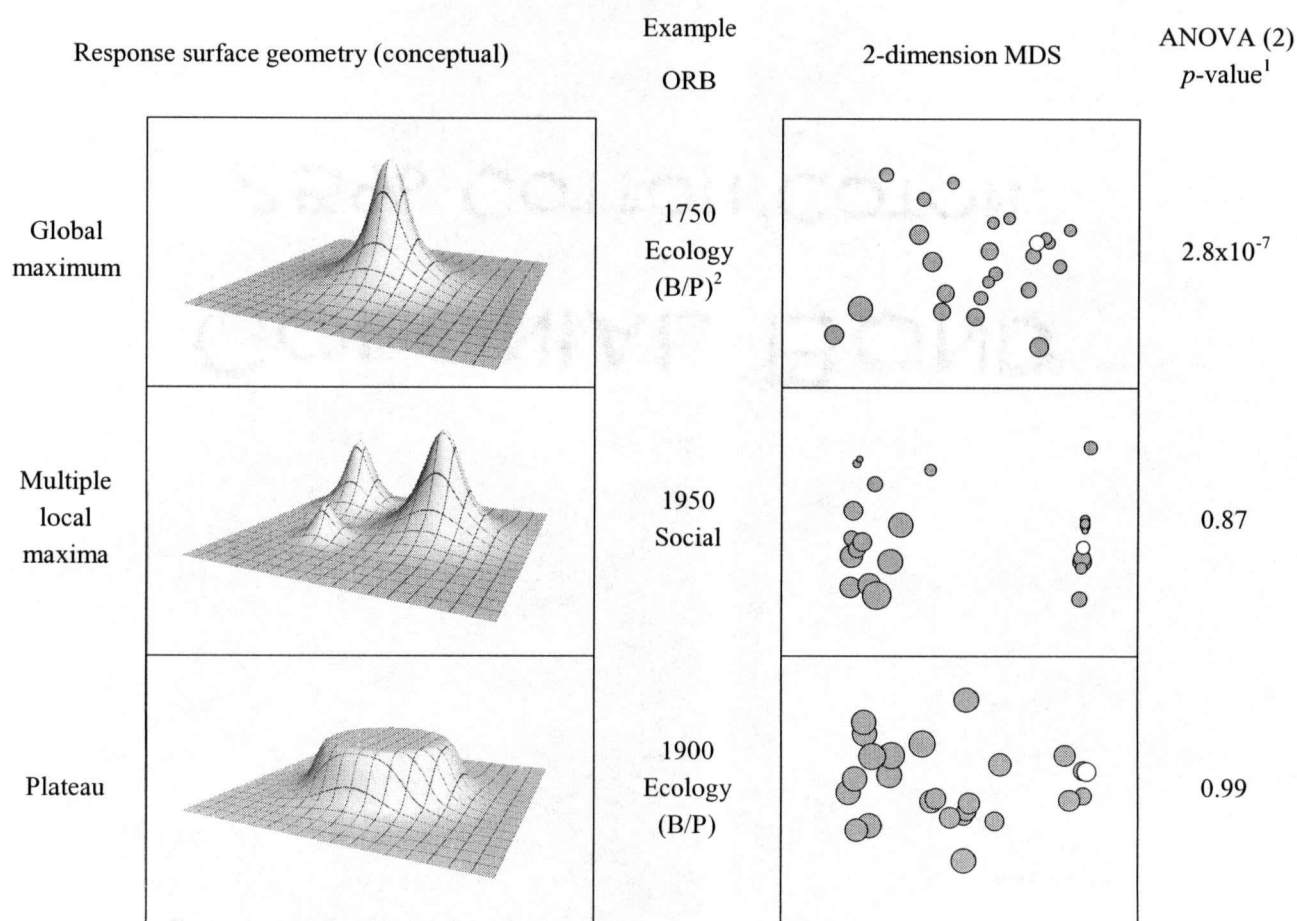


Figure 6.11 Response surface geometries. Conceptual illustrations of three geometries are presented, along with examples from the ORB targets calculated in this chapter. Multi-dimensional scaling (MDS) reveals the degree of clustering among optimal fleet-effort solutions determined using 25 random-F initializations. A two-way ANOVA test analyzes variance surrounding optimal fishing mortalities; a tight clustering of points is revealed by a low p value (i.e., global maximum); a high p value indicates a broad peak (with multiple local maxima or a plateau surface). The area of the circle is scaled to represent relative fishery value (dollars); white circles denote the representative optimizations presented in Appendix Tables A.6.2.1 and A.6.2.2. Geometry of the response surface holds implications for management. Pursuing a global maximum as a policy goal will require precise maintenance of fleet-effort to ensure maximum fishery benefits are realized. Multiple local maxima may yield similar net benefits overall, but provide very different allocations among fishing sectors. A plateau surface indicates that variations in the fishing pattern, as may be encountered in a practical input control management policy, will still result in near-maximum benefits.

¹ Full results presented in Table 6.5.

² An outlier was removed.

Table 6.5 Fishing rates of ORB solutions, analysis of response surface geometry and ecosystem stability. Hierarchical cluster analysis indicates relatedness of 25 ORB fleet-effort structures determined using random-F initializations. MDS clustering is based on similarity of optimal fishing mortality vectors measured using squared Euclidean distance; a high number of clusters indicates that dissimilar solutions are present. Two-way ANOVA shows the overall diversity of solutions. A low ANOVA p value indicates that all ORB solutions form a common peak on the response surface (global maximum). A high p value indicates a loose cluster of points, either multiple local maxima or a plateau. Local maxima are considered present when solutions form a small number of clusters (<10); a plateau is considered present when solutions form a large number (i.e., continuous solutions). The optimal fishing rates presented here produce ORB configurations represented in Appendix Fig. A6.2.1. A fraction of random-F repeats result in residual (generally cyclic) biomass dynamics; these solutions are less appropriate as policy targets than steady-state equilibrium solutions.

Period	Objective	Average fishing rate (y^{-1})								Response surface geometry			Residual biomass dynamics	
		Salmon	Halibut	Demersals	Inverts.	Rockfish	Juv. fish	Herring	Other	Cluster analysis ¹ (# of peaks)	ANOVA (2) (p - value)	Geometry ²	Mean biomass C.V. (last 5 years)	Fraction of solutions unstable at endstate ³
1750	Social	0.142	0.114	0.044	0.072	0.046	0.025	0.026	0.006	17	0.98	Plateau	1.8E-03	80%
	Economic	0.103	0.118	0.048	0.113	0.039	0.032	0.018	0.003	10	0.49	Local maxima	7.9E-04	56%
	Mixed	0.069	0.170	0.058	0.059	0.013	0.051	0.013	0.001	8	0.57	Local maxima	1.2E-04	8%
	Ecology (B/P)	0.037	0.118	0.037	0.010	0.011	0.036	0.016	0.001	17	< 0.05	Global maximum	5.9E-05	0%
	Ecology (Q-90)	0.071	0.123	0.036	0.036	0.021	0.031	0.026	0.005	9	0.42	Local maxima	2.8E-04	24%
1900	Social	0.707	0.321	0.117	0.076	0.065	0.050	0.022	0.006	5	< 0.05	Global maximum	1.7E-04	44%
	Economic	0.447	0.272	0.075	0.077	0.046	0.042	0.011	0.003	9	< 0.05	Global maximum	1.7E-04	24%
	Mixed	0.396	0.223	0.066	0.058	0.027	0.041	0.017	0.002	10	< 0.05	Global maximum	1.5E-04	0%
	Ecology (B/P)	0.112	0.152	0.027	0.011	0.010	0.027	0.020	0.001	15	0.99	Plateau	1.5E-04	4%
	Ecology (Q-90)	0.123	0.121	0.027	0.036	0.023	0.019	0.021	0.005	15	0.89	Plateau	9.1E-05	4%
1950	Social	0.782	0.355	0.131	0.053	0.081	0.044	0.026	0.008	8	0.87	Local maxima	1.2E-03	92%
	Economic	0.552	0.337	0.076	0.037	0.052	0.035	0.027	0.005	5	0.23	Local maxima	8.1E-04	100%
	Mixed	0.674	0.312	0.117	0.033	0.050	0.047	0.025	0.003	5	0.36	Local maxima	6.3E-04	92%
	Ecology (B/P)	0.205	0.289	0.042	0.015	0.012	0.042	0.027	0.002	7	1.00	Local maxima	8.3E-04	88%
	Ecology (Q-90)	0.081	0.046	0.036	0.043	0.083	0.013	0.026	0.007	18	0.35	Plateau	2.0E-04	72%
2000	Social	0.755	0.294	0.145	0.068	0.078	0.059	0.051	0.005	13	0.68	Plateau	9.5E-04	36%
	Economic	0.732	0.317	0.144	0.047	0.064	0.056	0.025	0.002	4	0.89	Local maxima	3.1E-04	4%
	Mixed	0.525	0.272	0.104	0.016	0.040	0.050	0.025	0.001	4	0.79	Local maxima	1.8E-04	24%
	Ecology (B/P)	0.089	0.201	0.050	0.010	0.012	0.050	0.025	0.001	13	0.46	Plateau	2.3E-04	92%
	Ecology (Q-90)	0.060	0.088	0.058	0.052	0.113	0.023	0.020	0.008	20	1.00	Plateau	1.7E-04	20%

¹ Between-group linkage based on optimal fishing mortality per gear type (Euclidean² distance)

² Criteria: Global maximum ($p < 0.05$); Local maxima ($p > 0.05$; # peaks < 10); Plateau ($p > 0.05$; # peaks > 10)

³ C.V. of unstable runs $> 2.1 \cdot 10^{-4}$ (i.e. median value of all optimizations - based on variations in total system biomass during last 5 years)

Addressing parameter uncertainty

In setting the ORB goals for ecosystem rebuilding, we should consider the inherent uncertainty of the models, and especially the uncertainty surrounding historical parameter estimates. If our baseline assessment of the ancient ecosystems is incorrect, in structure or function, then the final ORB configuration that we should pursue by use of the optimal F vector may also vary.

In this chapter, I introduced an arbitrary caveat for ORB ecosystems that no functional group should deplete below 5% of the historic biomass, in order to prohibit extirpations. This simple qualification is just one example of the kind of provision that policy makers may wish to impose on the ecosystem goal for rebuilding. For example, regulators may demand that charismatic or culturally important species remain abundant despite other considerations. Restoring historical production potential may not be sufficient for political reasons; some degree of aesthetic resemblance to the historical period may be required in order to enlist public support for a restoration goal based on the past ecosystem.

Drawing again on the two key example ORB ecosystems from 1950, economic and biodiversity maxima (these will be revisited in Chapter 7), Fig. 6.12 shows that the optimal F vector, as calculated by the policy search routine, may drive some functional groups below an acceptable level of depletion when compared to the historic system. The optimal fishing solution maximized for economics should, by design, only deplete 3 groups below 40% of the historic 1950 biomass level and no groups should be depleted below 5% (Fig 6.12). Both of those expected outcomes are violated in Monte Carlo retrials. For example, 4% of simulations contain extirpations of infaunal carnivorous invertebrates below the 5% historical biomass threshold. In addition, adult lingcod is depleted below the target ORB biomass in 8% of runs and juvenile lingcod is depleted below the target in 6% of runs (Fig 6.12; left). This can happen because the policy search routine (at present) does not run forecasts using a Monte Carlo technique to consider the implications of parameter uncertainty. As a result, the ORB systems are optimal only under mean parameter values. This shortcoming needs to be addressed in upcoming EwE revisions. It would also be wise to include a subroutine that considers stochastic variations in system productivity.

For the biodiversity optimum, at least three model groups are prone to deplete beyond the level predicted by the ORB solution. The consequences of uncertain input data will be more critical if these unexpected casualties include charismatic or otherwise valuable species groups. The prescribed equilibrium level of fishing effort may need to be constrained under a precautionary approach to ensure that the ecosystem goal for rebuilding is acceptable to all stakeholders.

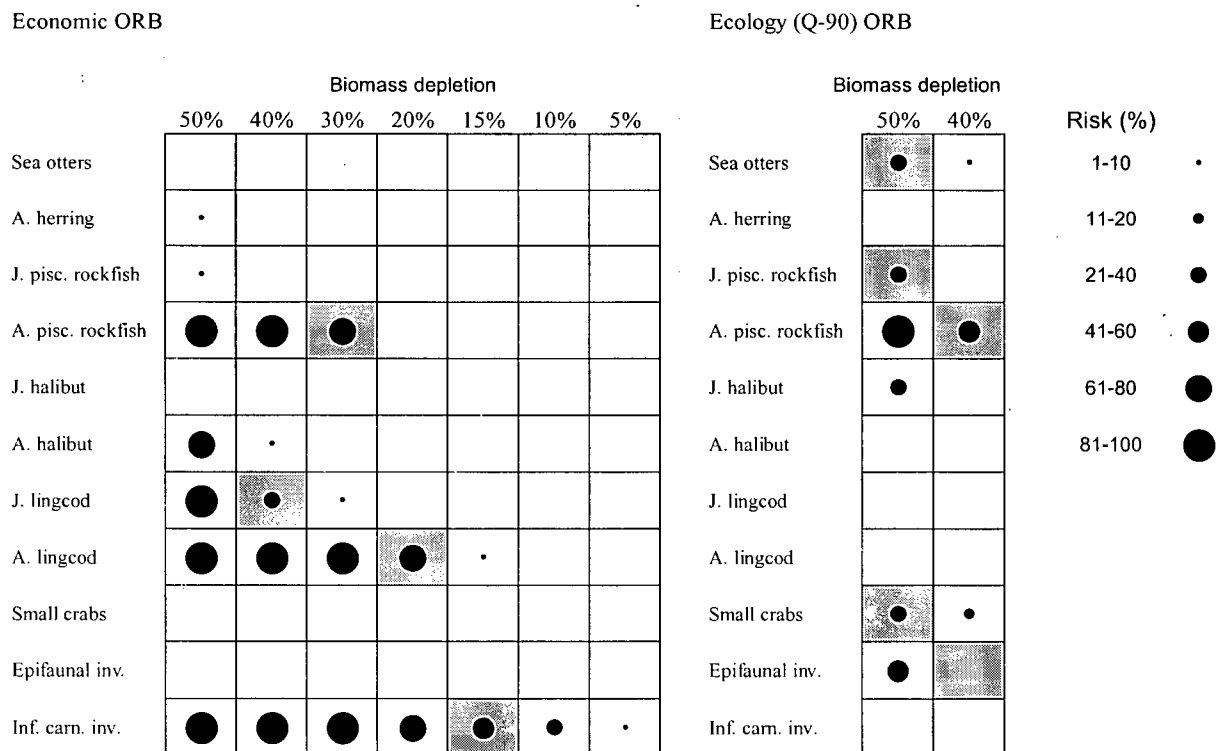
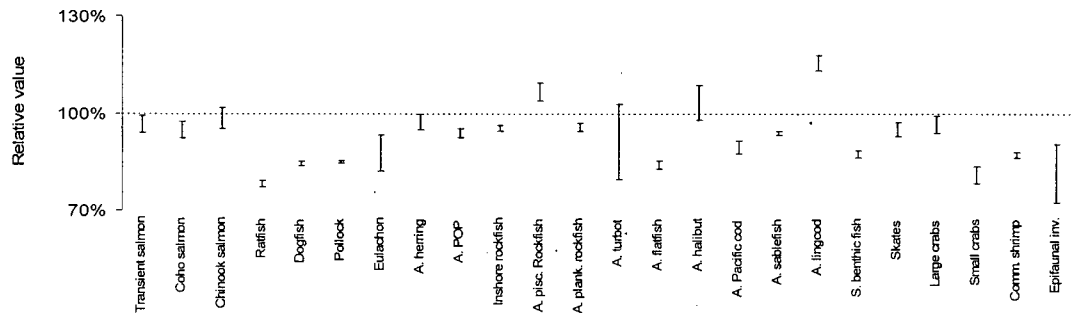


Figure 6.12 Biomass depletion risk of ORB solutions, considering Ecopath parameter uncertainty. Two ORB solutions are presented, the economic optimum and the biodiversity optimum, both based on the 1950 ecosystem. A Monte Carlo procedure draws random Ecopath parameter values for biomass (c.v. 10%) and production rate (c.v. 5%) from a uniform distribution. Grey cells indicate the level of depletion mandated by the ORB solution, compared to the historic 1950 system. Data uncertainty tends to result in the depletion of some functional groups by more than is expected under the optimal fleet-effort solution. Depletion risk is defined as the number of simulations (out of 100) that contain biomass depletions to any given level. Both objectives are prone to deplete groups beyond the level expected by the ORB solution. NB: The degree of parameter variation used is probably inadequate to assess the potential consequences of data uncertainty in the modeling historic systems (see discussion).

Economic ORB



Ecological (Q-90) ORB

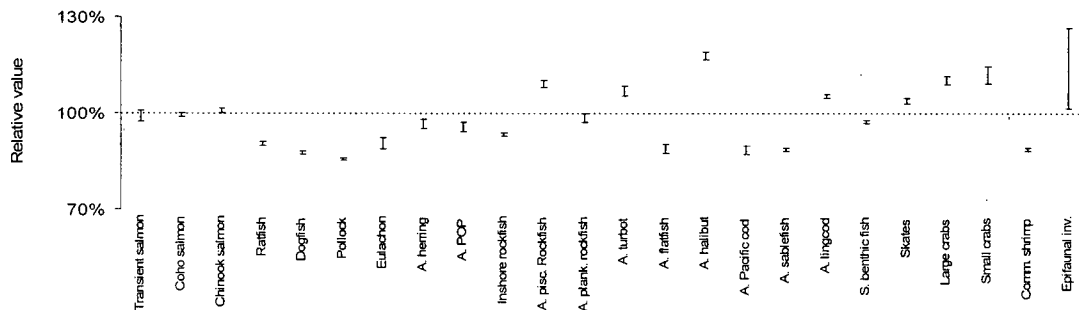


Figure 6.13 The effects of data uncertainty on ORB equilibrium values determined by Monte Carlo. Monte Carlo trials ($n = 100$) vary Ecopath biomass (c.v. 10%) and production rate (c.v. 5%). The centre line, at 100%, indicates the fishery value expected from example ORB solutions (see Appendix Table A.6.2.2). For most functional groups, the range of likely fishery value falls below the centre line, indicating that data uncertainty has the potential to compromise the projected ORB value. Less often, is the fishery value greater than we expected. The ecological solutions (bottom) tend to employ less fishing effort than the economic solutions (top), and so the range of possible fishery values varies less in the face of uncertainty. Error bars show mean ± 1 SD.

Fig. 6.13 assesses the economic consequences of uncertain data input on our optimal solutions. By varying the initial model structure through Monte Carlo resampling, deviations in fishery value per functional group are revealed. The economic solution tends to show a wider variation in ORB fishery values because the absolute catch rate is higher than under the biodiversity optimum. With some functional groups, the error range fails to intersect with the expected harvest value (centre line), indicating that data uncertainty is likely to have significant impact on

the profitability of fishing sectors exploiting those groups. When the error range falls below the centre line, as it does for the majority of groups, then the expected ORB fishery value will not be realized unless the models turn out to be very accurate. In other words, the F vector recommended by the policy search routine turns out to be suboptimal, because the baseline model was a poor representation of the ecosystem. The fishery value of other functional groups seems assured, regardless of data inaccuracies, because the error range falls above the centre line. For piscivorous rockfish, halibut, and in the case of the biodiversity optimum, some invertebrate groups, the fishery value predicted by the ORB solution is usually achieved or even improved upon by Monte Carlo retrials.

6.4 Discussion

The ORB concept

The policy search routine determines an optimal vector of fishing mortalities that generates the greatest sustainable wealth or harvest benefits when applied to historic systems. That optimal harvest plan will be realized, in practice, only following restoration efforts that have transformed the current ecosystem into the ORB configuration. Subsequently, the ORB system should be capable of sustaining those optimal catch rates. However, ORB, like MSY, relies on an equilibrium assumption, and the prediction of catch constancy comes with important caveats. Chapter 8 will more fully address the significance of the equilibrium assumption.

The actual fishing plan we use to approach the ORB ecosystem during restoration may or may not resemble the long-term harvest equilibrium determined in this chapter. It also depends where the actual biomasses are in relation to the ORB goal when the restoration starts. If a linear transformation of the ecosystem is required, then the very fishing mortalities determined here by the policy search routine should, over time, drive the present-day ecosystem directly towards the ORB goal. More complex restoration solutions, such as those requiring a non-linear or hysteretic state change, will probably not be accessible using simple equilibrium rate harvest strategies (see Chapter 7 section 7.4: Complex optimizations).

Comparison of ORB goals

In many respects, ORBs based on 1750 in northern BC emerge as the most desirable restoration goals. They promise the greatest financial and social rewards, and are able to maintain a level of biodiversity under any tested harvest objective that is superior to the base-state of the more recent ecosystems such as 1950 or 2000. However, since the pre-contact ecosystem is the least similar to the present day in overall terms of biomass, ORBs derived from this period represent more ambitious restoration targets. Commercially important functional groups must increase in biomass substantially; as such, these restoration targets would likely require painful conservation measures to achieve in practice.

The greatest economic incentive for restoring the pre-contact ecosystem lies in the re-establishment of abundant invertebrate populations. The profit potential offered by invertebrates in this ancient system may even be underestimated, since the 'epifaunal invertebrate' group is highly aggregated. Separating high value stocks such as Northern abalone (*Haliotis kamtschatkana*) would open additional avenues for the policy search routine, and may improve the profit potential for economic and social ORBs based on this most abundant system. Traditional commercial species also promise a sustainable fishery value that we are not accustomed to in modern times (Fig. 6.5; top). Chinook salmon, coho salmon and halibut stocks could sustain between 3 and 4 times the current annual value of landings. At the same time, current catch rates for some species in northern BC appear unsustainable (or at least sub-optimal) even for the immense resource base of this ancient ecosystem. This applies to several long-lived species such as inshore rockfish, Pacific ocean perch and sablefish. Any concern that this may raise regarding the current rate of exploitation is confirmed by Fig 6.5 (bottom). The current landed value of these species is much greater than the optimal value from the 2000 ecosystem, especially for Pacific ocean perch and sablefish (current annual landed value is between 50 and 100 times the long-term optimal rate).

For ORBs that focus on ecological maintenance, such as those determined under the ecological policy objectives, excessive depletion of the ancient system is undesirable. The policy search routine therefore manipulates the 1750 ecosystem to sustain a large recreational sector and

permits only a fraction of the overall fishing mortality used by the more exploitative optimums. In contrast, ORBs based on models of the more recent past (1950 and 2000) have only the recreational sector to rely upon for generating wealth and employment, even under the social and economic objectives, since further depletion of these systems will hurt economic performance in the long-run. This conclusion is evident from Fig. 6.8; the optimal fishing policy, even when directed by a social or economic objective, generally involves rebuilding biodiversity (the Q-90 term includes biomass) for the 1950 and 2000 ecosystems. Fig. 6.9 leads us to a similar conclusion; the 2000 ecosystem, especially, does not lose biodiversity under an economic optimization. The optimal policy therefore uses the recreational sector as a means to maintain an inflow of benefits at a minimum of disturbance to the ecosystem. It is used as the primary fishing instrument if depletion of the system is unacceptable, either because an ecological objective is in place, or because rebuilding biomass and diversity is prerequisite to satisfying an economic harvest objective.

This result is highly dependant on the price and jobs per unit catch value used for the recreational sector. Admittedly, the values in place are only rough estimates. They were not arrived at by any rigorous analysis, only expert opinion, and could definitely be improved upon. These parameters are difficult to set however, especially for the recreational sector where there can be much debate over appropriate figures. In this chapter, I have given sports fisheries catch a much higher value than the commercial fisheries for the same species, in order to take account of hotels and other service industries.

For ecological objectives using the more recent ecosystems, the recreational sector is still key. However, demersal fisheries, primarily for halibut, take on an increasingly important role. Halibut is relatively plentiful in the 1950 and 2000 models. Fisheries for this group, conducted by halibut longline and to a lesser extent, groundfish trawl, remain significant under any policy objective, but they become a proportionately greater fraction of the total take under the ecological objectives. One key observation is that the more recent ecosystems require that fishing effort is spread out evenly across fishing sectors for efficient management.

The ancient ecosystems, 1750 and 1900, tend to be depleted down to a more productive level by the social and economic objectives. However, ORBs based on the 1950 and 2000 ecosystems tend to bear smaller exploitation rates under the social and economic objectives, and the ecosystem typically builds biomass up to an optimally productive level. From the vantage point of 1950 or 2000 then, some degree of rebuilding is advocated by the optimization even from a purely economic perspective. In modern times, the ecosystem is depleted enough that the long-term goals of fisheries and conservation have converged, at least in regard to traditional target species. It should not come as a surprise. These findings indicate that this impoverished state has persisted for at least 50 years. Which parts of the ecosystem to rebuild, and which fisheries need to be maintained, are still matters of social priority.

In simulations, the 1950 model seems generally more responsive to changes in fishing mortality than the 1900 model. Commercial groups display a wider range of possible end-state biomass values following the optimal fishing programs. Because of this greater scope for growth, ORB fishing solutions based on 1950 tend to show greater potential for social and economic value than solutions based on 1900, despite a lower resource density. The fitting procedure used on the 1950 model was rigorous, however, and I suspect the discrepancy lies more with the 1900 model. If it is underestimating system responsiveness, it may be because baseline sources of fishing and predation mortality were under-represented in Ecopath. As those sources of mortality are removed through direct or indirect action of the optimal fishing policy, depressed groups could be expected to achieve a greater degree of biomass increase. Benefits expected from the 1900 system may therefore prove to be underestimated by future revisions to this work.

Response surface analysis

The geometry of the response surface will hold implications for management, and it should be considered when selecting restoration goals. The safest policy goal will lie in the centre of a plateau surface, where minor variations in fleet-effort will yield similar harvest benefits. A global maximum represents a more risky policy goal, because effort must be maintained close to optimal levels in order for maximum benefits to occur.

If multiple local maxima are present, then dissimilar fleet-effort configurations may yield similar harvest benefits. Distributional equity could vary greatly from peak to peak, however, as different fishing strategies are employed. One peak may hold a slight advantage over the rest, but a slim increase in Pareto efficiency is not a sufficient basis for a policy goal, unless the specific pattern of allocation is also considered. Each solution will favour different stakeholder groups and achieve different secondary management objectives. Multiple local maxima may therefore afford managers a choice. However, if two or more peaks are shown to provide similar benefits, then we can expand the evaluation criteria to include additional measures of utility, and reveal further distinctions between the optimal solutions.

One other possible response surface geometry overlooked by Fig. 6.11 is a mountain or valley chain. Under this form, certain core fisheries dominate the policy outcome. They need to be maintained close to their optimal effort levels in order for maximum benefits to occur, while less critical fisheries can vary from their optimal levels without impacting overall benefits. The prescribed fishing effort for these core fisheries is inflexible. Any deviation from the optimal solution, as may caused by shortcomings in management, technology or enforcement, will result in a major loss of benefits overall. For this reason, sectors with a poor track record of meeting management objectives should not form the 'core' fisheries. Instead, fisheries that are subject to a high degree of accountability and have a low degree of inter-annual variation should occupy the position central to the harvest policy.

Fleet structure

All the optimizations conducted in this chapter used the *lost valley* fishing fleet, a hypothetical set of contemporary and traditional gear types that minimize bycatch and habitat damage within technologically achievable limits. However, Ainsworth and Pitcher (2005b) tested the efficacy of various fleet designs in creating optimized ORB ecosystems, using preliminary versions of the northern BC models. They tested variations on the *lost valley* fleet, including one version without shrimp and groundfish trawlers, and one without a recreational sector in response to queries made by workshop participants as reported in Pitcher *et al.* (2002b).

They showed that ORB ecosystems can be structured to deliver similar harvest benefits, in dollar terms, regardless of the fleet design in use. Their findings contradicted earlier predictions made by Ainsworth *et al.* (2004). Since the policy search routine is at liberty to reduce or eliminate fishing effort on any gear type, it was expected that including additional gear types could only improve the dexterity of the policy search routine to manipulate the ecosystem, and increase the maximum fishery value. In fact, the rank order of value was consistent with the original hypothesis, although the results were not significant. The potential to supply jobs, however, did hinge critically on the fleet structure in use. The *lost valley* fleet consistently provided the most jobs. When trawlers were excluded from the optimal fishing plan, the ORB ecosystems provided fewer jobs regardless of the harvest objective or parent historical period. Without the recreational sector, employment potential was reduced even more.

The analysis conducted in this chapter did not factor in benthic habitat damage, which could impact juvenile fish survival and therefore the profitability of fisheries as well as ecosystem health. The economic and ecological consequences of excluding trawlers from the ORB solutions could be addressed with revisions to the current methodology. Revisions should begin with the use of mediating functions, to represent juvenile fish habitat contained in benthic structure¹³. Until the potentially major effects of benthic habitat damage can be properly assessed, there is little point in conducting a dedicated analysis to estimate strict trophodynamic effects associated with demersal and epibenthic gear activity. For this reason, the effects of fleet structure have not been closely considered so far, but I will revisit the issue of fleet design in Chapter 7 in application to rebuilding strategies.

System network indicators

Information theory (Ulanowicz, 1986; 1997) allows us to measure the level of organization present in an ecosystem. In a relatively disorganized system there is a prevalence of generalist feeders; this leads to a redundant trophic structure - energy may pass from one trophic level to

¹³ Some applications of mediation functions in EwE are described by Okey *et al.* 2004 (sea floor shading by plankton blooms), and Cox *et al.* 2002 (tunas mediating forage fish mortality caused by birds).

the next through multiple pathways. Such a system carries with it great 'strength in reserve' (Ulanowicz, 1986). That is, the system can fend off perturbations by rerouting trophic flow around a depleted hub. As the system becomes more complex, specialists come to fill every niche. The highly organized system makes more efficient use of available energy, supports a broader biota with longer trophic chains, but has lost trophic redundancy and therefore resilience to perturbation. Under cataclysm, the organized system may be reduced to a disorganized state. Likewise, chronic damage in a marine system caused from over-harvesting may be detected by observing the system's progression towards a less organized state. The disorder, or 'freedom' of the system is measured in terms of overhead (Φ), where an increase in Φ denotes simplification of the ecosystem.

Following the methodology suggested by Heymans (2004), Ainsworth *et al.* (*unpublished manuscript*¹⁴) analyzed various ORB ecosystems of northern BC using information theory. They found that ORBs based on the 1750 ecosystem tended to preserve a greater amount of the initial system order than the other periods (i.e., change in Φ was lower under fishing), but ORBs based on the other periods lost a proportionately greater amount of order. The pre-contact ecosystem therefore seemed able to support large harvests without losing trophic complexity, and it was expected that 1900 would show a similar quality. Instead, the second lowest change in Φ was seen in the 2000 system, followed by 1950 and then 1900. An examination of historical Newfoundland models showed a different result (Heymans *et al.*, *unpublished manuscript*¹⁵). In that case, the pre-contact ecosystem (c. 1450 AD) displayed the greatest change in system order upon harvesting, not the least. Again, there was no clear trend to link the historical periods. Overhead (Φ) is dependant on total system throughput (the sum of trophic flows). Since throughput is heavily influenced by primary production and other climate factors (J.J. Heymans, UBC Fisheries Centre, *pers. comm.*), it may be difficult to apply this measure to evaluate the

¹⁴ Ainsworth, C., J.J. Heymans, W. Cheung and T.J. Pitcher. 2003. Evaluating Ecosystem Restoration Goals and Sustainable Harvest Strategies in Northern BC. *Unpublished manuscript*. Available from author: [c.ainsworth@fisheries.ubc.ca].

¹⁵ J.J. Heymans, C. Ainsworth, W. Cheung and T.J. Pitcher. 2003. Newfoundland BTF Results. Available: [s.heymans@fisheries.ubc.ca].

response of ancient ecosystems to fishing unless the productivity regime can be more carefully accounted for.

Metrics of system information have therefore shown inconclusive results so far in BTF applications, although the treatment has been simplistic. However, additional work by Heymans (2003) has applied information theory to interpret ecosystem changes described by EwE models; they met with more sensible answers. Also, recent additions to EwE software now allow dynamic monitoring of system information content (Christensen and Walters, 2004a), which could facilitate a more thorough analysis of this type. The approach is still under development, but work is underway to use indices of information theory to interpret dynamic responses in EwE models (e.g., Pinnegar *et al.*, 2005).

What is an appropriate restoration target?

This chapter demonstrates a new method to estimate the value of historic ecosystems as restoration targets, and it offers a new quantitative ecosystem-based goal for marine restoration, the ORB ecosystem. Although the benefits of restoration can be presented to policy makers using helpful indices, the decision on what ecosystem to restore will ultimately require a value judgment. A trade-off between socioeconomic and ecological priorities is inevitable regardless of the historical period chosen.

A restoration goal based on the pre-contact ecosystem could sustain a fishing industry worth 5.5 times its current value (social objective: Appendix Table A.6.2.2) while maintaining a higher level of biodiversity than we are currently familiar with (Fig. 6.8). This conclusion should make the point clearly that myopic exploitation of resources has hurt the fishing industry and degraded the ecology of northern BC over the last 250 years. It should also inspire us, because the ecosystem seems able to provide enormous sustainable harvest benefits when fisheries are designed optimally, and make use of responsible gear types. Climate, biology, technology and economics will primarily determine whether a goal such as the pre-contact ecosystem is realistic, and whether anthropogenic damage is reversible over the course of decades or centuries. For our

part, as custodians of the marine system, we may need to settle realistically for a more modest restoration goal. Yet even plans based on 1950 improve greatly over the present-day situation.

As the next chapter (7) will demonstrate, restoration plans to that goal may be possible, even practical. Chapter 7 will investigate strategies to achieve restoration to some example ORB ecosystems based on 1950, and it will estimate costs and benefits associated with ecosystem restoration while identifying potential barriers.

7 ACHIEVING RESTORATION

To keep every cog and wheel is the first rule of intelligent tinkering

Aldo Leopold

Sand County Almanac (1953)

7.1 Introduction

Having established whole ecosystem goals for restoration in Chapter 6, I now introduce a procedure to calculate optimal fishing strategies that would restore an ecosystem to a specific configuration of biomass per functional group. Beginning with the present-day ecosystem, optimal equilibrium-level fishing mortalities are applied to each gear type that will manipulate the biomasses of functional groups over time, through direct and indirect influences of fishing, to resemble the goal ecosystem. Optimal fishing mortalities are determined using a new algorithm developed for Ecosim's policy search routine.

A new objective function called "*specific biomass*" (SB) is described here for use with Ecosim's policy search routine (Christensen and Walters, 2004b); the algorithm has been incorporated into V5.1 Ecosim software, along with additional refinements made for the BTF approach (available from author). Two key parameters, the unit of improvement and the marginal improvement valuation model, are used to guide the optimization and establish criteria for ecosystem improvement towards the goal ecosystem. A new output form allows the user to monitor the progress of the optimization towards achieving the goal ecosystem, and displays ecological consequences of the rebuilding plan. After demonstrating the functionality of the new SB routine and verifying its policy outcomes, I draft restoration scenarios that would transform the present-day ecosystem into an ORB-structured ecosystem.

The ORB ecosystem (optimal restorable biomass; Chapter 6) is a practical target for restoration based on historic ecosystems, where group biomass is tailored to support maximum sustainable harvest benefits from the ecosystem overall. Restoration plans targeting the economic ORB (based on 1950, and optimized to deliver maximum fisheries profit) are compared with plans targeting the 1950 historic ecosystem, to demonstrate ORB's superior ability to support profitable fisheries. Restoration plans targeting the 1950 ORB ecosystem optimized for biodiversity are also presented, and the properties of the SB algorithm are evaluated with respect to the restoration of species biodiversity. These example ORB ecosystems were developed in Chapter 6. Finally, a cost-benefit analysis evaluates the economic pay-offs from particular whole-ecosystem restoration trajectories, where the intensity and duration of the restoration program is varied. The internal rate of return of restoration plans is calculated and compared with bank interest. Although motivation to restore an ecosystem will likely include non-monetary considerations, a conservation agenda may be easier to implement if some potential for economic return can be demonstrated.

The need for a new rebuilding algorithm

Chapter 6 describes the policy search routine from Christensen *et al.* (2004a), and the economic, social and ecological criteria that compose the objective function for the numerical optimization. The fourth criterion listed in eq. 6.1, mandated rebuilding, was originally designed to help the user determine what management action leads to the quickest recovery of depleted groups. That information could be useful to policy makers in the case of threatened species, for example.

I used the existing EwE mandated rebuilding routine in preliminary work to generate an ORB-structured ecosystem, but found that it was not adequate to achieve restoration when multiple groups were mandated for recovery. Nor was there adequate control over the policy outcome. There were two main problems. First, the algorithm recognized improvement only in the positive direction, as groups increased in biomass towards their target levels. There was no benefit tendered to the objective function if a group decreased in size in order to approach its target. However, in all historical and ORB ecosystems evaluated here, there are at least some ecosystem components below the present-day level of abundance. Moreover, selective biomass

declines may be desirable in restoration scenarios to facilitate growth in other functional groups through direct or indirect trophic interactions. Secondly, the routine assumed that restoration was equally desirable in all functional groups. This does not pose a problem if one is concerned about restoring only a small number of groups. However, for whole ecosystem restoration this severely restricts the types of policy outcomes.

I found that results were always conservative using the existing mandated rebuilding objective function - meaning that the optimal fishing mortalities were never far from the gear type's baseline level. The policy search routine tried to improve a maximum number of functional groups. Consequently, it would not allow large reductions in fishing mortality to rebuild depleted groups, if the restoration plan upset an equal or greater number of groups that would otherwise have been easy to restore (e.g., groups under high levels of baseline fishing mortality, highly volatile groups or groups already close to the biomass target). Unfortunately, in order to restore large and severely depleted groups, as commercial groups often are, it may be necessary to ignore other ecosystem components and permit their biomass to increase or decrease far from the target level. A more sophisticated definition for the rebuilding objective function is therefore introduced here that can determine the fishing pattern needed to manipulate the ecosystem into a precise configuration and execute judgments, according to user-defined criteria, as to what functional groups deserve overriding attention.

7.2 Methods

Specific biomass objective function

The SB objective function is designed to make any prioritization between groups explicit, and to give the user more control over the trade-offs inherent in whole ecosystem restoration. The new function is used to estimate optimal fishing mortalities that will adjust group biomasses to specific levels. The SB objective function is summed for each mandated functional group (i) and time step (t), so that the term maximized by the policy search routine, \overline{SB} , is represented in eq. 7.1.

$$\overline{SB} = \sum_{t=1}^T \sum_{i=1}^n SB_{it} \quad \text{Equation 7.1}$$

Here, n is equal to the total number of mandated functional groups and T is the total number of time steps (by default in Ecosim, the number of simulation years times twelve to represent monthly time steps). The algorithm used to compute the new SB function essentially governs the trade-off between restoring biomass of a large number of easily recoverable groups, versus a smaller number of critical 'problem' groups. Those are, groups that resist biomass change in either the positive direction (as with slow growing, or heavily depleted groups) or the negative direction (as with groups not subject to direct harvests). Based on Ainsworth and Pitcher (*in review*), the SB function uses two main parameters to define the desired trade-off: the measure of ecosystem 'improvement' towards goal biomass, and the choice of model used to calculate the value of marginal improvement.

At each time step of a draft harvest policy, the SB algorithm evaluates the biomass of every functional group that has been flagged by the user for rebuilding. The difference between starting biomass, when $t = 1$, and the goal biomass for restoration defines the potential scope for improvement. At each time step, an internal term (θ) is calculated based on group biomass. It represents the proximity of the group's biomass to the rebuilding goal, where θ is a unitless multiple of the initial start-goal biomass difference. Initially, $\theta = 0$ if starting biomass is less than goal and $\theta = 2$ if starting biomass is greater than goal. If group biomass in time step (t) has moved closer to goal than the starting position, θ will approach 1. This happens through population growth if $\theta_{start} = 0$, or through attenuation if $\theta_{start} = 2$. When $\theta = 1$, the group biomass has been exactly restored to the goal level, and the functional group contributes its maximum to the SB term.

Proximity to goal (θ) is calculated as in eq. 7.2,

$$\theta = \begin{cases} \frac{B_{current} - B_{start}}{B_{goal} - B_{start}} & \text{if } B_{start} < B_{goal} \\ 2 - \left[\frac{B_{current} - B_{start}}{B_{goal} - B_{start}} \right] & \text{if } B_{start} > B_{goal} \end{cases} \quad \text{Equation 7.2}$$

$B_{current}$ is the functional group's biomass in time step t , B_{goal} is the goal biomass and B_{start} is baseline biomass when $t = 0$.

Each group's contribution to the SB function is calculated based on θ . Groups will contribute their maximum to the objective function when $\theta = 1$. The more groups that approach their goal biomass, the larger the overall objective function will become. The contribution of each group can be weighed as a function of the group's absolute or relative biomass change towards target using the 'unit of improvement' setting, or as a function of its current distance to target using one of three marginal improvement valuation models.

Unit of improvement

If we define the unit of ecosystem 'improvement' towards the target configuration strictly as a change in group biomass (hereafter called the biomass criterion), then in depleted ecosystems the policy search routine will tend to advocate fishing strategies that greatly reduce fishing mortality from baseline levels. The optimization will try to rebuild severely depleted groups, especially groups that exhibit a large potential scope for growth (i.e., where scope for growth is related to the absolute start-goal biomass difference and the amount of production available for growth). Groups with a large potential scope for growth will include massive commercial groups that are subject to a high level of baseline fishing mortality. Removing fishing mortality will help satisfy the biomass criterion for severely depressed groups. Small biomass groups may be neglected under the biomass criterion, however, and drift further away from their target biomass.

If we define the unit of ecosystem ‘improvement’ in terms of a percent change towards target (hereafter called the percent criterion), then the optimization procedure will be able to affect recovery for the greatest number of groups by applying only minor changes to the existing fleet-effort structure. Every ecosystem evaluated here as a candidate restoration target contains at least some functional groups which are similar in abundance to their present-day counterparts (this is true of historical ecosystems, ORB ecosystems and probably most potential ecosystem goals). With these functional groups, 100% improvement is easily achieved by increasing or decreasing biomass by only a slight amount (or none at all). If these groups constitute a major portion of the ecosystem, then drastic changes in the fleet-effort structure, as would be required to rebuild commercial productivity for example, would introduce unwanted secondary trophic effects and perhaps incidental captures. If these effects disturb groups that were already close to their target biomass, then the overall objective function SB could suffer. Fishing strategies designed to restore ecosystem productivity will be conservative under the percent criterion, optimal fishing mortalities will remain close to baseline levels and most of the restoration improvement will be realized in groups that were already close to their target in absolute terms.

When the unit of improvement is taken as percent progress towards target, proximity to target (θ) is passed directly to the marginal improvement model as $\theta_{percent}$ for each functional group. If the unit of improvement is biomass, then $\theta_{biomass}$ is first calculated as in eq. 7.3,

$$\theta_{biomass} = \theta \cdot |B_{goal} - B_{start}| \quad \text{Equation 7.3}$$

Between these extremes, the new algorithm can also quantify the SB objective function based on a combination of the biomass and percent criteria, weighted in relative proportion so that the combined term ($\theta_{combined}$) is determined by eq. 7.4,

$$\theta_{combined} = X \cdot \theta_{biomass} + (1 - X) \cdot \theta_{percent} \quad \text{Equation 7.4}$$

X is a weighting factor between 0 and 1, accessed by a slider bar on the policy search input form [frmOptF.frm] (Fig. 7.1). There are eleven positions on the slider bar (left to right, $X = 0, 0.1, 0.2 \dots 1$), where 0.5 assigns equal weighting to the biomass and percent criteria. If the unit of ecosystem improvement is biomass, then the maximum value achievable by the objective function will be equal to the sum of all mandated (goal) biomass values. If the unit of improvement is percent, then the most each group can contribute is 1, and the maximum value achievable by the objective function is n , the total number of mandated functional groups. Table 7.1 details the new policy search controls.

Figure 7.1 Controls added to Ecosim's policy search interface for SB algorithm. EwE form is [frmOptF.frm]; see Table 7.1 for description of controls.

Table 7.1 Available settings for the *specific biomass rebuilding objective function*. EwE: [frmOptF.frm].

Control	Usage
Restore ORB	Activates new restoration objective function.
No Extinctions	Prohibits extinctions below threshold biomass (default 5% of initial biomass. Activates frame "Extinction threshold")
Ecol is Q90	Q90 biodiversity index replaces ecosystem maturity (B/P) as ecological optimization criterion.
ORB results	Enables dynamic progress display form (see Fig. 7.4) [frmORBresults.frm]
SByear	Rebuilding function is summed over last x years (counted from end). Default x = total simulation years, all years included.
Marginal improvement value	Selects valuation model for marginal improvement: linear, quadratic or gamma.
Unit of improvement	Slider bar sets relative weightings of 'percent' and 'biomass' restoration criteria in combined objective function. Left (%); centre (equal weighting); right (biomass).
Time preference	Introduces 'discounting' term into the objective function making immediate improvement more valuable; it produces 'fast-track' restoration plans. Fixed discount rate is 10%. This feature is not related to the economic evaluation.

Marginal improvement valuation model

The marginal improvement valuation model allows the user to weigh the relative contribution of a functional group to the objective function SB according to the group's current distance from the goal biomass. Under the linear valuation model (Fig. 7.2a), all functional groups are weighted equally in the calculation regardless of their distance to target. For each mandated group and time step, the SB function is represented by eq. 7.5,

$$SB_{lin} = \begin{cases} \theta & \text{if } B_{start} < B_{goal} \\ -\theta + 2 & \text{if } B_{start} > B_{goal} \end{cases} \quad \text{Equation 7.5}$$

The linear model will make whatever trade-offs are necessary between groups to achieve the greatest overall reduction in biomass residuals versus the desired ecosystem configuration (i.e., minimizing $\sum_i |\theta_i - 1|$ for each functional group i and time step t).

Under the quadratic model (Fig. 7.2b), the greatest marginal increase in the objective function occurs when groups first begin to move towards their goal biomass. More groups will improve in the optimal fishing policies than under the linear model, but the average improvement in the proximity function θ will be lower. The quadratic model is somewhat precautionary because the objective function decreases rapidly as group biomasses drift away from their goals in either positive or negative direction. There will be a strong incentive, therefore, for the policy search to include as many groups as possible in the rebuilding plan. The quadratic model is given by eq. 7.6,

$$SB_{quad} = -\theta^2 + 2\theta \quad \text{Equation 7.6}$$

The gamma valuation model (Fig. 7.2c) is the most precautionary. It is asymmetric, so improvement from functional group biomass growth contributes more to the objective function than improvement from group biomass decline. Optimal policies will more often overshoot target biomasses than fall short. The gamma model is represented by eq. 7.7,

$$SB_{gam} = \phi \cdot \frac{\left(\frac{\theta}{\beta}\right)^{\gamma-1} \cdot e^{-\left(\frac{\theta}{\beta}\right)}}{\beta \Gamma(\gamma)} \quad \text{where: } \Gamma(\gamma) = \int_0^{\infty} t^{\gamma-1} e^{-t} dt \quad \text{Equation 7.7}$$

Phi (ϕ) is a scaling term for the Y-axis, γ is the shape parameter and β is a scaling term for the gamma function (Γ). In pilot work, I tested a simpler asymmetric relationship, lognormal, but the complex gamma model provided more flexibility to define the shape of the valuation curve. It allowed me to standardize the model response into something arbitrary but interpretable. Parameters are set so that the gamma objective function is worth 0.5 when proximity to goal (θ) is equal to 0.5 or 2.0. In other words, a functional group will contribute the same to the SB objective function when it is 50% short of its goal biomass as when it is 100% in excess of its goal biomass – and that value corresponds to 50% of the maximum possible contribution for that group towards the objective function. Parameters used to set this relationship were $\phi = 1.74$, $\gamma = 3.21$ and $\beta = 0.45$.

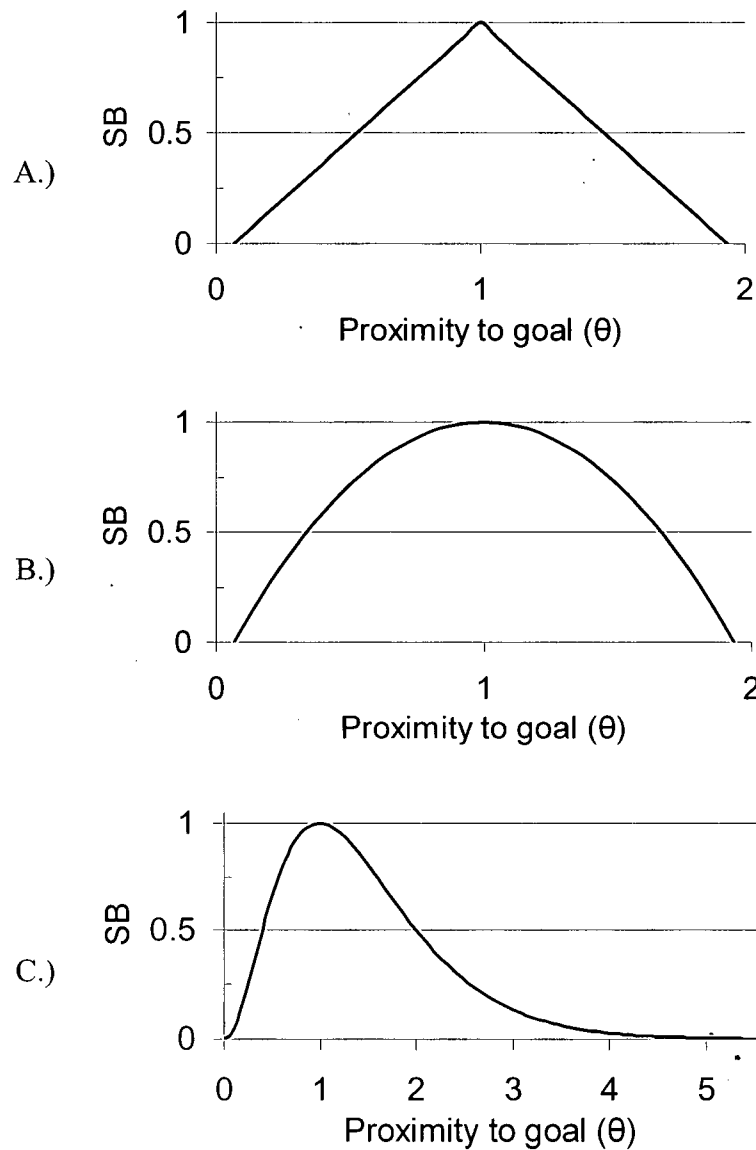


Figure 7.2 Three models describing marginal improvement in SB function as group biomass approaches goal. The initial ecosystem condition is taken as $\theta = 0$ if start biomass is less than goal, or $\theta = 2$ if start biomass is greater than goal. As the functional group approaches its goal biomass, θ approaches 1. A.) Linear model weighs a unit of improvement the same, regardless of the current distance from goal; B.) Quadratic model gives the greatest improvement to objective function when groups first begin to move towards target; C.) Asymmetric gamma model is precautionary; biomass increase towards target is more valuable to the objective function than biomass decrease.

Caveats

For groups already close to target

With either unit of improvement, biomass or percent, θ is measured with regards to the initial biomass differential between start and goal. When the start and goal biomasses are quite similar, then the absolute biomass differential represented by the multiple (θ) will be small. In that case, even minor unfavorable changes in group biomass could constitute many multiples of the initial biomass difference between start and goal, leading to large negative values of θ . This would be avoided by the policy search routine unless favorable changes in other functional groups could make up the difference. Moreover, θ will be undefined when $B_{\text{start}} = B_{\text{goal}}$. I therefore introduced a caveat. The absolute reference frame that defines θ cannot be smaller than 10% of the goal biomass. In cases where B_{start} is already within 10% B_{goal} , the initial value of θ in time step $t = 1$ will be greater than zero (if $B_{\text{start}} < B_{\text{goal}}$) or less than 2 (if $B_{\text{start}} > B_{\text{goal}}$). The value B_{start}' then substitutes for B_{start} in equation 7.3, where B_{start}' is defined as in eq. 7.8,

$$B_{\text{start}}' = \begin{cases} 0.9 \cdot B_{\text{goal}} & \text{if } |B_{\text{goal}} - B_{\text{start}}| < 0.1 \cdot B_{\text{goal}} \\ B_{\text{start}} & \text{otherwise,} \end{cases} \quad \text{Equation 7.8}$$

Under this caveat, groups that begin exactly at their goal biomass will have an initial proximity of $\theta = 1$, and in the first time step of the simulation they will contribute their maximum value to the objective function. The shape of the marginal improvement model will therefore be constrained as in Fig. 7.3.

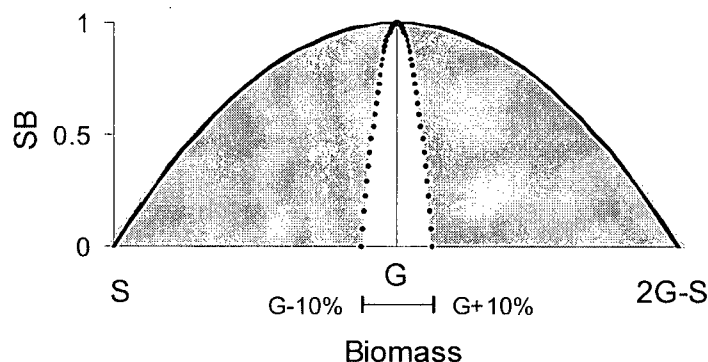


Figure 7.3 Constrained marginal improvement model (MIM). Quadratic form is presented. Solid curve shows the MIM when starting biomass (S) is less than 10% of goal biomass (G). The dotted curve shows the most leptokurtic form of the MIM allowable, when the absolute start-goal biomass differential is less than 10% of goal biomass. The shaded area shows the total range that the MIM curve can occupy. The subject functional group will contribute a negative value to the SB term when current biomass is less than S or greater than $(2G-S)$.

Negative values of specific biomass term

Throughout a harvest simulation, if the biomass of any particular functional group moves further away from its target than the baseline condition, then θ becomes less than 0 or greater than 2; and the objective function SB becomes negative. For the linear and quadratic valuation models, this can happen in either the negative or positive direction (i.e., when current biomass falls further from goal than the baseline condition (if $B_{\text{start}} < B_{\text{goal}}$), or when current biomass overshoots the goal beyond the baseline condition (if $B_{\text{start}} > B_{\text{goal}}$). The gamma model will only produce a negative SB when biomass falls further away from the goal than the baseline condition in the negative direction (i.e., through group depletions). Of course, the cumulative \overline{SB} term (summed over all functional groups) can only increase as the target ecosystem is approached.

An EwE model that is constructed to be steady state (i.e., where all group biomass trajectories are a flat-line under baseline fishing mortalities) will necessarily have a \overline{SB} value of zero on the

first iteration. It is possible, however, that the first iteration will carry a negative value of \overline{SB} if baseline fishing mortalities drive the system away from the goal configuration; this can only happen if biomass accumulations are included in Ecopath. If that is the case, then longer simulations (i.e., including more simulation years) will assume a greater negative value of \overline{SB} than short simulations - the system having had more time to be disturbed by baseline fishing levels. To accommodate negative numbers in the optimization a minor change had to be made to the existing Ecosim code in module [Fletch.bas]. Originally, the term maximized by the optimization was equivalent to $\overline{SB}_{current}/\overline{SB}_{baseline}$; but increasing this quotient represents improvement only if $\overline{SB}_{baseline}$ is positive. In the new revision, the term remains unchanged if $\overline{SB}_{baseline}$ is positive, but if $\overline{SB}_{baseline}$ is negative then the term becomes $\overline{SB}_{current}/|\overline{SB}_{baseline}| + 2$. In this way, the \overline{SB} term maximized by the policy search routine has been standardized so that it will always equal 1 upon the baseline iteration, and improvement towards the target ecosystem configuration will correspond to an increase in the term.

Additional parameters for specific biomass function

Summation year

The new objective function can use several auxiliary parameters. The parameter “SByear” shown in Fig. 7.1 allows the user to specify the number of years over which \overline{SB} is summed. By default, \overline{SB} is summed over every year in the simulation. The default value appearing in the text box is equal to the total number of simulation years. The user can specify that fewer years be considered in the calculation of \overline{SB} (eq. 7.1). The value in the text box is counted backwards from the final simulation year, so that entering 5 in the “SByear” text box will sum \overline{SB} over the last five simulation years only. An optimal fishing plan identified under this setting will have no incentive to achieve restoration quickly, but the final agreement with target may be slightly better than under default settings. Moreover, unstable and cyclic dynamics will not be penalized as long as the end-state ecosystem resembles the target biomass configuration. Under this setting, the optimization will register great improvements in the objective function even if the end-state ecosystem structure is unstable, and agreement with the goal biomass vector is

momentary. I recommend keeping the default value. This will effect a quicker restoration, and the policy search will have incentive to produce stable biomass trajectories.

Fast track

An option is included in the SB interface to permit a faster recovery towards target. By checking “Done fast” under time preference, near-term restoration benefits are weighed more heavily than distant future benefits in calculation of the term \overline{SB} . Under this ‘fast-track’ option, eq. 7.1 is modified to include a standard discounting model, as shown in eq. 7.9:

$$\overline{SB} = \sum_{t=1}^T \sum_{i=1}^n \left[\frac{SB_{it}}{(1+\delta)^t} \right] \quad \text{Equation 7.9}$$

The discount rate δ represents our preference for early restoration benefits. It is fixed at 10%. This modification can be used to determine what management actions are needed to approach the target ecosystem quickly. As such, much of the benefit to the objective function will result in the fishing down of groups to target levels, rather than through cultivation and re-growth. Note that the “Done right” option is selected by default.

Extinction threshold

The text input box under “Extinction threshold” allows the user to specify a minimum level of functional group biomass (as a percent of baseline biomass) below which it is considered managerially unacceptable for the optimal harvest plan. At each time step of the simulation, the corresponding code evaluates whether the biomass of any functional group has fallen below this relative threshold. If so, the value of the objective function is set to zero for all simulation years, and the harvest plan is discarded from consideration. This feature is functional with the SB objective function as well as all previous objective functions (i.e., economic, social, ecological, mandated rebuilding and portfolio-log utility). This feature may be most useful for constraining exploitative harvest plans optimized, for example, under the economic and social criteria for harvest benefits. However, it is also useful for constraining rebuilding plans that do not

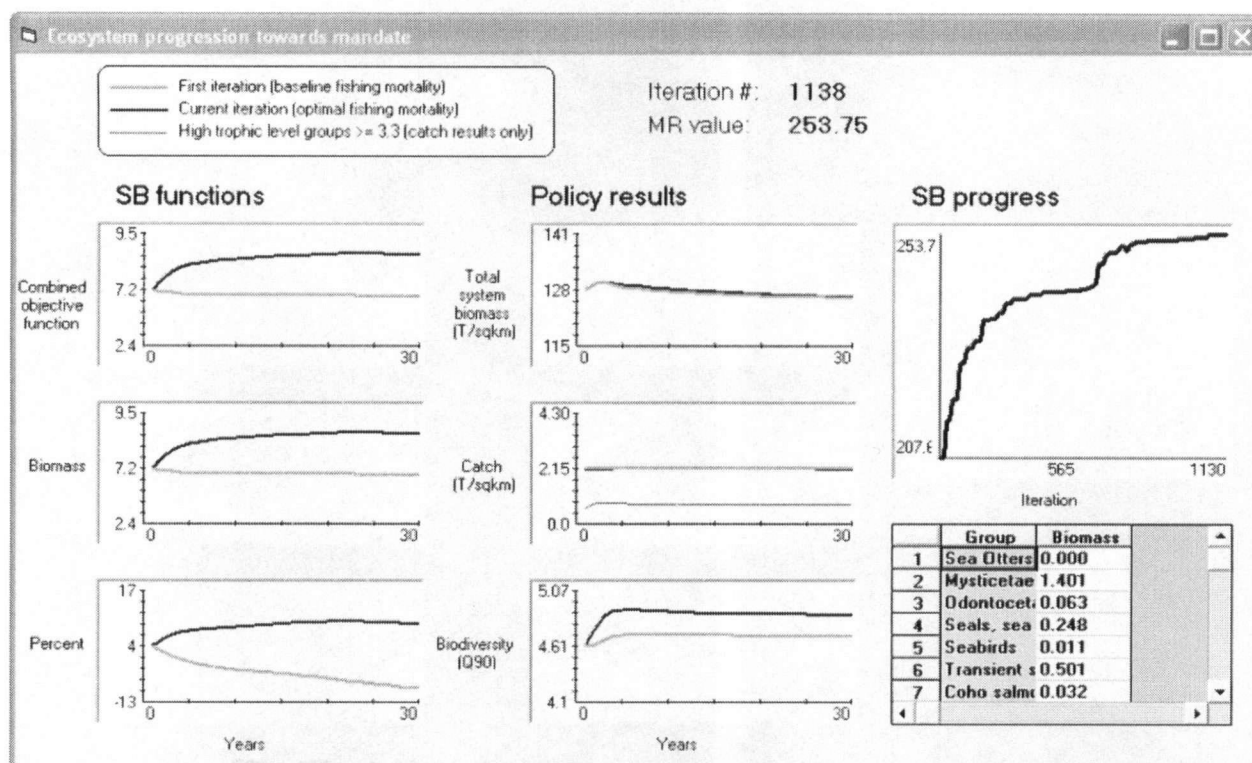


Figure 7.4 Dynamic progress display form to monitor rebuilding success of the SB algorithm. Form is enabled by 'ORB Results' check box. Figures under column heading 'SB functions' show the values of the SB term for each simulation year using biomass and percent unit of improvement; 'Combined objective function' combines these terms using weightings specified by the user. The green lines show the result of the first iteration (i.e., baseline fishing mortalities) and the blue lines show the best solution discovered by the optimization. Graphs under "Policy results" display the effects on the ecosystem in terms of total ecosystem biomass, catch and biodiversity (Q-90). 'SB progress' shows \overline{SB} , the cumulative objective function summed over all years and functional groups. The value of \overline{SB} is plotted against the iteration number; it corresponds to the area under the blue curve in the combined function graph (top left). The spreadsheet in the lower right provides the end-state ecosystem structure (biomass, $t \cdot km^{-2}$); it is updated every ten iterations.

explicitly include all functional groups (e.g., rebuilding plans concerned only with commercial groups). All harvest plans evaluated in this chapter employ a 5% extinction cut-off threshold, the default in the new interface.

Dynamic progress display form

Checking the "ORB Results" check box on the input form (Fig. 7.1) enables the dynamic progress display form during optimizations (Fig. 7.4). Readouts appear that allow the user to monitor the progress of the objective function throughout the optimization, and the ecological

consequences of the optimal policy in terms of system biomass, catch and biodiversity. The *SB* objective function is displayed for each simulation year, and the cumulative term \overline{SB} (summed over all simulation years and functional groups) is displayed each iteration. A spreadsheet is accessible in the lower-right that updates every 10 iterations to provide the end-state ecosystem biomass configuration. Dynamic progress display form is [frmORBresults.frm].

Maximum dexterity fleet

Larkin (1996) said, “Existing fleets are a blunt instrument for fine tuning the relative abundance of species”. Where gear types catch multiple species, this limits our ability to manipulate the ecosystem. We cannot easily choose to rebuild one weak species, while increasing catch on a sympatric species using unselective gear. If, however, each gear type were to catch one and only one target species (through application of bycatch reducing devices and techniques, for example), this would increase our ability to selectively fish the ecosystem and approach the configuration that permits maximum sustainable benefits. Any limitation imposed by the unselective nature of gear can only cause the restoration policy to fall short of the ideal ecosystem configuration.

Achieving perfect gear selectivity would be impossible in practice, but it is possible to simulate and there is heuristic benefit in testing this unrealistic gear structure. If under these ideal fishing conditions we can reshape the ecosystem into the desired configuration, then we can be confident first, that the restoration algorithm is operating correctly, and second, we can determine the extent to which ecosystem reconstruction is ecological feasible. Because this special fleet configuration eliminates limitations imposed by the technology of fishing, the restoration policies represent ‘best case’ scenarios limited only by the ecology of the ecosystem. This allows us to set the benchmark for restoration and evaluate the effectiveness of candidate fleets to be used for the task of rebuilding.

All diagnostic procedures in this chapter used to verify behaviour of the *SB* algorithm apply this idealized fleet structure, called the ‘maximum dexterity’ fleet, or *maxdex*. The *maxdex* fleet uses 27 gear types to capture 29 functional groups; fisheries for halibut and lingcod were said to catch

juveniles and adults. Catch per gear type is provided in Appendix Table A7.1.1. Total catch for each group remains unchanged from the 2000 northern BC model (including both directed catch and discards; see Appendix Tables A5.1.4 and A5.1.5). A new fishery, seal cull, is added to give the policy search routine maximum capability to manipulate the ecosystem. A small catch for seals and sea lions was set (5% of biomass); only a non-zero number for seal and sea lion catch is required to open up this avenue for manipulation to the policy search routine. Year zero was included in all rebuilding optimizations, so the actual amount of seal and sea lion catch set in the base model is irrelevant. Provided, that is, that the addition of the new fishery did not cause the EE of the group to exceed 1, which would have then require additional balancing to correct.

Cost-benefit analysis of restoration

Following the methodology of Ainsworth and Pitcher (*in review*), an economic analysis of restoration evaluates the potential value of rebuilding. I restricted the analysis to restoration plans targeting the economic ORB, since it proved to be a competitive economic goal when compared to the historical 1950 ecosystem (see Fig. 7.15). Fig. 7.5 shows the basis of the cost-benefit analysis.

The stream of benefits from the restoration plan can be divided into three stages. In the rebuilding phase (α), optimal fishing mortalities reduce harvests and allow the system to rebuild. The profit sacrificed from baseline is considered the 'cost' of restoration. As the ecosystem rebuilds, the small 'medicinal' harvest rate is rewarded with a slight increase in catch as commercial groups grow in abundance. The second phase is transitional (β); fishing effort is adjusted to cancel biomass accumulations and establish new group-biomass equilibriums. In the third phase (γ), final equilibrium harvests are maintained until the end year. Since the standing level of commercial biomass has increased, fisheries can draw more yield and/or profit sustainably from the restored system than from the original system. The difference is the 'benefit' of restoration. Note that I have assumed completely malleable capital in that there is no penalty associated with fleet restructuring or decommissioning of fishing effort.

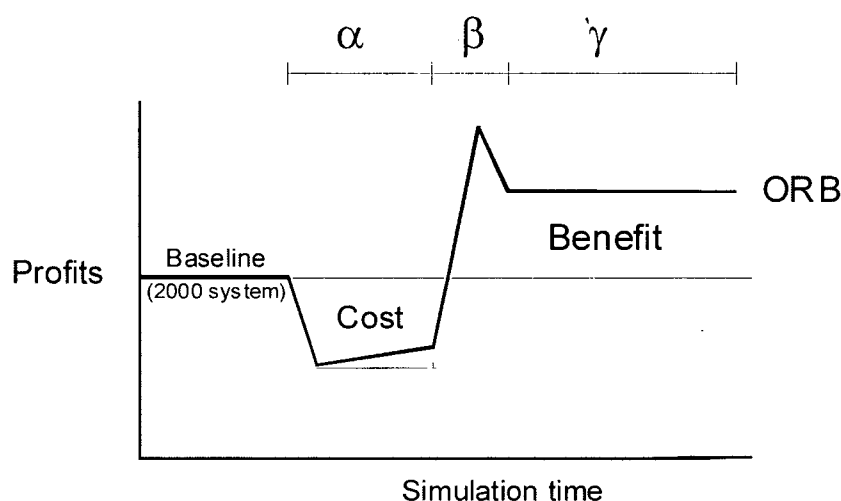


Figure 7.5 Conceptual diagram showing cost-benefit analysis. Restoration plan consists of three phases: Optimal fishing for ecosystem rebuilding (rebuilding phase) (α); readjustment of fishing effort to establish new profit equilibrium and cancel biomass accumulations (settling phase) (β); sustainable fishing at new profit equilibrium (equilibrium harvest phase) (γ). Costs and benefits of restoration are taken in relation to forecasted profits from the 2000 system (baseline) assuming a status quo level of profit.

Applying optimal fishing mortalities during the rebuilding phase generates a stream of profits, biodiversity and other indices analyzed in this chapter. They are obtained through the standard Ecosim CSV output and program interface (Christensen *et al.*, 2004a). In order to get dynamic information regarding the settling phase, the EII export/import procedure was used to generate a new equilibrium model at the termination year of the rebuilding phase. In rare cases, minor changes were made to the imported models to recreate mass-balance, but any effect on the equilibrium position was minimal. All indices measuring harvest benefit were assumed to be at steady-state for the equilibrium harvest phase following the termination year of the settling phase.

In the economic analysis of restoration, costs and benefits in each year are considered in relation to the *status quo* profit (i.e., current profit from fisheries). The profit, I assume, will remain constant over the next 50 years, and therefore that the current levels of fishing mortality in

northern BC are sustainable. There may be reason to doubt this where declining species are concerned, notably rockfish and other valuable demersal fish. I am therefore using a generous estimate of baseline profit, at least towards the end of the simulation. In reality, the current real-world level of profit may be expected to decline over the next 50 years. In that case, the benefits of restoration would be underestimated, and consequently, the NPV of the restoration plan.

7.3 Results

Algorithm diagnostics

This section evaluates the behaviour and success of the new SB objective function in obtaining a desired ecosystem configuration. Starting with the present-day 2000 ecosystem, I draft restoration scenarios to reconstruct the historic ecosystems of 1950 and 1900. I first tested the parameters that set functional group priorities for rebuilding: the unit of improvement and the marginal improvement model. Next, I evaluate how closely the ecosystem can be manipulated to resemble the target. All restoration scenarios in the diagnostics section use the *maxdex* fleet, so that the full ecological potential for restoration can be represented without the impediment of unselective fishing gear.

Unit of improvement

Fig. 7.6 shows the performance of the SB algorithm towards achieving an ecosystem target, the historic 1950 ecosystem. The algorithm operates as expected. Using 'biomass' as the unit of improvement allows a greater overall change in functional group biomass towards the target configuration, than either the 'mixed' or 'percent' unit of improvement. This is revealed by the sum of squares of the functional group biomass values in the restored ecosystem versus the goal ecosystem (Fig. 7.6a). Selecting 'biomass' as the relative unit of improvement will result in the rebuilding of mainly large and depleted functional groups. The percent criteria for ecosystem improvement results in the greatest average percent improvement across functional groups; where 100% improvement corresponds to perfect restoration in all groups (Fig. 7.6b). The percent and mixed unit of improvement affects a greater number of functional groups than the

biomass unit of improvement, although the total change in system biomass is not as drastic, indicating that low-biomass functional groups most often restored under these settings (Fig. 7.6c). In all cases, restoration plans concerning only commercial functional groups are able to achieve a closer match to the target ecosystem, then when all functional groups are considered. There are two reasons for this. First, fewer conflicting dynamics need to be resolved in an optimization of narrow purpose, and second, commercial groups can be neatly restored through the direct action of fisheries (more precisely the inaction of fisheries), rather than through indirect trophic effects.

Marginal improvement valuation model

Fig. 7.6 also shows what effect the marginal improvement valuation model has on the policy outcome. As expected, the linear model produces the greatest overall change in the ecosystem (Fig. 7.6a); this is revealed in a low sum of squares versus the target configuration. The quadratic and gamma valuation models are not able to restructure the ecosystem to match the target configuration as precisely. The average percent improvement across functional groups is approximately the same for the linear and quadratic valuation models; however, the gamma model sacrifices precision by allowing more groups to exceed their target biomasses (Fig. 7.6b, also see Fig. 7.7). The number of functional groups that see improvement is not dependant on the valuation model selected (Fig. 7.6c). In all cases, restoration plans concerning only commercial functional groups are able to achieve a closer match to the target ecosystem then when all functional groups are considered.

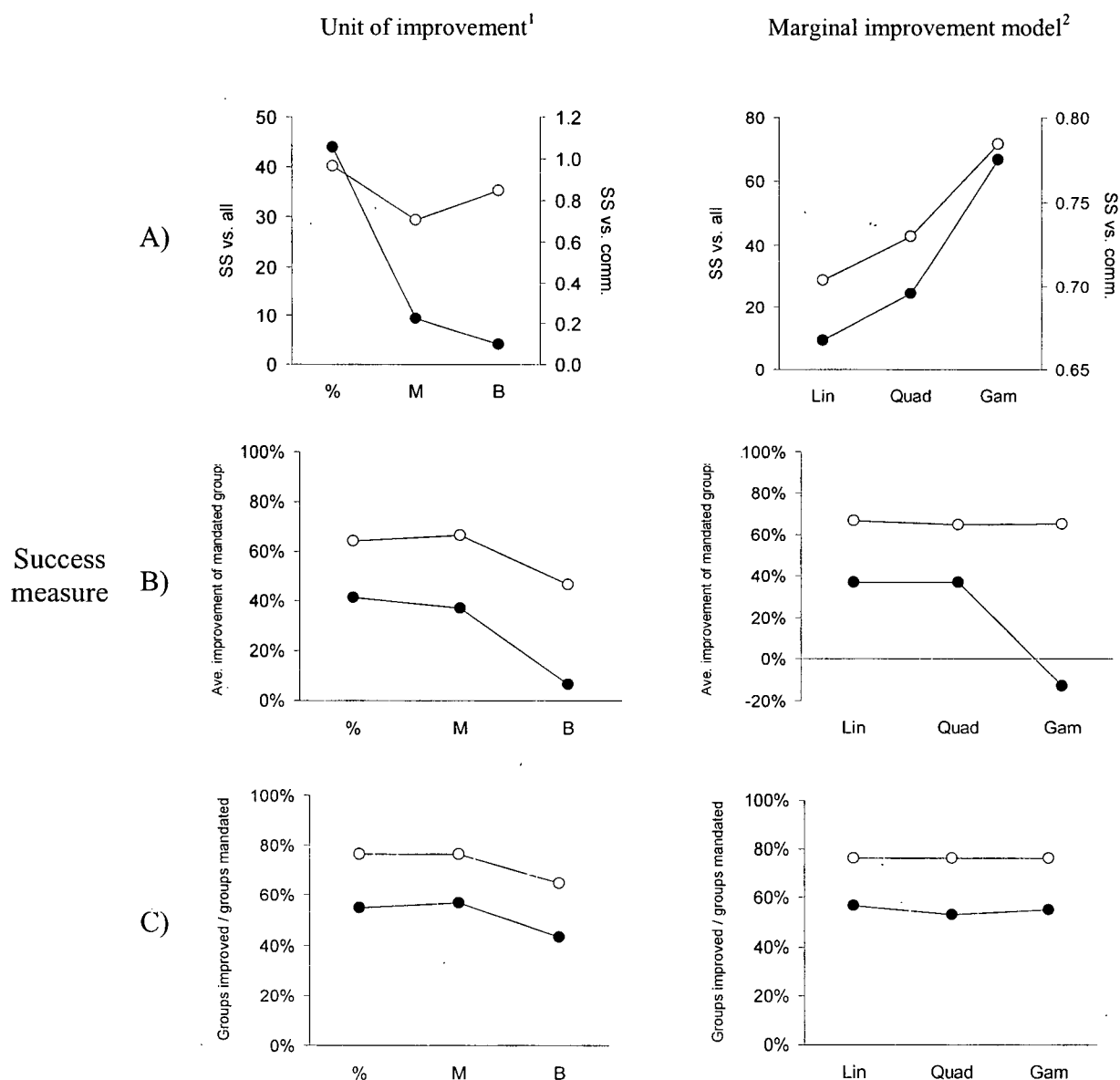


Figure 7.6 Performance of SB algorithm towards achieving historic 1950 ecosystem structure. Shown are 50-year forecasts beginning from the present-day, applying optimal fleet fishing mortalities with the *maxdex* fleet. Closed circles show optimizations where all groups are mandated for rebuilding; open circles show commercial groups mandated only. Optimizing for biomass change as the unit of improvement (B) concentrates rebuilding efforts on fewer, large functional groups and sum of squares, which is based on biomass, improves; percent (%) improvement affects a greater number of groups, and the groups tend to be smaller. Mixed (M) runs show intermediate solutions. The linear marginal improvement model achieves a greater overall change in biomass than the more precautionary quadratic and gamma models. The optimization can approach its goal more closely when fewer groups are mandated (e.g., commercial groups only).

¹ Uses linear marginal improvement model. Initial (2000) SS is 7.9 versus all groups; 1.58 versus commercial.

² Uses mixed unit of improvement. Initial (2000) improvement is ~ 0%.

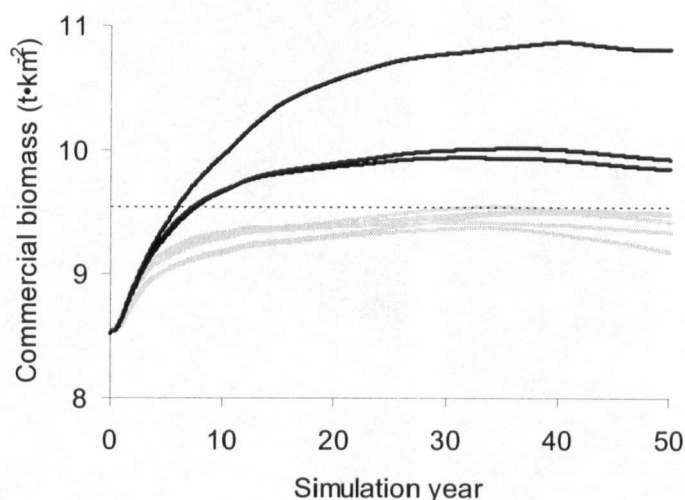


Figure 7.7 Commercial biomass increase under various restoration plans. Target ecosystem is 1950 (dotted line). The results of nine optimizations are shown, each varying the unit of improvement (percent, mixed, biomass) and marginal improvement model (linear, quadratic, gamma). Black lines are 'gamma' model optimums, grey lines are 'quadratic' and 'linear' optimums. The precautionary gamma model overshoots the target biomass; the greatest biomass (top line) is achieved by the gamma model using "biomass" unit of improvement. All solutions are optimized for a 50-year harvest using the *maxdex* fleet.

Achieving a specific biomass structure

Fig. 7.8 demonstrates the success of the SB algorithm in reconfiguring the ecosystem to resemble a specific goal configuration. The target for restoration in this case is the 1950 ecosystem. Principle components analysis (PCA) summarizes the similarity of the end-state functional group biomass vector with the target ecosystem configuration after a 50-year restoration plan. Fig. 7.8a shows that effective ecosystem reconfiguration is possible when the biomass of all functional groups is mandated for adjustment. Since the PCA recognizes changes in biomass, only plans optimized under the biomass unit of improvement can be expected to achieve results. However, when fewer groups are mandated for restoration (Fig. 7.8b), the distinction between biomass and percent unit of improvement becomes less important (i.e., the solutions converge). In that case, all optimizations achieve success regardless of parameter settings.

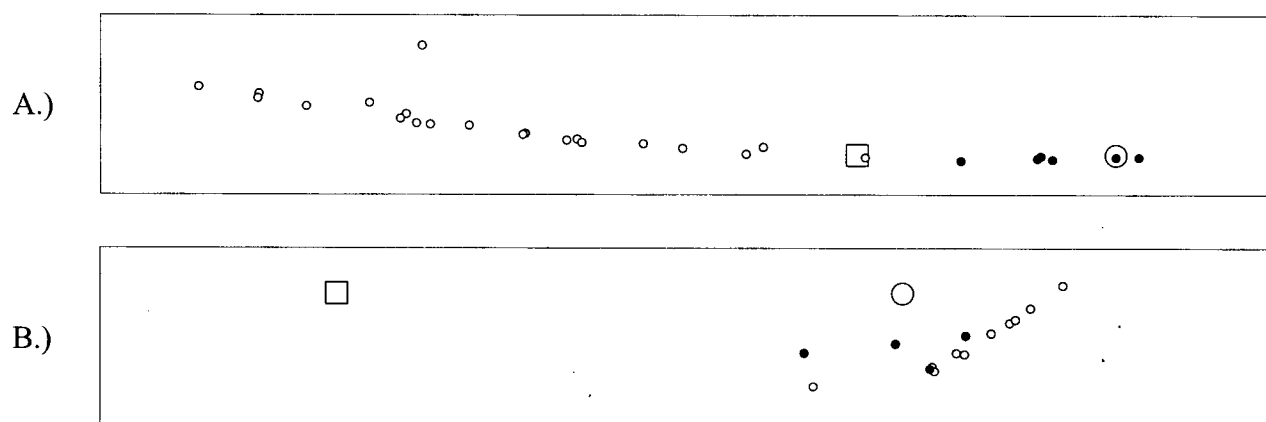


Figure 7.8 Principle components analysis showing ecosystem configurations after restoration. Large square represents the initial ecosystem configuration (2000); large circle represents the goal configuration (historic 1950 ecosystem); small closed circles show harvest simulations optimized for biomass recovery (unit of improvement is biomass); small open circles show additional simulations (percent and mixed unit of improvement). Fishing plan uses *maxdex* fleet. A.) All groups mandated for restoration. B.) Commercial groups mandated only. Simulations optimized for biomass recovery (unit of improvement is biomass) tend to cluster around the goal ecosystem, indicating restoration success. When fewer functional groups are mandated for restoration the goal is more easily approached, and the distinction between percent improvement and absolute improvement in biomass disappears (simulations converge on the same solution). Plots have been rotated to reveal restoration success along the X-axis.

Fig. 7.9 shows the average improvement in functional group biomass for all restoration plans tested that target the historic 1950 ecosystem. All scenarios represent 50-year restoration plans to allow the ecosystem to settle on a new group-biomass equilibrium, and variations on all key parameters are tested (unit of improvement and marginal improvement valuation model). When 100% agreement is achieved, functional group biomass exactly resembles the target. The range of end-state biomass is broad for most functional groups, indicating that there is a good deal of variation in policy outcomes depending on the algorithm settings. For many functional groups, end-state biomass regularly exceeds B_0 (unfished biomass: calculated by equilibrium routine in Appendix Fig. A5.5.1). This is the result of ecosystem restructuring; functional group biomass is driven above the unfished level through the manipulation of predator, competitor and prey biomasses. The carrying capacity of these groups has been changed in the ecosystem to meet our requirements. Most functional groups demonstrate the potential to grow (or decline) to the biomass level prescribed by the target 1950 historical ecosystem. Use of the key parameters will determine which groups achieve the greatest restoration improvement, and which 'supporting'

groups must remain far off their target biomass. When fewer groups are mandated for restoration the goal is more closely approached (Fig. 7.9b).

Fig. 7.10 shows the end-state functional group biomass in relation to the target ecosystem (historical 1950) as determined by the most successful optimizations (judged visually). Key parameters varied include the unit of improvement and the marginal improvement valuation model. As demonstrated earlier, average improvement is less when all groups are included in the restoration objective (Fig. 7.10a). However, the axes in Fig. 7.10a are heavily aggregated across functional groups, and the aggregation belies the true improvement on a group-by-group basis. Fig. 7.10b more clearly demonstrates the ability of the SB algorithm to restructure the ecosystem; it shows the results of a simpler restoration objective that considers commercial groups only.

Fig. 7.11 shows the rebuilding success to the 1900 ecosystem target. Although there is improvement, the rebuilding plan tested is not sufficient to restore the ecosystem to this ancient condition. Likely, complex optimizations would be required to approach the target more closely, that is, using dynamic optimal fishing mortalities as opposed to constant equilibrium-level mortalities (see discussion).

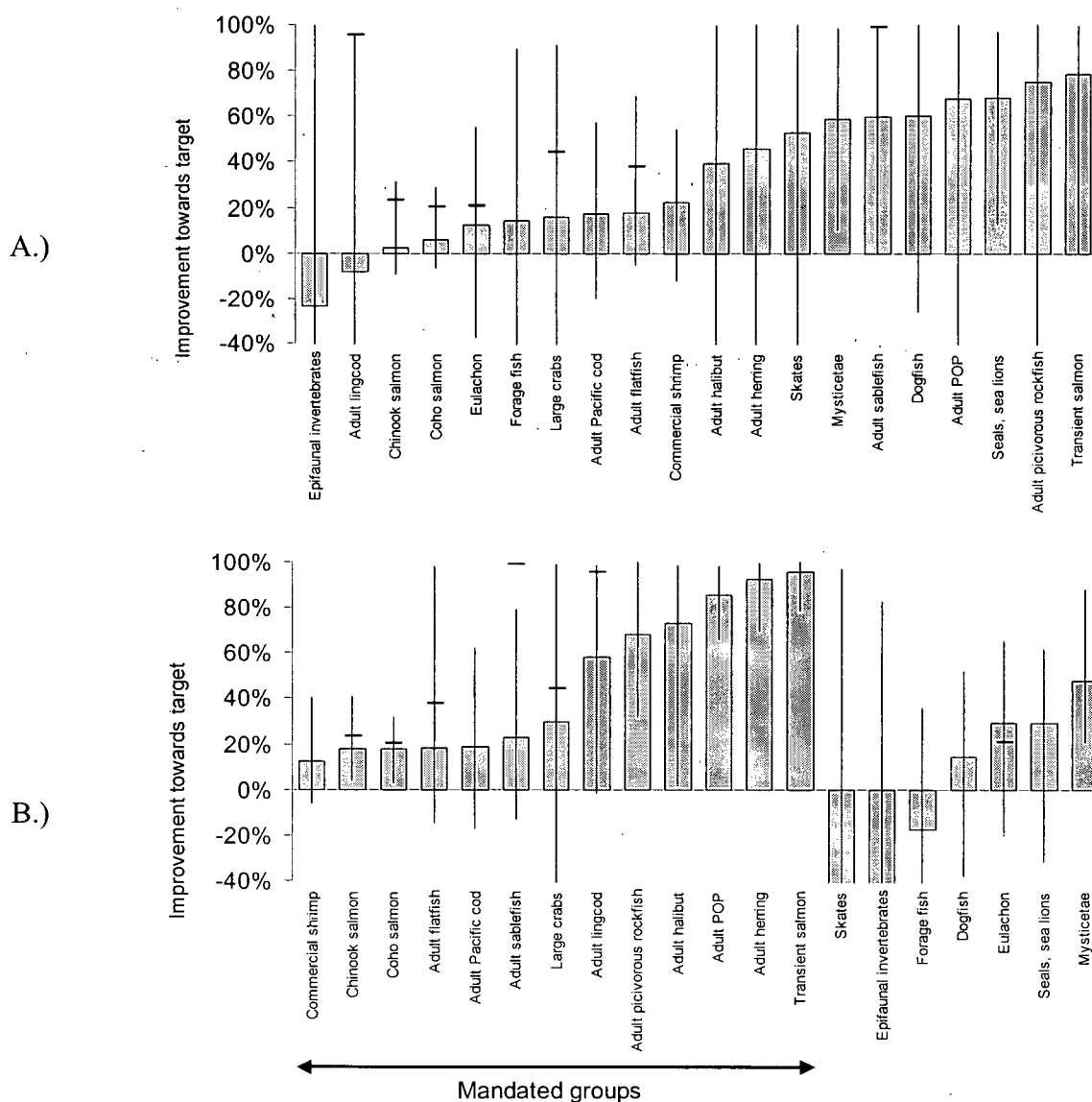


Figure 7.9 Average improvement in functional group biomass towards target level after restoration. Fifty year restoration plans use varied settings for unit of improvement and marginal improvement valuation model. For all restoration scenarios, the *maxdex* fishing fleet is used to manipulate the ecosystem into a form resembling the historic 1950 condition. Y-axis indicates percent improvement towards the target biomass level by simulation end-state. At the first iteration of the policy search, functional groups begin at ~0%. At 100%, functional group biomass equals its target exactly. ‘Improvement’ may describe population growth or decline. Cross bars show unfished biomass B_0 estimated by the 2000 model under baseline conditions (equilibrium analysis; Appendix Fig. A5.5.1). The B_0 value is off the scale for most groups, indicating that the optimal biomass is below B_0 . Error bars show total range of output observed from the optimizations. A) All groups mandated for restoration; B) commercial groups mandated only. Groups are not shown whose start biomasses are within 10% of goal. In many cases, the biomass of functional groups is driven above B_0 in pursuit of target biomass levels. Carrying capacity has been increased by the restoration policy.

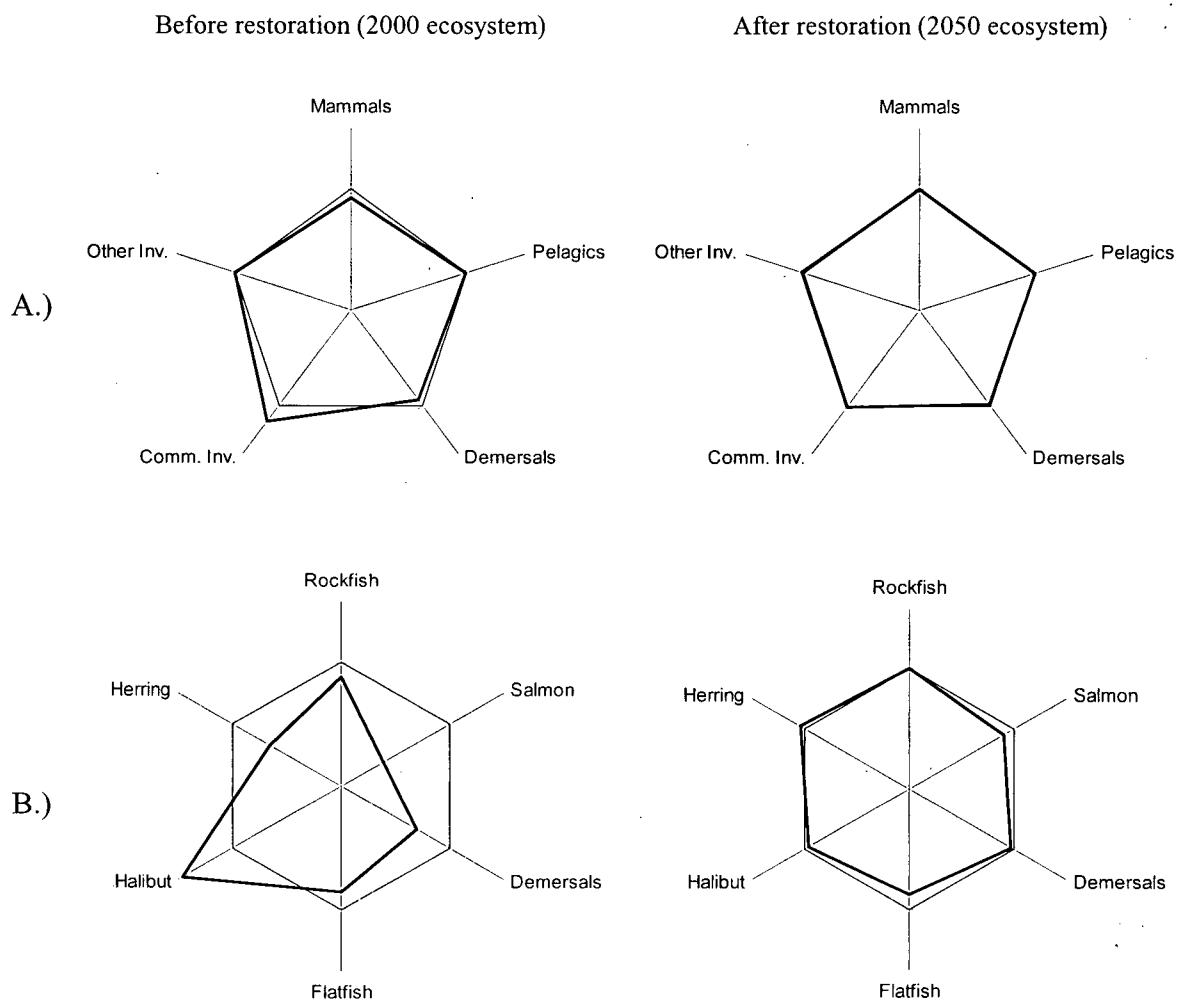


Figure 7.10 End-state group biomass after rebuilding relative to target 1950 goal biomass. Results shown for best optimizations¹ using the *maxdex* fleet. Thin line shows 1950 goal biomass (set at 1); thick line shows initial 2000 biomass (left) and restoration end-state biomass (right). A.) All functional groups mandated for restoration. B.) Commercial functional groups mandated only. Species groups shown represent aggregate biomass values for several functional groups.

¹ A.) Unit %, marginal model gamma; B.) Unit biomass; marginal model linear.

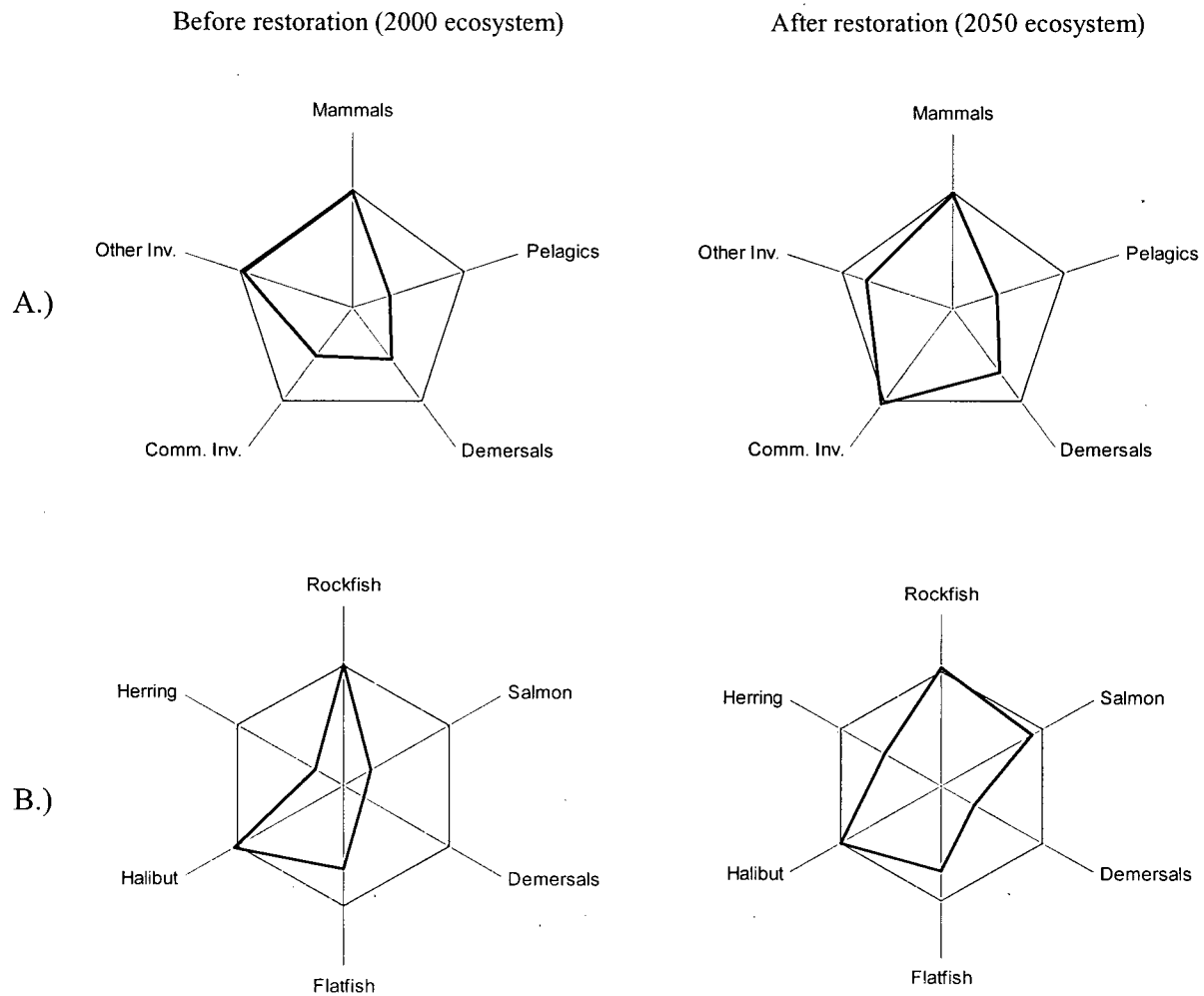


Figure 7.11 End-state group biomass after rebuilding relative to target 1900 goal biomass. Results shown for best optimizations¹ using the *maxdex* fleet. Thin line shows 1900 goal biomass (defined as 1); thick line shows initial 2000 state (left) and restoration end-state (right). A.) All functional groups mandated for restoration. B.) Commercial functional groups mandated only. Species groups shown represent aggregate biomass values for several functional groups.

¹ A.) Unit %; marginal model gamma. B.) Unit mixed; marginal model gamma.

Quantifying tradeoffs

Restoration plans featured in Fig. 7.12 attempt to restore the historic 1950 ecosystem. Restoration plans are tested that incrementally increase the relative weighting of the economic criterion in the objective function versus the SB rebuilding criterion. Policies that increase biodiversity tend to generate less end-state profit, and plans that generate more profit tend to sacrifice any potential gain in biodiversity. Considering only runs that successfully reduced the sum of squares versus the target ecosystem by more than 50%, a convex relationship illustrates potential benefits of a restoration policy directed towards this ecosystem target. Because the relationship is convex, restoration scenarios that leave the ecosystem in the middle of this range achieve an advantageous compromise between profit and biodiversity.

Fig. 7.13 shows end-state results after restoration for all optimizations tested in the diagnostics section varying key parameters (unit of improvement and marginal improvement valuation model). In these optimal harvest scenarios, the 2000 ecosystem is manipulated under a fifty year restoration plan, and the closed circles represent the end-state ecosystem condition, tailored to resemble either the 1950 or the 1900 target ecosystem. Commercial biomass is contrasted against biodiversity (represented by Q-90 and the Shannon-Weaver entropy function) (Shannon and Weaver, 1949). The 1950 ecosystem appears to be more achievable than the 1900 ecosystem, as its biomass configuration lies closer to the initial 2000 condition, the starting point for the optimization (initial sum of squares versus 1950 goal = 7.9; sum of squares versus 1900 goal = 861.6). The biodiversity indices give qualitatively similar results, although the optimization routine is unable to affect an adequate increase in the Q-90 index to match either the 1950 or the 1900 level.

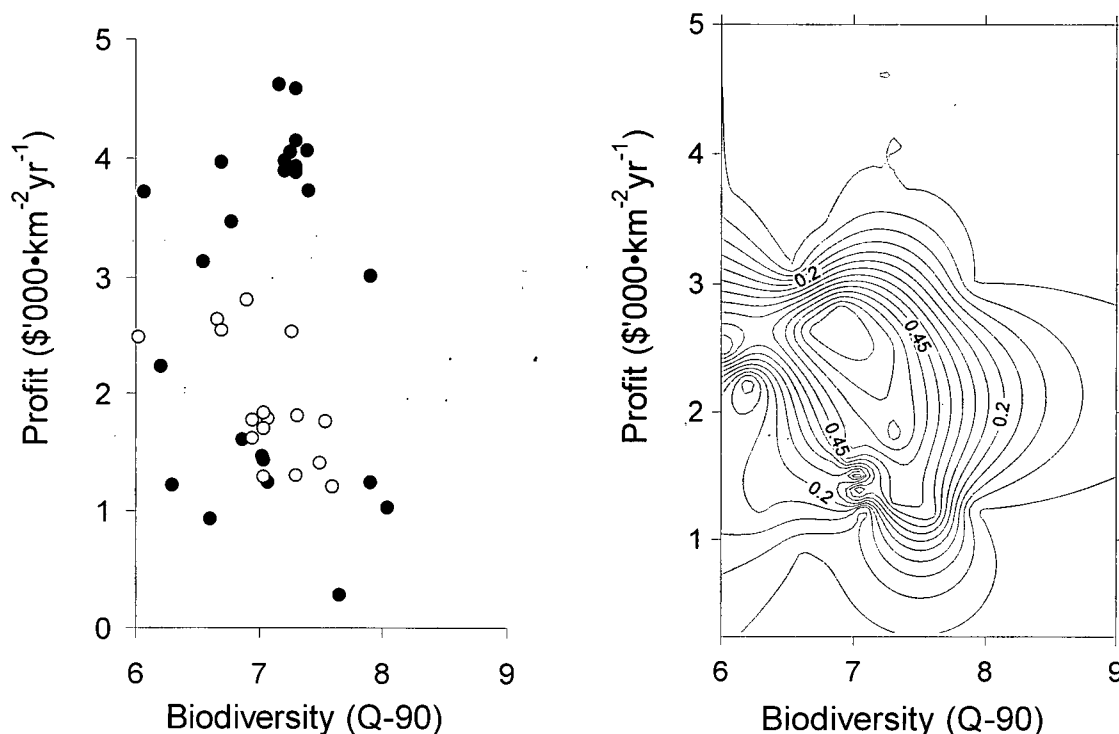


Figure 7.12 End-state profit and biodiversity of restoration plans targeting the historic 1950 ecosystem. *Maxdex* fleet is used. The optimal equilibriums were calculated using various weightings in the objective function for ecosystem restoration and economic performance. All functional groups were considered in the restoration objective; plans used biomass as the unit of improvement and employed a linear marginal improvement valuation model. All solutions were optimized using random-F starting points. Left: Open circles represent restoration plans that successfully reduced the sum of squares by $\geq 50\%$ versus target ecosystem group biomasses; closed circles represent less successful runs. Right: contours show reduction in sum of squares; peak represents 65% reduction. Initial ecosystem values: profit $\$480\cdot\text{km}^{-2}\cdot\text{yr}^{-1}$; biodiversity 7.4.

In order for the initial 2000 ecosystem to resemble its target (1950 or 1900), the restoration scenario must permit an increase of both (net) commercial biomass and biodiversity. However, the SB objective function does not include explicit reference to either of these requirements. Any increase in the biodiversity, for example, comes incidentally from the reconstruction of individual functional groups under the SB algorithm. Testing confirms that the 2000 model is capable of matching the goal ecosystem (1950 or 1900) in either of these broad terms, biodiversity or commercial biomass (i.e., in an optimization devoted to those criteria). However,

in that case, the net biomass or biodiversity gain would be achieved through manipulation of different functional groups than the ones required to match the target ecosystem under the SB objective function.

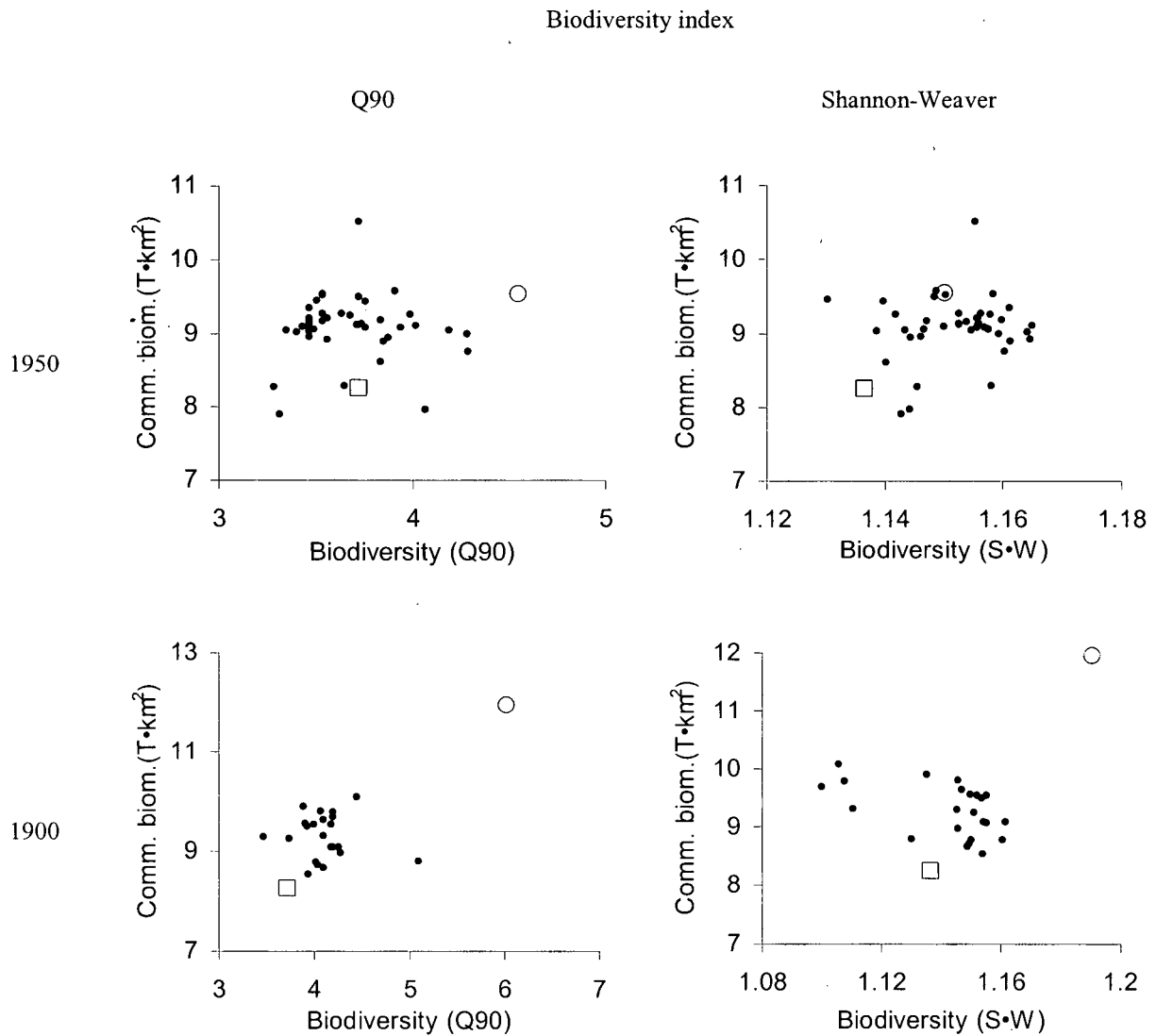


Figure 7.13 Progress towards goal ecosystems 1950 and 1900 for all diagnostic optimizations. *Maxdex* fleet is used. Open squares show initial ecosystem configuration (2000); open circles show goal ecosystem (1950 or 1900); closed circles show end state ecosystem configuration for all optimized fishing solutions. Optimizations vary in complexity, simulation length and other variables. X-axis shows biodiversity; Y-axis shows commercial biomass.

Evaluating restoration policies

The remainder of this results section will evaluate restoration trajectories to two ORB targets based on 1950 and a restoration target based on the historical 1950 ecosystem for comparison. The ORB targets are optimized using the biodiversity (Q-90) and the economic objective functions. The derivation of these ORB targets was presented earlier in Chapter 6. Details on the ORB targets are provided in Appendix Table A6.2.1 (biomass, biodiversity) and A6.2.2 (landings, value).

Restoring biodiversity ORB

Fig. 7.14 demonstrates the end-result of restoration plans targeting the biodiversity ORB using the *maxdex* fleet. These results represent the best improvement in biodiversity that can be achieved through equilibrium-level optimal fishing mortalities under the new rebuilding algorithm. Key parameters varied include the unit of improvement and marginal improvement valuation model. Biodiversity, as measured using the Q-90 index, is improved the most when using a mixed unit of improvement, but Shannon-Weaver biodiversity is better achieved through the percent unit of improvement. This result highlights functional differences between the biodiversity measures. Where Shannon-Weaver measures species evenness alone, the Q-90 index also considers species richness. Therefore, the Q-90 index responds better to restoration plans that value an increase in biomass, like the mixed plans. The level of commercial biomass required by the biodiversity ORB is $7.32 \text{ t}\cdot\text{km}^{-2}$, less than the initial 2000 state ($8.52 \text{ t}\cdot\text{km}^{-2}$). A close match is achieved by the linear model using biomass as the measure of improvement. The gamma model tends to overshoot the target biomass compared to the linear and quadratic models.

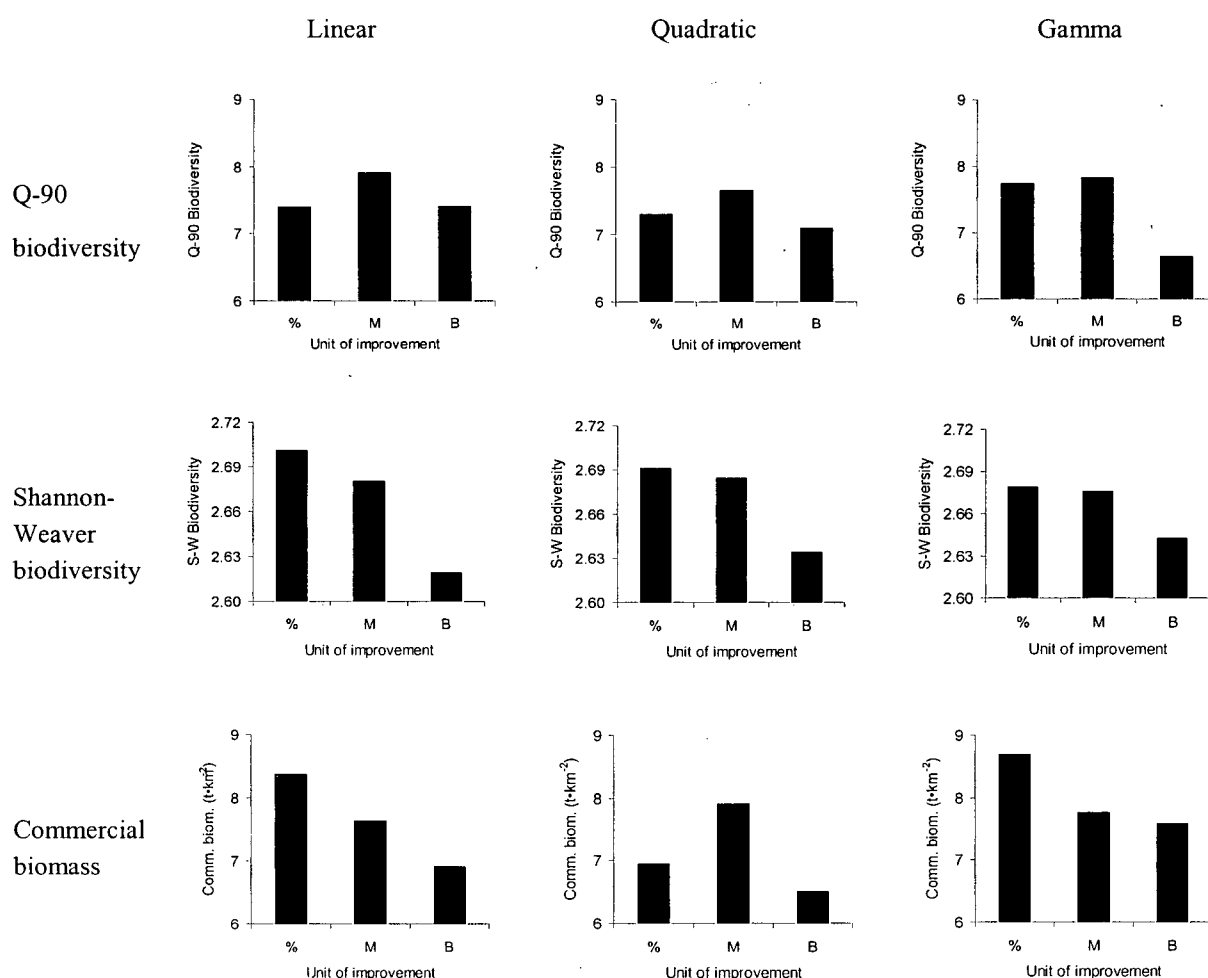


Figure 7.14 End-state ecosystem condition of nine restoration plans targeting the biodiversity ORB. *Maxdex* fleet is used. Optimizations vary the marginal improvement valuation model (linear, quadratic and gamma) and the unit of improvement (percent (%), mixed (M) and biomass (B)). Shown are end-state biodiversity (Q-90 and Shannon-Weaver) and commercial biomass. The target commercial biomass is $7.32 \text{ t}\cdot\text{km}^{-2}$; it is best achieved by the biomass criterion unit of improvement under the linear marginal valuation model. Plans use a 50-year simulation length; restoration of all functional groups is considered in the objective function.

Comparison of restoration goals: historic system versus economic ORB

Comparisons of candidate restoration strategies targeting the historic 1950 system and the economic ORB using the BC fleet are presented in Appendix Fig. A7.2.1; restoration plans using the *lost valley* fleet are in Appendix Fig. A7.2.2 and plans using the *maxdex* fleet are presented in Appendix Fig. A7.2.3. Simulations plotted in the Appendix vary the relative contribution of SB

and economic criteria to the overall objective function, in order to produce a spread of restoration trajectories. However, all restoration plans presented achieve at least some degree of improvement in the sum of squares criterion.

Fig. 7.15 shows the end-state ecosystem condition after 50 years of restoration using the SB algorithm. Each point represents a different candidate restoration plan, optimized using various SB parameter settings. The restoration simulations begin from the present-day (2000) ecosystem. The target for restoration is the 1950 historical ecosystem or the economic ORB ecosystem. This example demonstrates that for any equivalent reduction in the sum of squares versus the two target ecosystem configurations, the economic ORB produces more profit than the (unmodified) historical ecosystem. This is evidence of the fact that the economic ORB ecosystem has been retooled to support higher populations of valuable species. The closer that the optimization was able to approach its target, the more profit was made at end-state. The opposite is true for runs considering commercial groups only for restoration – in that case, profit is reduced to allow rebuilding of commercial groups; and the most successful harvest plans (i.e., those that reduce SS discrepancies versus target) make less profit.

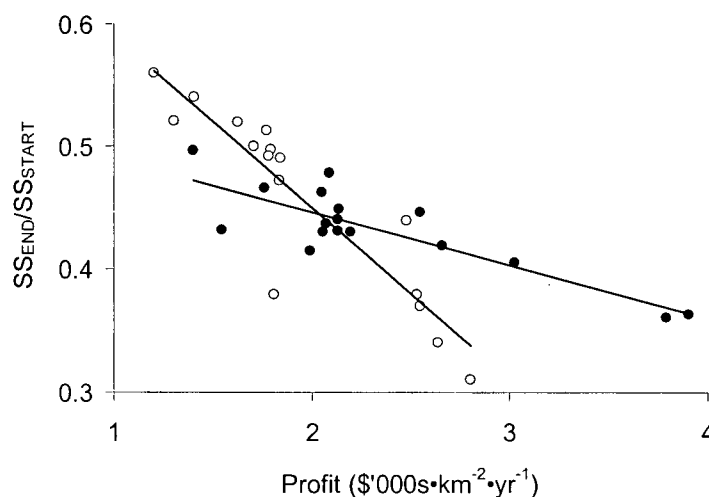


Figure 7.15 End-state profit after 50 years of restoration versus sum of squares against goal ecosystem. Goal ecosystem: 1950 historical ecosystem (open circles); economic ORB ecosystem (closed circles). All functional groups were considered in the restoration objective; plans used biomass as the unit of improvement and employed a linear marginal improvement valuation model. The optimal equilibriums were calculated using the *maxdex* fleet by applying various weightings in the objective function for ecosystem restoration and economic performance.

Fig. 7.16 shows that biodiversity and average trophic level of functional groups tends to increase from model baseline (present-day 2000) under optimizations targeting the 1950 economic ORB ecosystem, but optimizations targeting the 1950 historic ecosystem do not see as consistent of increase in either factor. The biodiversity of the economic ORB lies above the baseline condition, despite the fact that biodiversity was not a criterion for developing the ORB. The increase in trophic level stimulated by the ORB goal is expected, since the ORB represents an economically optimized target - and since predator fish tend to be worth more than low trophic level fish (market prices: Appendix Table A5.1.6). The 1950 target has a lower average functional group trophic level than the baseline condition, so it is not surprising that the simulations targeting this ecosystem configuration, in many cases, reduce the average trophic level of the system. However, biodiversity of the 1950 ecosystem lies above the baseline condition and this is not reflected in the results. In fact, the 1950 historic ecosystem has a higher biodiversity than the economic ORB (4.35 versus 3.36; compare Appendix Tables A5.1.2 and A6.2.1), yet optimizations targeting the ORB perform better in this respect. This may suggest that high biodiversity is in some way linked to the optimal economic performance of the ecosystem. For example, if valuable predators consume a wide range of diet items, then the supporting prey base would tend to be diverse. Further research may substantiate this.

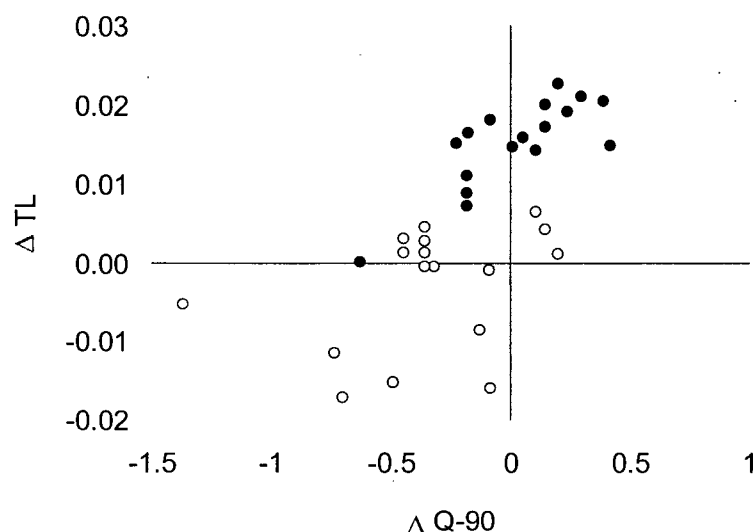


Figure 7.16 Change in average system trophic level and biodiversity following restoration. Y-axis shows change in trophic level (ΔTL); X-axis shows change in biodiversity ($\Delta Q-90$) after 50 years of restoration to the 1950 historical ecosystem (open circles) and the economic ORB ecosystem (closed circles). The origin represents the present day (2000) ecosystem. Goal values for the economic ORB are +0.074 (ΔTL) and +0.30 ($\Delta Q-90$); goal values for the 1950 historical system are -0.043 (ΔTL) and +0.43 ($\Delta Q-90$). Plans used biomass as the unit of improvement and employed a linear marginal improvement valuation model; all functional groups were considered in the restoration objective. The optimal equilibriums were calculated using the *maxdex* fleet by applying various weightings in the objective function for ecosystem restoration and economic performance.

Effect of fleet structure

Three fleets are compared in Fig. 7.17 in their ability to restore target ecosystems (economic ORB derived from 1950, and historical 1950). Consistently, the *maxdex* fleet achieves a better match to the target ecosystem as measured by the sum of squares, followed by the BC fleet and the *lost valley* fleet. The *maxdex* fleet has 27 gear types, the BC fleet has 17 gear types and the LV fleet has 12 gear types. Additional gear types generally provide the optimization routine with improved ability to manipulate the ecosystem, but there is an exception. When all groups

are mandated for rebuilding under the ORB goal the *lost valley* fleet out-performs the more intricate fleet structures. Since the *lost valley* fleet was itself used to determine the ORB structure, it may have an inherent advantage over the BC and *maxdex* fleets (see discussion). A similar result is found when attempting to restore the ORB structure optimized for biodiversity. In that case, the *lost valley* fleet again achieves a closer match to the target ecosystem (84% reduction in sum of squares) than the BC fleet (80%) or *maxdex* fleet (80%).

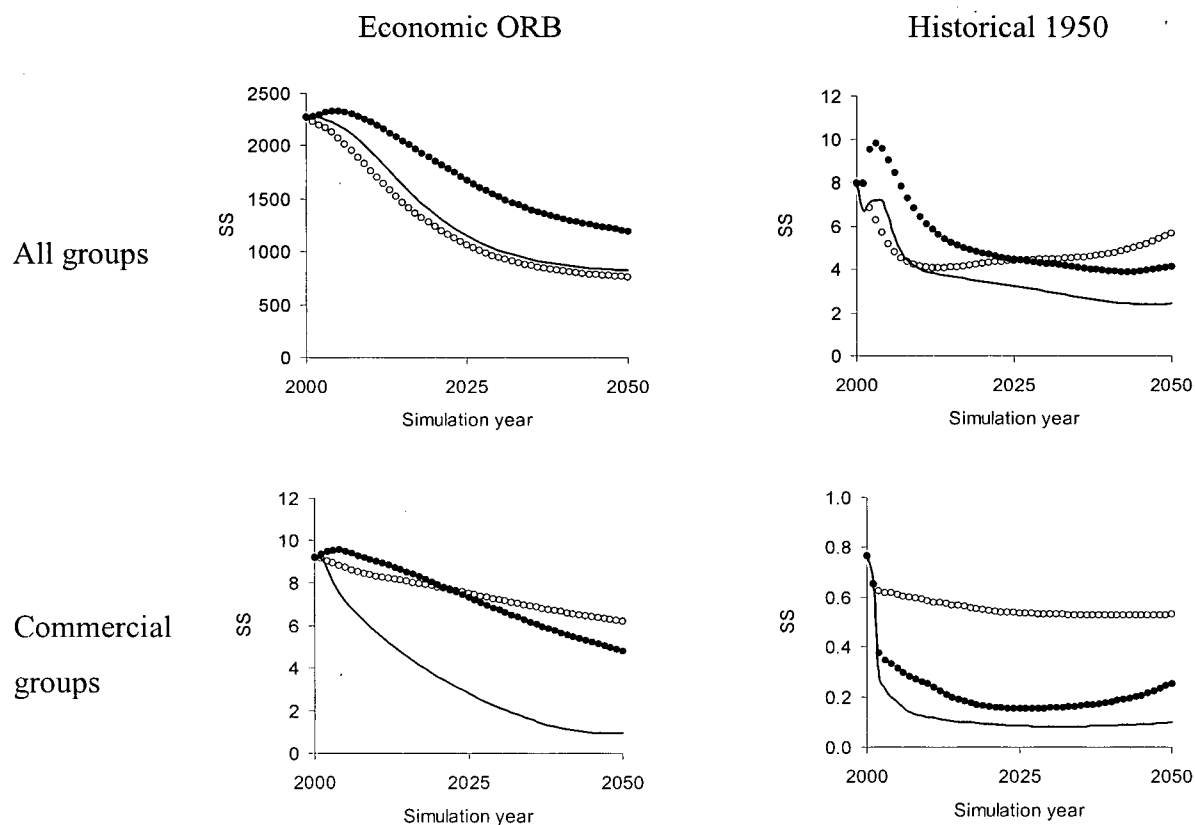


Figure 7.17 Best reduction in sum of squares versus target system achieved by SB algorithm. Target ecosystems are 1950 economic ORB (left) and historic 1950 ecosystem (right). Fleets used are: British Columbia fleet (closed circles), *lost valley* fleet (open circles) and the *maxdex* fleet (line).

Ecological limits on restoration

Fig. 7.18 shows the full range of possible restoration outcomes tested in this report. All parameters are varied, including restoration goal, fleet structure, objective function and search variables. The broken horizontal and vertical lines mark the current ecosystem condition (2000). In the upper-right quadrant, all restoration plans achieve an improvement in profit and biodiversity. The boundary established by the outermost range of points defines the ecological limit to restoration achievable using simple (i.e., equilibrium level) optimal fishing mortalities like the ones tested in this report. There is the potential to increase overall gains in profit and biodiversity beyond this level using complex multi-stage restoration strategies, but this was not done due to software limitations (see discussion).

Each point displayed in Fig. 7.18 achieves an improvement in the sum of squares versus its target ecosystem configuration, although only a subset of those optimizations reveal an increase in both profit and biodiversity. Any improvement in profit or biodiversity above baseline levels has resulted indirectly by restructuring ecosystem components towards the restoration goal. However, by including an optimization criterion for profit or biodiversity, in addition to the rebuilding criterion, we could direct the policy search to uncover only those desirable optima.

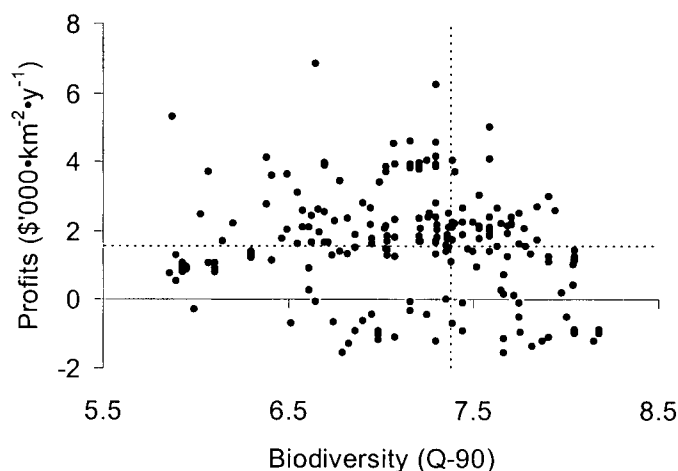


Figure 7.18 End-state profit and biodiversity after restoration for all 50-year restoration plans tested. Scenarios vary restoration goal, fleet structure, objective function and search variables. Horizontal and vertical broken lines indicate the baseline (2000) ecosystem condition.

Cost-benefit analysis of restoration

Fig. 7.19 provides a worked example of restoration that could be done immediately using the BC fleet. It targets the economic ORB developed from 1950. Profits are reduced from baseline levels for 30 years, following the restoration plan advocated by the SB algorithm using biomass as the unit of improvement and using the linear marginal improvement valuation model. As with all economic restoration plans targeting commercial species, profit is reduced during the rebuilding phase. This plan causes an overall increase in catch to trim down functional group biomass to economically optimal levels. Commercial group biomass levels in the economic ORB ecosystem are typically greater than the present-day 2000 ecosystem

(Appendix Table A.7.1.1), requiring population growth throughout the rebuilding phase to achieve the target. However, the biomass of some functional groups must be decreased from

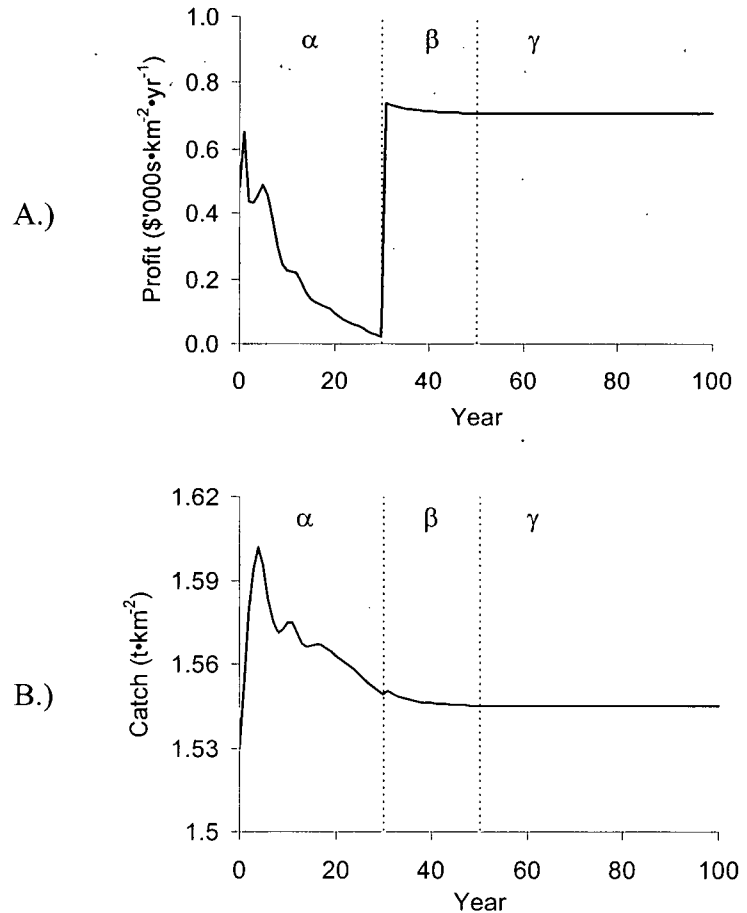


Figure 7.19 Worked example of a 30 year ecosystem restoration plan. Restoration plan includes rebuilding phase (α), settling phase (β) and equilibrium harvest (γ). A.) Profit (baseline is $0.48 \$'000s \cdot km^2 \cdot yr^{-1}$); B.) Catch (baseline is $1.53 t \cdot km^2$). Example shows rebuilding towards economic ORB biomass levels (commercial groups only); unit of improvement is biomass, marginal improvement valuation model is linear.

present levels to achieve the economic ORB (e.g., herring, halibut and rockfish (except inshore rockfish)).

Varying speed of restoration

Fig. 7.20 evaluates the NPV of restoration to the economic ORB ecosystem using the *lost valley* fleet and the BC fleet. These scenarios apply optimal fishing mortalities to rebuild the system until an arbitrary restoration target is achieved. For policies that target commercial groups only, restoration is considered 'complete' when the sum of squares has been reduced by 20% against the goal configuration. For policies that target all functional groups, sum of squares must be reduced by 45%. These values were chosen to provide the widest spread of points along the X-axis in Fig. 7.20. A variety of restoration plans are tested that incrementally increase the weighting of the economic benefit criterion versus the SB rebuilding criterion in the policy search objective function. By adding an economic component in the optimization, we slow down restoration. The rebuilding phase takes longer to achieve the specified reduction in the sum of squares but a greater annual share of profits is enjoyed by resource users.

The rebuilding plans shown in Fig. 7.20 allow some level of fishing to occur throughout the rebuilding process. This important attribute could make restoration more socially responsible and acceptable to stakeholders than a bang-bang approach to rebuilding, like the one presented by Clark and Munro (1975). Fundamentally, a bang-bang approach would not help this kind of precise ecosystem manipulation since we are not strictly eliminating fishing effort in these rebuilding plans. We are instead using selective fishing as a tool to manipulate the ecosystem, and careful application of fishing effort (e.g., to remove competitors of depressed species) should out-perform a blanket policy of fishing cessation. Moreover, non-malleability of fishing capital and other economic externalities should slow down the optimal rate of rebuilding in real applications compared to the idealized bang-bang solution, as would encumbering species interactions that can now be explicitly managed with this ecosystem-based approach.

NPV is calculated in Fig. 7.20 assuming a standard discount rate of $\delta = 5\%$, similar to long-term bank interest¹⁶. Restoration plans are devised using various weightings on economic and SB rebuilding criteria in the policy search to produce a spread of points. At one extreme, the plans achieve restoration quickly at the expense of profits. At the other extreme, restoration proceeds slowly and costs are spread out over a longer period. The *lost valley* fleet is able to achieve restoration more quickly than the BC fleet when all functional groups are considered and more cost-effectively than the BC fleet when commercial groups are considered. Results indicate that slow restoration scenarios using the *lost valley* fleet can outperform bank interest when restoring commercial groups (*lost valley* fleet scenarios that take 25+ years to achieve the rebuilding benchmark have a positive NPV). Restoration using the BC fleet is never quite as profitable as bank interest. The relationship between the speed and profitability of restoration is clear when commercial groups are considered alone, since to rebuild commercial groups the policy search has only one option, to reduce fishing.

However, when all groups are considered the success of restoration does not closely relate to the annual share of profits. There may be several avenues open to the policy search routine that can achieve a similar reduction in the sum of squares, and only a fraction of those policies will involve restoring commercial species to a higher sustainable level of profitability. When restoring all groups to the economic ORB, the BC fleet can achieve some profitable scenarios. In this case, fishing levels on commercial groups are never reduced far from baseline, and the greatest improvement in the sum of squares is seen in non-commercial groups. Of the five restoration plans tested in Fig. 7.20 (BC fleet; all functional groups), sum of squares for the entire system is reduced an average of 47% versus the target ecosystem, but the sum of squares regarding commercial groups nearly doubles due to depletions, contrary to the requirements of the ORB target (commercial group biomass depleted ~23%).

¹⁶ Government of Canada marketable bonds, average yield (10+ years) was 4.29% as of Aug., 2005 [www.bankofcanada.ca]

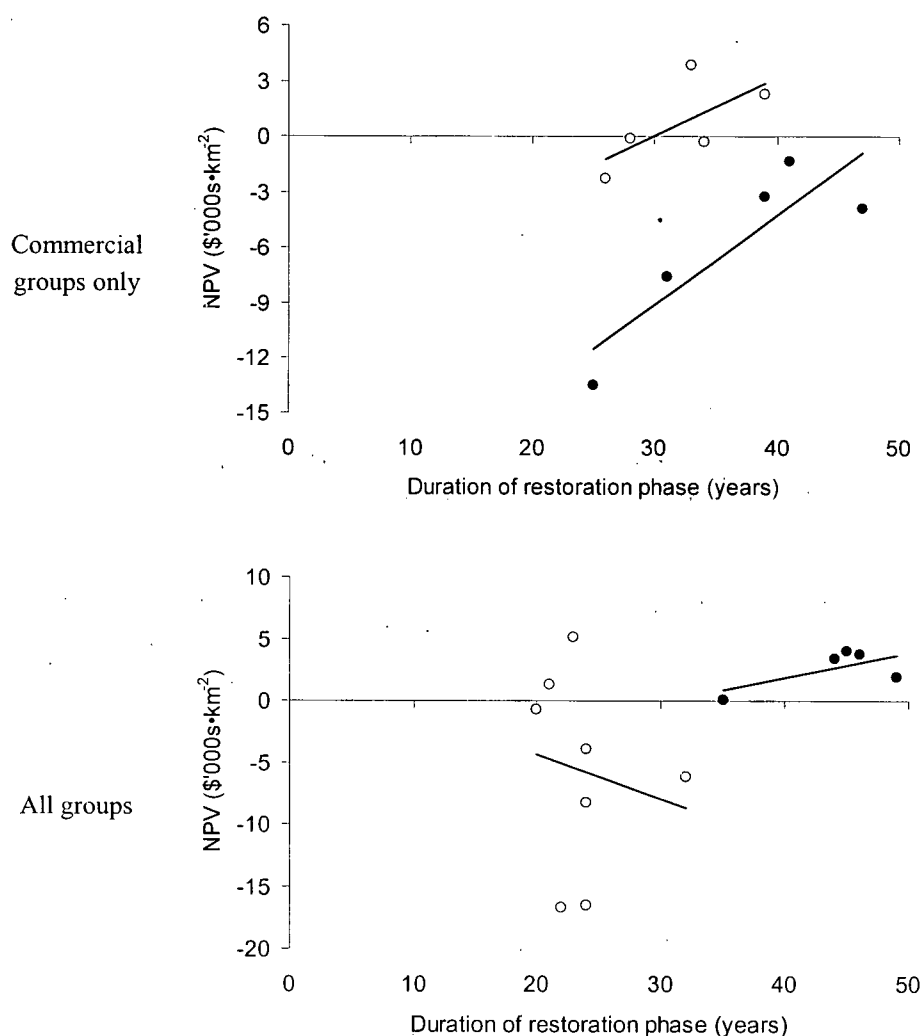


Figure 7.20 Net present value of restoration plans achieving a minimum reduction in residuals versus goal. Each scenario includes a restoration and harvest phase. Discount rate (δ) = 5%. Target ecosystem is the 1950 economic ORB. Fleets used are: BC fleet (closed circles) and the *lost valley* fleet (open circles). Criterion for successful restoration is a 20% reduction in the sum of squares versus commercial groups, when commercial groups are considered alone, and a 45% reduction when all groups are considered. These criteria were chosen to provide a maximum spread in points along the X-axis. Plans optimized for restoration achieve the SS reduction criterion quickly, but plans that include an economic objective achieve restoration more slowly and maintain higher annual profit during the rebuilding phase.

Varying restoration time

Where the previous section assigned an arbitrary criterion to determine when restoration is complete, this section will draft plans that last a fixed amount of time. Plans optimized to facilitate rebuilding tend to achieve a greater degree of restoration during the fixed period, while plans that include an economic optimization criterion tend to achieve less restoration but maintain higher annual profits.

Fig. 7.21 shows the results of restoration plans that target the economic ORB ecosystem based on 1950. Policies were determined by the optimization routine by including various weightings on the economic benefit criterion and the SB rebuilding criterion. The upper limit to profit (and the lower limit to biodiversity) represents restoration plans that achieve only a slight improvement towards the target ecosystem, measured using a least squares criterion versus the target ecosystem. The resulting spread of points in Fig. 7.21 therefore represents the full range of possible scenarios that can be used to transform the current ecosystem into one resembling the economic ORB. Prioritizing economic benefit any higher in the optimizations will prevent any improvement from being made towards the target ecosystem. Runs that are purely optimized for rebuilding are shown, and these represent the upper limit to restoration achievable using simple equilibrium-level optimal fishing mortalities.

The duration of the rebuilding phase is varied: 5, 7.5, 10, 15, 20 or 30 years. Presented in Fig. 7.21 is the new end-state profit equilibrium available from the system after restoration is completed (i.e., equilibrium harvest phase profit). Generally, longer restorations plans are able to achieve a closer match to the target ORB system, and so can deliver a greater sustained profit once the new equilibrium is reached. The restoration plans tested here use equilibrium-level optimal fishing mortalities, one for each gear type, which are applied to every year in the rebuilding phase. Most of the restoration benefit occurs within the first 15-20 years. It takes that long for the system to equilibrate to a new biomass configuration according to the imposed fleet-effort pattern.

Both the BC fleet and the *lost valley* fleet can be used to draft restoration plans that lead to an increase in annual profit. The profit is realized during the equilibrium harvest phase, while biodiversity increases throughout the rebuilding phase and the sum of squares is reduced versus the target ecosystem. The *lost valley* fleet has an advantage over the BC fleet for restoring profit but the BC fleet is better able to improve biodiversity. This is likely a modeling effect that has to do with the functional group aggregation style used for the fisheries (see Discussion).

Fig. 7.22 represents the NPV of restoration/harvest plans targeting the 1950 economic ORB ecosystem using the BC fleet and *lost valley* fleet. The economic evaluations assume a rebuilding phase of 5, 7.5, 10, 15, 20 or 30 years, a setting phase of 20 years, and continued harvesting at the restored profit equilibrium until year 100. Using either the BC or *lost valley* fleet, restoration scenarios are able to deliver a greater NPV than status-quo exploitation, which is assumed constant at the year 2000 profit level. As an investment in natural capital, restoration plans involving both fleets are able to outperform bank interest, and restoration plans using the *lost valley* fleet remain profitable even after 30 years of reduced fishing. The conventional NPV and the intergenerational NPV are not directly comparable since the intergenerational model includes bequest value in addition to monetary value (Sumaila and Walters, 2005). However, even restoration plans drafted using a strong ecological optimization criterion (i.e., lower points along the Y-axis spread) appear worthwhile under the intergenerational model of discounting, but only plans containing a strict economic optimization criterion (i.e., higher points along the Y-axis spread) are advisable under the conventional model. Gear structure is demonstrated to have a critical effect on the economics of ecosystem restoration.

Fig. 7.23 shows the internal rate of return (IRR) needed to make restoration/harvest plans economically worthwhile under conventional and intergenerational discounting approaches. Results from *lost valley* fleet reflect the potential economic advantage of this efficient fleet structure, but even the BC fleet is able to facilitate restoration scenarios that outperform bank interest. Using the BC fleet, plans that extend the rebuilding phase to 15 or more years are still able to provide attractive returns, albeit this mainly includes plans heavily weighted for economic performance (i.e., higher points along Y-axis spread). The *lost valley* fleet is able to proffer lucrative restoration plans that out-perform bank interest even after 30+ years of

restoration. In fact, the IRR of the *lost valley* fleet appears to asymptote at a level above the alternate rate of return. This indicates that profit taken during the rebuilding phase is greater than the *status quo* model baseline (and profits taken at the readjusted level during the equilibrium harvest phase are greater still). In other words, even the 'medicinal' fisheries designed to rebuild the system generate a greater profit using the *lost valley* fleet than current real-world fisheries do today.

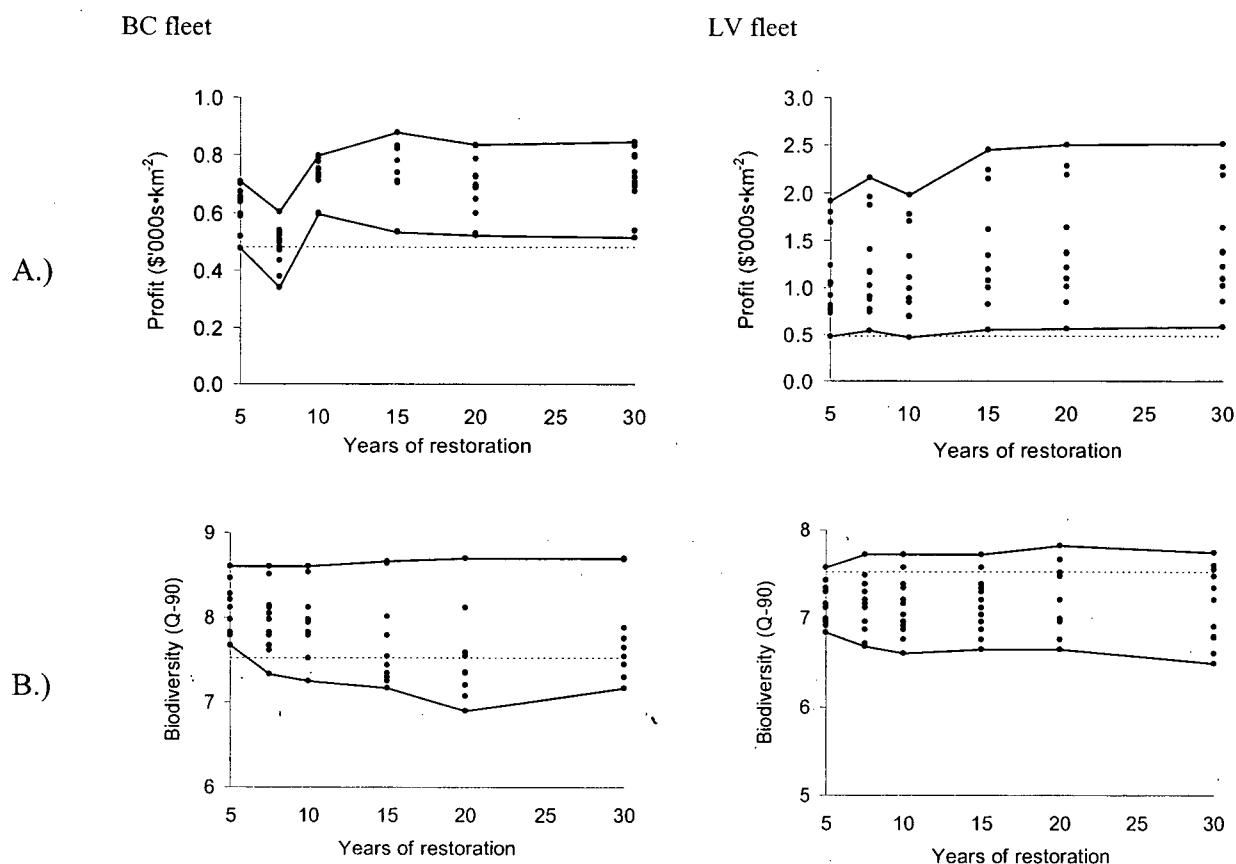


Figure 7.21 Equilibrium level profit and biodiversity achieved by restoration scenarios. A vertical spread of points is produced by varying the relative weightings on the economic and SB rebuilding optimization criteria. (A) shows profit achieved by various restoration scenarios; (B) shows biodiversity. The rebuilding phase, which followed previously, lasted for 5, 7.5, 10, 15, 20 or 30 years (X-axis). When recuperative fishing mortalities are applied for a longer period in the rebuilding phase, the ecosystem tends to enter the equilibrium harvest phase equipped with greater biodiversity and a higher profit equilibrium. The broken line indicates baseline (2000) profit and biodiversity, or the *status quo*; solid lines indicate maximum and minimum values obtained by any restoration plan. The target ecosystem for these rebuilding plans is the economic ORB ecosystem based on 1950 (commercial groups only). All optimizations consider biomass as the unit of improvement, and employ a linear marginal improvement valuation model. All restoration scenarios presented achieve a reduction in the sum of squares to some degree versus the target ecosystem. For the analysis of profit, the vertical spread of points along the Y-axis indicates runs optimized for economic returns at top, and runs optimized for strict rebuilding at bottom. The pattern is reversed for the analysis of biodiversity.

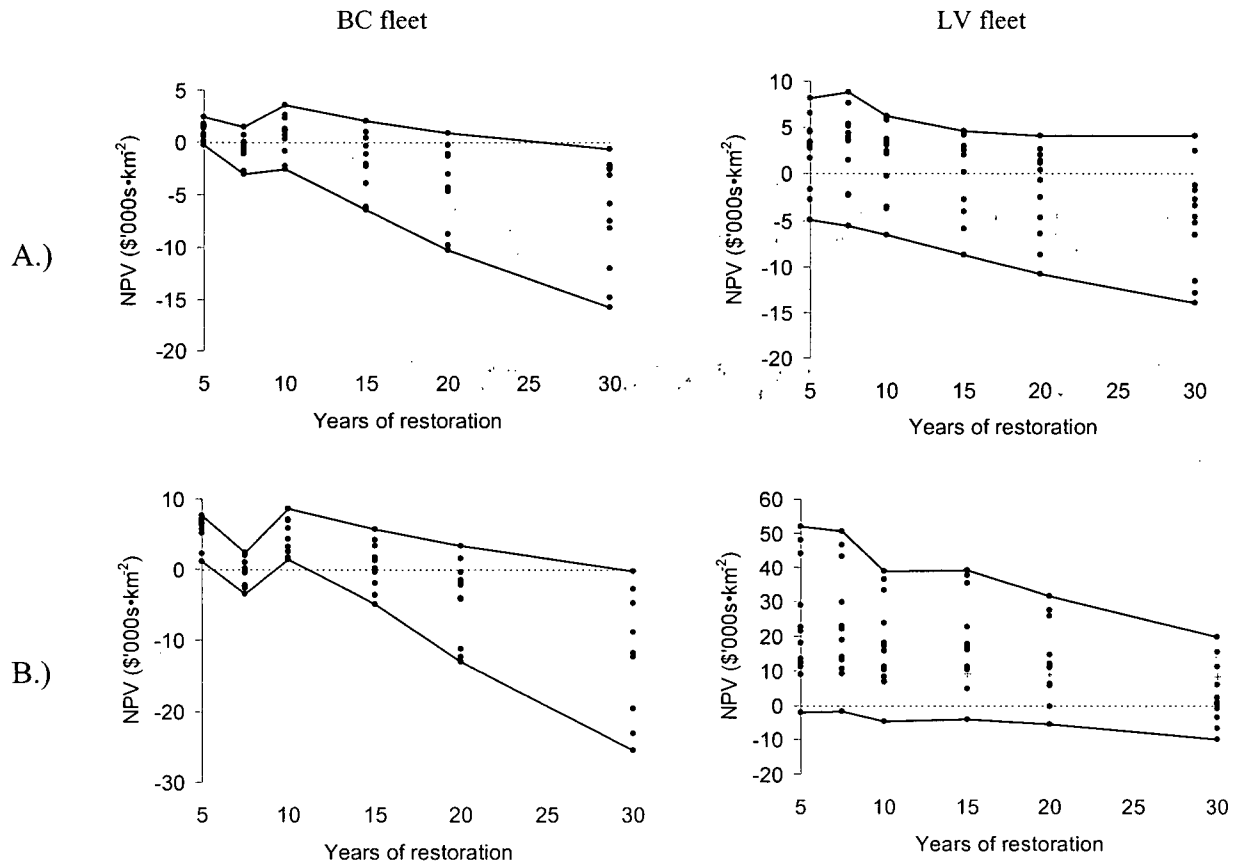


Figure 7.22 Net present value of restoration scenarios. Fleet used for restoration is BC fleet (left) and *lost valley* fleet (right). Restoration scenarios last 100 years, and include a rebuilding phase and an equilibrium harvest phase. Costs associated with rebuilding are incurred during the rebuilding phase, and benefits of rebuilding are accumulated throughout the equilibrium harvest phase. Rebuilding phase lasts 5, 7.5, 10, 15, 20 or 30 years (X-axis); equilibrium harvest phase continues until year 100. The profit stream predicted by each restoration scenario is evaluated under two discounting models, conventional and intergenerational. A.) Conventional discounting model uses a discount rate similar to long-term bank interest ($\delta = 5\%$). B.) Intergenerational discounting model uses the same standard discount rate ($\delta = 5\%$) and an equal intergenerational rate ($\delta_{fg} = 5\%$). Broken line indicates *status quo* NPV (assumed constant harvest benefits at year 2000 level). All optimizations use biomass as the unit of improvement, and employ a linear marginal improvement valuation model. Target ecosystem is ORB optimized for economics (commercial groups only) and based on 1950. Various weightings on economic and SB rebuilding optimization criteria provide a spread of possible restoration plans, yet all achieve a reduction in the sum of squares to some degree versus the target ecosystem. The vertical spread of points along the Y-axis indicates runs optimized for economic returns at top, and runs optimized for strict rebuilding at bottom.

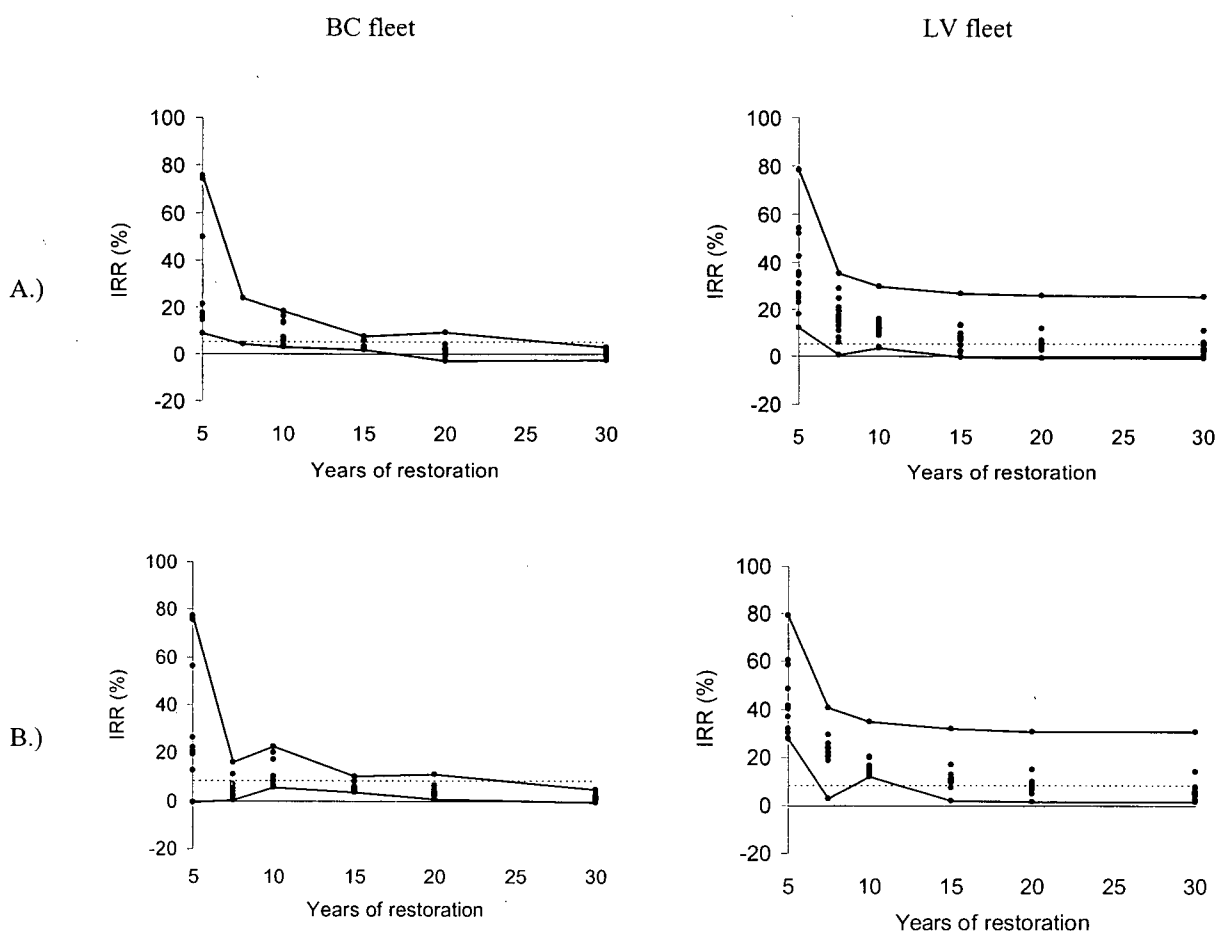


Figure 7.23 Internal rate of return (IRR) making restoration/harvest scenarios economically worthwhile. Fleets used are BC fleet (left) and *lost valley* fleet (right). Costs associated with rebuilding are incurred during the rebuilding phase, and benefits of rebuilding are accumulated throughout the equilibrium harvest phase. Rebuilding phase lasts 5, 7.5, 10, 15, 20 or 30 years (X-axis); equilibrium harvest phase continues until year 100. The profit stream predicted by each restoration scenario is evaluated under two discounting models, conventional and intergenerational. A.) Conventional discounting model. Broken line represents alternate rate of return from bank interest ($\delta = 5\%$). B.) Intergenerational discounting model. Broken line represents alternate rate of return from bank interest, which has been converted to the equivalent intergenerational rate (δ and $\delta_{fg} = 8.3\%$). All optimizations use biomass as the unit of improvement, and employ a linear marginal improvement valuation model. Target ecosystem is ORB optimized for economics (commercial groups only) and based on 1950. Various weightings on economic and SB rebuilding optimization criteria provide a spread of possible restoration plans, yet all achieve a reduction in the sum of squares to some degree versus the target ecosystem. The vertical spread of points along the Y-axis indicates runs optimized for economic returns at top, and runs optimized for strict rebuilding at bottom.

7.4 Discussion

SB algorithm

To effect restoration in commercial functional groups, as is required for the economic ORB solutions and for profitable solutions in general, the SB algorithm must achieve a gross increase in commercial biomass. Therefore, using biomass as the unit of improvement offers the most potential from an economic perspective. Likewise, the most accurate manipulation of biomass towards target levels occurs using the linear marginal improvement valuation model.

Rebuilding heavily depleted populations becomes less of a concern when the restoration policy focuses on improving biodiversity, as with the biodiversity ORB. In this case, the mixed unit of improvement allows improvement in both species evenness and richness. This is evident by the fact that the Q-90 index (which contains reference to both biodiversity qualities) is satisfied best under this setting, while the Shannon-Weaver index (which considers evenness alone) is satisfied best using percent change as the unit of improvement. In that case, the policy affects a maximum number of functional groups (Fig. 7.6). Fig. 7.13 confirms that Shannon-Weaver biodiversity can be improved more easily by the new algorithm than Q-90 biodiversity. This is probably due to the fact that species evenness can be improved by the intentional depletion or re-growth of group biomasses, while species richness can only be improved through biomass growth.

Complex optimizations

Edits were made to the Fletcher optimization routine in EwE to increase the maximum number of gear-year blocks searchable by the policy search routine. I hoped that complex optimizations using decadal or yearly time blocks would reveal multi-stage restoration scenarios, where distinct fishing regimes could be seen preparing the ecosystem for succession by an artificially increased table fish population. However, complex optimizations were abandoned for this work as a bug in the policy search routine became evident. Since this work, an error has been identified in the way that parameters are scaled when estimating the numerical derivatives in the

Fletcher-Powell optimization routine (C. Walters, UBC Fisheries Centre, *pers. comm.*). The bug has now been fixed (August, 2005) and a corrected version of the policy search routine will soon be available in an upcoming EwE release.

All the optimizations conducted in Chapters 6 and 7 therefore use a 'single block' optimization – a single fishing mortality assigned to each gear type and held constant throughout all simulation years. Revisions to this work could explore the use of complex, multi-staged restorations plans, which would likely be able to provide more effective restoration than has been presented here using equilibrium-level fishing mortalities. If we can achieve a more effective restoration, it will become possible to compare restoration ORBs based on the 1900 and 1750 ecosystems in addition to the 1950 solutions tested here. Unfortunately, equilibrium-level fishing mortalities can only propel the 2000 system a short way towards these lofty restoration goals.

A multi-staged optimal fishing solution that could potentially be identified by complex optimizations is a cyclical plan that alternates between pulse fishing events and re-growth of target fish populations. It has long been known that periodic pulses of harvest may be an effective strategy under certain circumstances to maximize yield, as was shown for terrestrial systems (Walters and Bandy, 1972). This may apply to fisheries as well (Clark *et al.*, 1973, Hannesson, 1975). It has been demonstrated that pulse fishing can increase yields in simplified ecosystem models based on lake-locked salmon populations (C. Walters, UBC Fisheries Centre, *pers. comm.*). As in agriculture, when fields are left fallow between harvests, it is hypothesized periodic fishing can increase yields if the target species has, first, the potential to over-eat its food supply, and second, the potential to stimulate growth of predator populations. Complex calculated fishing patterns may hold the potential to improve the profitability of rebuilding schemes through strategies not yet described, and make restoration more attractive economically.

Complex restoration schemes may be identified using the SB algorithm that would advance the ecosystem into an alternative stable state desirable from a policy perspective. It is theorized that non-linear and hysteretic change may prevent a complex marine system from reverting to its wilderness state, once fishing pressure is removed (Scheffer *et al.*, 2001; Hughes *et al.*, 2005). However, such a recovery may be possible through intentional, directed manipulation of the food

web. The obvious Canadian application would be to examine Northern cod recovery strategies. Through that work it would be possible to test regime-shift hypotheses that explain the recovery failure (e.g., environment, mammal predation, poaching/bycatch; Rice and Rivard, 2003, compensation/depensation; Walters and Kitchell, 2001). If trophodynamics, rather than the environment, is responsible for the persistent alternative state, then we could draft ecological recovery plans that use direct and indirect trophic effects as tools for ecosystem restoration. In the case of northern BC, a pre-industrial assemblage may only be achieved through careful manipulation of keystone groups and maintenance of key trophic interactions (e.g., sea otter-kelp-urchin triad; Estes and Duggins, 1995; Steneck *et al.*, 2002). We now have a method to evaluate these ambitious policy objectives in a quantitative and predictive way.

However, using the policy optimization routine to locate complex solutions would present new challenges. For example, the optimization may be unable to find a radical solution as may be required to reverse a severe change in community state, if it requires a pointed readjustment of fishing activity too extreme for algorithm to locate. Firm knowledge of ecosystem behaviour would be required only to guide the optimization to a plausible solution. A deep understanding of ecosystem functioning would clearly be necessary to implement such a policy. If the optimal solution required substantial modification of current fishing practices, then social and political resolve may be a greater barrier to implementation.

Restoration time

The limitation in the policy search routine described above prevented any detailed analysis of the effects of restoration time. Most of the benefits of restoration are achieved quickly in these rebuilding plans, first from an initial redistribution of fishing effort that reduces group biomasses towards their targets, and then from subsequent growth in the remaining groups. Generalizing, most restoration scenarios achieve maximum success within 20 years, and then the new biomass equilibrium is reached. Longer restoration periods do not confer any additional advantage when using these equilibrium-level solutions. By increasing the complexity of the optimizations, longer restoration periods would have the potential to achieve a greater increase in harvest benefits and/or agreement with the target ecosystem. Due to this limitation, all plans tested in

this chapter involve fifty-year optimizations, which were assumed long enough to achieve an equilibrium level fleet-effort configuration.

Fleet structure

It is demonstrated that the *lost valley* fleet is able to achieve better restoration to the economic ORB and biodiversity ORB than either the BC fleet or the *maxdex* fleet, despite having fewer gear types and therefore less precision in ecosystem restructuring (see Fig. 7.17). This is probably related to the fact that the *lost valley* fleet was itself used to determine the ORB configurations (Chapter 6). That assertion is supported by the fact that the *maxdex* fleet achieves a far better restoration solution than the *lost valley* fleet when restoring the historical system (which was not derived from a previous optimization), or when restoring ORB biomasses for commercial groups only (in which case the specific catch composition offered by the *lost valley* fleet imparts no particular advantage).

Any biases in the ORB structure that reflect innate properties of the host fleet will be most easily replicated by the policy search routine if it uses a similar fleet for restoration. It is therefore important to tailor the ecosystem goal with practical limitations of the proposed fleet structure in mind. It is otherwise difficult to imagine what benefit the *lost valley* fleet can offer in principle over the *maxdex* fleet, since the *maxdex* fleet can achieve any degree of species selectivity (even to the point of matching the *lost valley* catch composition exactly). I conclude that there are computational limitations in the Fletcher optimization routine that prevent it from finding the *maxdex* fleet-effort configuration that approximates *lost valley* fleet behaviour.

The species selectivity of the gear structure has a large impact on the effectiveness and profitability of the of restoration plan. As Larkin (1996) observed, the existing fleets are a blunt instrument for fine tuning the relative abundance of species, and bycatch is a confusing influence on the species mix. However, the work in this volume suggests that immediate costs to improve the selectivity of fishing gear may be worthwhile in the long-term, if the upper-bound to ecosystem profitability is increased.

Quantifying trade offs

The analysis in this chapter supports the finding of Ainsworth and Pitcher (*in review*. Conference proceedings: 4th World Fisheries Congress, Vancouver, BC.)¹⁷. There exists a convex trade off between end-state profit and biodiversity inherent in some rebuilding plans. Although relative benefits received depend greatly on the fleet structure and the goal used for restoration, there appears to be an optimal rate of restoration that corresponds to mid-range scenarios, which include some economic priority for harvest benefits. Restoration plans in this region will generate the greatest restoration benefits at the least cost to resource users.

Economic analysis

This chapter demonstrates that marine ecosystem restoration can pay for itself in northern BC, and can even outperform bank interest when viewed as an investment in natural capital. Restoration proceeds slowly under the most economically attractive scenarios, but continued benefits to resource users may make these solutions appealing as potential restoration plans. It should be noted that fishing the system optimally, alone, could offer a great improvement over current real-world profit levels since fisheries in BC are not currently optimal, nor directed by any overarching ecosystem-based strategy. Also, it is possible to achieve an improvement in profit, biodiversity, and any other desirable ecosystem value without adopting a plan to explicitly restore historic systems, or their ORB derivatives.

Fleet structure has a critical effect on the profitability of restoration, and the results from Fig. 7.21, 7.22 and 7.23 demonstrate that the *lost valley* fleet has a clear advantage over the BC fleet where economics is concerned. Switching the current fleet into a more responsible form like the *lost valley* fleet will entail certain economic and social costs not considered in this report, but these hereto unknown costs cannot be used as an excuse to defer restoration, since it is shown to be affordable even under the existing fleet structure.

¹⁷ Ainsworth and Pitcher (*in review*) used a preliminary form of the northern BC models based on Ainsworth *et al.* (2002), and a more primitive rebuilding algorithm.

A greater improvement in biodiversity can be achieved using the BC fleet than the *lost valley* fleet (Fig. 7.21). The BC fleet has more gear types than the *lost valley* fleet (17 versus 12), and so can produce a more even manipulation of ecosystem components as required by the biodiversity criterion. Equilibrium profits, in contrast, are determined through the action of fewer functional groups and so advantage tendered by the *lost valley* fleet becomes clear. Future revisions to the *lost valley* fleet should include a more detailed gear structure for accurate comparisons with the BC fleet, including at least an equal number of gear types. The reduced level of bycatch occurring in the *lost valley* fleet may then be expected to provide a clearer advantage in ecosystem manipulation.

Restoration policies containing an economic optimization criterion are shown to be affordable under conventional discounting, yielding a positive NPV that is greater than *status quo* exploitation (Fig. 7.22), and yielding an internal rate of return that is at least comparable to bank interest (Fig. 7.23). This applies even for scenarios budgeting for 15 and 20 years of restoration using the present BC fleet structure. Under intergenerational discounting the evaluated restoration/harvest scenarios are even more attractive since we consider bequest value in the calculation of the present value term, and since restoration plans leave a higher profit equilibrium than the *status quo* alternative.

8 CONCLUSIONS

He will win whose army is animated by the same spirit throughout all its ranks.

Sun Tzu.

Art of War. c.500BC Tr. Giles (1910)

8.1 Summary

In this thesis, mass-balance simulation models (EwE) were developed for the marine ecosystem of northern British Columbia (BC) for the historical periods 1750, 1900, 1950 and 2000 AD (Chapter 5). Time series data were compiled for catch, fishing mortality and biomass using fisheries statistics and literature values; additional analyses were conducted to improve estimates of biomass (Chapter 3) and catch (Chapter 4). Using the assembled dataset, dynamics of the 1950-based simulations were fitted to agree with observations over 50 years to 2000 through the manipulation of predator-prey vulnerability parameters and the addition of climate factors. The fitted vulnerabilities showed a positive correlation with predator trophic level, indicating wasp-waist trophic control. The climate factors reflected documented environmental series, most strongly sea surface temperature and the Pacific Decadal Oscillation index. Dynamic parameterization of the other historical periods was achieved by transferring the vulnerabilities from this 1950 fitted model, and assuming stationarity in density-dependant foraging tactics. The 1900 model exhibited an improved fit to data using this approach.

The historical models were next used to develop potential goals for ecosystem restoration based on the Optimal Restorable Biomass concept (ORB: Chapter 6). ORB ecosystems are designed to preserve the diversity and relative abundance of the original historical period, while providing sustainable benefits to resource users. New indicators were developed for use with EwE to explain the potential value of ecosystem restoration (Chapter 2). Using the new *specific biomass* algorithm designed for EwE's policy search routine, restoration plans were drafted that would

use selective harvesting as a tool to manipulate a depleted, present-day ecosystem into one resembling a more productive ORB state (Chapter 7). Various simulated ORB ecosystems based on 1950 were restored and compared to the historical ecosystem. Plans that restore biomass quickly tend to produce less profit and vice versa. However, I found a convex relationship between the amount of profit that can be sustained by the fishery and the biodiversity that can be restored in a given time, suggesting that there may be an optimal rate of restoration which corresponds to mid-range policies lending priority to both rebuilding and maintenance of socioeconomic benefits. With the historical 1950 period as our goal, that rate roughly equates to an average 30-50% reduction in current fishing effort for commercial fish groups, except halibut.

It was demonstrated in Chapter 7 that the current BC fleet, with existing gear-types, can be used effectively for ecosystem rebuilding, increasing species biodiversity and profitability of fisheries simultaneously. Reducing bycatch by use of a more responsible fleet structure greatly increases the benefits of restoration. Nevertheless, a cost-benefit analysis using the present-day BC fleet demonstrated that conservative restoration plans, lasting 10 to 20 years, and including some priority for continued harvest benefits, could offer a rate of return superior to bank interest when viewed as an investment in natural capital.

8.2 An ecosystem approach to management

In 1977, Peter Larkin made a prediction, "In another 20 years, the understanding of community dynamics may have proceeded to the point that we could be rather cute at manipulating species compositions while preserving the stability and qualitative integrity of aquatic communities." (Larkin, 1977). By his own review in 1996, he acknowledged that purposeful ecosystem manipulation was still beyond reach saying, "There may come a day when it is possible to create upwelling wherever and whenever desired, and to control which species will constitute a food web, but for the foreseeable future those choices are only available to the fish culturist" (Larkin, 1996).

As with terrestrial systems, human managers have limited control over climate and the environment, but there can be no argument that human beings strongly influence the composition

of marine food webs. We would be remiss as managers unless we strived to predict and mediate human impacts using the latest ecosystem tools. Moreover, a formalized and comprehensive strategy to ensure the best possible long-term use of ecosystem resources is overdue. The first aim of this strategy should be to introduce an overarching goal for management so that the livelihoods of people are not reliant on the dividends of a directionless regulation policy. Larkin (1996) agreed that the biological objective of ecosystem management should be to specify the species mix that is desired in the yield, albeit with wide variation related to year-class strength, and to optimize that mix for overall ecosystem benefit.

Now in 2006, many challenges remain before we can claim the ability to safely manipulate ecosystem structure to meet the needs of people. However, ecosystem-based approaches continue to develop rapidly, and we can expect quick advancement as they become more widely used. I hope that the contributions of this volume are a step in the right direction, and that they can provide a theoretical foundation for optimal system use.

This chapter will discuss the strengths of the Back to the Future approach, address methodological challenges, and suggest future lines of research. Some policy recommendations will be provided based on the general conclusions of this study.

Engaging stakeholders

Under this methodology, the expected socioeconomic and ecological qualities of an ecosystem restoration goal can be demonstrated during the policy planning phase. This should engage stakeholders, and allow allocation issues to be addressed before managers commit to a policy action. In fact, the technique introduced here for setting restoration goals is greatly strengthened by stakeholder participation. First, in modeling the ecosystem, since there are many unknowns that the experience of mariners can help us with. Second, in designing the fishing fleet that is to be used as a restoration tool. Third, through the vetting of optimal fishing strategies to determine which elements are socially acceptable.

To manipulate stock biomasses effectively and achieve higher sustainable catch rates, problems of gear selectivity need to be addressed (Chapter 7). Much of the improvement we have seen in recent decades in the reduction of bycatch is thanks to changes undertaken by fishers, in where and how they fish (Chapter 4); an effective restoration plan will require the continued contribution and innovation of fishers. Much more so will the process of achieving restoration require stakeholder consent, since an ecosystem approach demands the coordination of many industries and user groups. The BTF project has so far attempted to integrate community knowledge into the modeling (Chapter 3) and stimulate participation among stakeholders. Perhaps this will foster a sense of ownership in the process of setting restoration goals.

The methodology needs only to be scaled up to be a significant new policy aid. Fortunately, interaction with communities is increasingly common in western fisheries management, and we may expect local ecological knowledge to become a more important resource for scientists and managers as regulatory budgets are cut in response to declining stock value.

Optimal restorable biomass (ORB)

It is a reasonable biological objective for EBM to specify the desired species mix in the ecosystem, since fishing commonly changes the relative abundance of species from the natural state (Larkin, 1996). The candidate goals presented here as ORB targets take this important step in a quantitative way. The species mix designated by the ORB ecosystem can be used to gauge progress towards a specific restoration goal, which sets tangible benchmarks for management. The optimization procedure also ensures that the needs of resource users and the environment will be explicitly considered in the policy decision.

A whole-ecosystem analogy to B_{MSY}

MSY (and its associated biomass, B_{MSY}) is used world-wide as a management goal for single species fisheries management, despite well understood shortcomings. By capturing only the quantity of fish allowed by MSY, it was once believed by many that the sustainable take could be maintained year after year. Like spending only the interest on an investment, the principle, in

this case a living resource left in the water, would never diminish. However, MSY has been applied in management beyond the intentions of its creators. Related indices, for example maximum economic yield (MEY) and optimal sustainable yield (OSY; Roedel, 1975), address some of its deficiencies. But it was not until the development of ecosystem models that an optimal quantitative solution could be proposed that adequately takes into account trophic interactions and other technicalities missing from the MSY concept.

The origin of the MSY concept, beginning in the 1930s especially with papers by Russell (1931) and Graham (1935), is reviewed by Larkin (1977) and Punt and Smith (2001). It was developed during a period when management for single species was appropriate, since a small number of species were exploited, and since the exploitation levels were usually low enough so that interacting species were unaffected (Sissenwine, 1978). That is no longer the case, and attitudes are shifting to favour ecosystem-based approaches (Browman and Stergiou, 2004).

Table 8.1 Criticisms of MSY and their applicability to the ORB concept. An ORB 'report card' (right column) shows how well these criticisms are addressed by the ORB concept. From best to worst: A(+/-), B(+/-), C(+/-).

MSY criticism	ORB response	ORB 'report card'
1. Does not reduce the risk of catastrophe from compensatory mortality.	ORB solutions can be designed to prohibit biomass depletions. Risk of depletion can be assessed based on data uncertainty and climate variation using Ecosim's Monte Carlo routine. However, the Monte Carlo routine cannot yet accommodate wide confidence intervals for input data.	B-
2. Ignores changes in catchability resulting from biology or technology.	Optimal ORB fishing mortalities may be inflexible, and managers will need to regulate fishing effort with the tools already available. Although, the EwE models are flexible enough to be used in adaptive management, and ORB solutions can be selected to be robust against unintended increases in fishing mortality.	C
3. Ignores multispecies interactions like predation and competition.	Multispecies effects are considered explicitly by ORB solutions. Non-trophic effects can also be represented.	A+
4. Is not economically optimal.	ORB solutions explicitly consider value and fishing costs. However, fleet decommissioning costs need to be accounted for more carefully.	B
5. Ignores social values.	ORB solution can explicitly consider social harvest values, however the social objective currently in use is quite simple (assumed proportional to catch value).	B
6. Overestimates average yield under stochastic uncertainty.	Restoration simulations can be subjected to variable climate regimes, similar to population viability analysis (e.g. Pitcher <i>et al.</i> , <i>in press</i>). This was not done here. See text, section 8.2: Equilibrium assumption.	C+
7. Eliminates less productive sub-stocks and genetics.	Ecopath functional groups can be designed to consider subpopulations explicitly. Depletion risk can be determined in Ecosim, and mediated by the ORB configuration.	A-

Despite early warnings that MSY was no silver bullet for achieving the sustainable exploitation of stocks, fisheries management bodies throughout the world have adopted the concept. It first gained acceptance as a management goal in the late 1950s; it was widely used, but fell under intense criticism in the 1970s. The 1982 United Nations agreement, the Convention on the Law of the Sea (UNCLOS, 2005) endorsed MSY as a management objective, and this encouraged its use by member countries. The success of MSY in establishing itself into the bureaucracy of world fisheries management probably has less to do with its biological merits, and more to do with its ease of interpretation, the (false) promise of catch constancy, its accessibility under data-limited conditions, and the fact that total allowable catch limits (TAC) are set by the biology of the stock, resolving conflict between regulators and industry.

Major criticisms levied against MSY are listed in Table 8.1, and comment is provided on how well the new ORB goal addresses these concerns. The methodology presented in this paper, including ecosystem modeling, multi-criterion optimization and evaluation of uncertainty helps to resolve some of the problems associated with MSY. More discussion regarding MSY and its shortcomings can be found in Larkin (1977; 1996), Sissenwine (1978), Punt and Smith (2001), Mace (2001) and Walters *et al.* (2005).

Equilibrium assumption

It should be noted that ORB as a management goal is an equilibrium concept, and therefore subject to some of the same criticisms as MSY. The fishing policies developed in this volume assume deterministic fish growth and mortality. However, natural fluctuations in fish stocks will necessitate that the optimal fleet-effort vectors presented here represent only an average long-term fishing solution. Natural variations in population production and biomass will cause the equilibrium-level fishing mortalities to be sub-optimal in any given year. However, EwE has some capacity to assess the implications of environmental stochasticity, and new techniques are in development that may soon allow us to consider it in the optimal ORB solution.

EwE can represent annual climactic forcing factors on primary production, stock recruitment and other parameters. Previous authors have modeled the effects of variable ocean regimes and stochastic environmental factors. Pitcher and Forrest (2004) challenged the northern BC 1750 model of Ainsworth *et al.* (2002) using transformed annual temperature data from tree rings (Gedalof and Smith, 2001). Two other studies tested predictive forecasts, like those prepared for Chapter 7, using annual climate drivers based on real environmental data. The effects of future climate variability were estimated through an ecosystem based population viability analysis for fish populations in Lake Malawi (T.J. Pitcher and E. Nsiku, UBC Fisheries Centre, *pers. comm.*). In that work, primary productivity was driven by randomized data obtained from biogenic silica deposits (Johnson *et al.*, 2001). More recently, Pitcher *et al.* (*in press*) examined the effect of potential climate fluctuations on harvest policies, where variability in climate is based on randomized dendroclimatic records. Those authors examined several ecosystems, including northern BC based on the EwE models developed in this volume. Micropaleological evidence of fish abundance from scales (e.g., Walker and Pellatt, 2003) could similarly be used to validate predictions made by historic models of northern BC.

Criticisms of the EwE approach

Review papers by Christensen and Walters (2004a) and Plagányi and Butterworth (2004) describe shortcomings of the EwE approach. This section reviews only fundamental issues that affect BTF modeling.

Representing abiotic influences

EwE primarily models trophic interactions. Some facilities exist to represent non-trophic effects, such as facilitation and protection. Predator and prey behaviour are also modeled through the vulnerability matrix to some extent. Ecospace adds more realism by representing current movements and habitat quality. Most other abiotic influences are difficult to represent in EwE without resorting to a black-box production modifier.

The effects of salinity, temperature, nutrient and oxygen concentrations can be represented explicitly in an ecosystem model, as well as the microbial loop and more detailed transport processes (e.g. Atlantis: Fulton *et al.*, 2003; LakeWeb: Håkanson and Gyllenhammar, 2005). However, it is not clear how well trophodynamic modeling alone can be made to reproduce system-wide dynamics, especially over time scales important to BTF modeling. Therefore, I restricted the detailed analysis of restoration to 1950 and its ORB derivatives. Restoration plans targeting 1750 and 1900 are more likely to be compromised due to a poor representation of environmental and abiotic conditions during those periods.

Subjectivity in modeling

The behaviour of EwE models is certainly influenced by the bias and prejudices of the modeler. An EwE model is largely founded on empirical data, but it is relatively easy to manipulate dynamics by altering parameters that have no empirical support. Model structure also affects the dynamic output (Fulton *et al.*, 2003; Pinnegar *et al.*, 2005), and it represents a subjective interpretation of the ecosystem. However, an impressive collection of theoretical studies now uses EwE to explain real biomass observations. Christensen and Walters (*in press*) list 28 models that have been fitted to time series data. Neither the present analysis or similar studies have necessarily explained any real ecosystem dynamics, but they are able to put forward plausible explanations to relate mortality and production trends seen in recent decades.

For any given ecosystem model, a similar fit to data, in terms of a least square criterion, could be achieved by several different means. For example, the data fitting procedure outlined in Chapter 5, from an objective standpoint, utilized predation mortality to a great extent to fit model predictions to data. As a result, little additional improvement in the sum of squares was obtained through application of climate forcing patterns (e.g., Fig. 5.3). Any further attempt to attach mediating functions to improve the dynamic fit or to add additional climate forcing patterns would likely achieve only a small reduction in residuals, since model behaviour is now mainly resolved through 'hard-wired' predator and fishery dynamics. However, a similar fit to data could have been achieved in other ways. For example, applying forcing patterns earlier in the

process would ensure that more of the model's discrepancy with reality is explained through climate factors.

There are almost an infinite number of ways to fit a model to data. Few solutions, if any, may accurately reflect what happened in the past. However, I argue that broad-scale trophic impacts can usually be forecast, and there are several indications that real ecosystem dynamics have been revealed in this work. However, even if the models can be trusted to recreate past events, the past may be a poor predictor for the future.

The assumption of stationarity

Constructing Ecopath models of historic ecosystems presents some difficulties. These problems are not intractable, but simplifying assumptions must be made (see reviews by Pitcher, 2004; Pitcher and Heymans, 2004). When we move into dynamic simulations, additional challenges arise, such as choosing an appropriate trophic flow model (i.e., consumer versus nutrient/donor driven). With models of the recent past, time series catch, abundance and biomass data can be used to tune dynamic behaviour (e.g., this volume; Cox *et al.*, 2002; Stanford, 2004; Christensen and Walters *in press.*). However, predictions made into the future using EwE have always relied on the assumption that certain parameters will remain unchanged and applicable throughout the simulation time horizon. In the case of BTF, the behaviour of ancient ecosystems is also subject to this assumption.

However, there is some evidence that the ecosystem of northern BC behaves differently now from the ecosystem of the 18th and 19th century. For example, tree ring data suggests that strong oscillations in the PDO seem to be a 20th century phenomenon (Francis *et al.*, 2001). However, Chapter 7 did test the 1900 system behaviour using available time series, and found encouraging results. The assumption of parameter stationarity, at least with regards to the critical vulnerability matrix, seems to be a reasonable one because dynamics are predicted accurately by the 1900 model using trophic flow parameters adopted from the 1950 model. Nevertheless, uncertainty regarding past ecosystem behaviour should be considered when setting restoration benchmarks. This was attempted here through use of Monte Carlo simulations (Chapter 6), but

more could be done to improve the prediction of policy consequences given uncertain and variable climate factors.

Economic assumptions reconsidered

If the biology represented in the northern BC models is credible, then this report has demonstrated that ecosystem restoration is possible from an ecological point of view. However, economics may prove to be a greater adversary to the restoration agenda. Chapter 7 contains hints that restoration becomes surprisingly affordable under intergenerational valuation of living resources, but to evaluate the feasibility of an ecosystem restoration project on a scale as large as northern BC, a more thorough economic analysis is warranted.

The effect of changing prices on fisheries rent will have an impact on ORB solutions, and much more could be done to improve the bioeconomic model used by the policy search routine. Especially, the malleability of fishing capital should be considered more carefully. Some fisheries scientists and managers view overcapacity and overcapitalization as the single greatest threat to the long-term viability of fish stocks (Mace, 1997; Gréboval and Munro, 1999; Ward *et al.*, 2001). However, in this work I assumed there was no cost or penalty associated with fleet restructuring or decommissioning. The impact of this economic consideration must be carefully featured in any practical application of this methodology. Work is presently underway to incorporate a fleet buy-back scheme into the optimization procedure (W. Cheung and R. Sumaila, UBC Fisheries Centre, *pers. comm.*). The ORB calculations in this report could benefit from such an analysis.

Social values reconsidered

The social value of fishing should be defined more completely in the simulation models. Optimal fishing rates for rebuilding stocks are determined using the present methodology, but restructuring the fishing fleet holds great implications for resource users, stakeholders and people whose social and cultural values depends on a healthy and viable marine environment. The

social impact of shutting down fisheries to allow rebuilding may be more disruptive to society than the economic loss suffered by the fishing industry. A restoration policy, like the ones presented in this volume, could only be implemented in practice if we had a clear idea of the likely social impacts. Unfortunately, EwE quantifies social benefits of fisheries in a very basic way, by considering only direct employment in the fishery.

Instead of the simple assumption made by EwE, that employment in the fishing industry is directly proportional to the catch value, one could easily argue that the number of jobs supplied by fisheries can follow a step function, where the employment level remains constant over a wide range of catch and value. There may also be an upper limit to the number of jobs set by the physical space on the boats, total capacity of the fleet, space limitation in fishing areas, etc. The existing EwE code also predicts an instantaneous and linear change in employment with fishery revenues. For many fisheries, these may not be valid assumptions. In some cases, there may be a lag in the industry's response to changing conditions of profitability. If economic and sociopolitical factors restrict entry of participants or create a gold rush mentality, then the assumption of a linear change in employment with revenue will also be invalid. The optimization procedure to maximize fishery jobs could be improved by making the social objective function more flexible, and introducing gear-specific employment models. This would be a relatively simple addition to EwE code.

However, there is no easy way to represent complex social values beyond sheer employment numbers. Future work could perhaps consider the issue in more detail by using economic proxies for social benefits. Coastal real estate values are an example of one such proxy that could be used to judge the social value of the marine system (R. Ommer, University of Victoria, *pers. comm.*). A detailed spatial analysis may also contribute useful information; for example, knowing where fishing occurs would allow us to consider issues of safety and convenience. However, to quantify something as subjective as social and culture values of fisheries and a healthy ecosystem, an encompassing approach would be required; the analysis would need to expand beyond the simple bioeconomic approach used here. It may include surveys, interviews and sociological studies. The methods of integrative research initiatives like Coast Under Stress

[www.coastsunderstress.org] and existing government-stakeholder consultations, could be utilized to make the optimal harvest plans more beneficial and acceptable to the public.

Ecological limits to restoration

An objection made of the BTF approach is that marine ecosystems may not easily 'rewind' to historic states under time scales relevant to fisheries management. The main criticisms relate to issues of climate change and regime shifts. For example, if trophic energy flows up the food web have become more linear and simplified, then the energy budget of coastal marine ecosystems may no longer support a broad diversity of specialized predators. If directional climate change has occurred since historic times, then restoration policies directed towards historic states will be fighting a natural shift in the assemblage. Concerns have also been raised regarding the reduced fitness of populations due to founder effects, evolutionary change in response to fishing, and irreversible species introductions. The problem of extinctions is also recognized (Pitcher, 2004), and the loss of locally adapted populations and keystone species (Pitcher, 2005).

These are ecological limits to restoration, and some of them will never be overcome. Others will require the use of new and existing techniques that were not described in this volume. However, new ecosystem tools are providing us with our first opportunity to consider some of these obstacles and to learn from them. For example, I suggested in Chapter 7 that non-linear and hysteretic changes in ecosystem structure may be reversible if the persistent stable state is held in place by trophodynamic relationships, and not environmental factors. Such a plan could use staged depletion and succession events to force the ecosystem into an alternative stable state beneficial to stakeholders.

Considering climate change

In the North Pacific and elsewhere, climate shifts affect the biomass and composition of harvested species (e.g., Ware, 1995; Barenge, 2002; Chavez *et al.*, 2003). Changing temperature regimes can alter the species assemblage (McFarlane and Beamish, 2001; Benson and Trites,

2002) because temperature influences the distribution of species (DeYoung and Rose, 1993). For example, temperature variations have been linked with groundfish distributions (Perry *et al.* 1994), and it is warned that a modest rise in temperature could drive sockeye salmon from BC waters (Welch *et al.* 1998).

Although global warming may ultimately change and degrade the coastal marine ecosystem at its foundation, we also must consider in our ecosystem management plan the more immediate potential for ocean-basin-scale regime shifts. These shifts tend to cause extreme temperature changes that can persist for decades (Steneck *et al.*, 2002). Unfortunately, the full extent to which climate affects marine populations is not yet understood (Parsons and Lear, 2001), and if the effects of climate change on marine populations are non-linear (e.g., Hare and Mantua 2000) than the policy implications may be difficult to predict.

However, problems associated with interpreting the variable effects of climate challenges conventional stock assessment (Hofmann and Powell 1998) as well as ecosystem science. Yet ecosystem models can offer a useful tool to predict the consequences of climate change beyond traditional approaches. For example, trophic impacts of a temperature-induced influx of warm water species have been modeled in Ecosim (the "Russell cycle" in the English Channel: Russell, 1935; Southward *et al.*, 1995; Stanford and Pitcher, 2004). A similar approach could be used to predict the future of fisheries that rely on periodic, temperature-driven movements of fish (e.g. hake in southern BC). A northward distribution shift of warm water species may be foretold by seasonal movement patterns during brief warming interludes caused by El Niño (McFarlane and Beamish, 2001).

If species distribution changes because of global warming, then Pacific ecosystems southward of northern BC may serve as a spatial analogue to what the future holds. Like using the past as an analogue for the future in the BTF approach, we may account for broad changes in the species assemblage and forecast succession caused by climate change. See Pitcher and Forrest (2004) for a suggested method. The analysis needs to carefully consider possible interactions and nonlinearities. For example, if predators lag behind their prey in the succession, then the species complex may not be comparable across latitudes until equilibrium is reestablished, if at all.

8.3 The developing role of ecosystem models

Traditional stock assessment models may never be surpassed in their ability to help set harvest regulations. Ecosystem models are in their infancy, and there are certainly few examples, if any, where they are actively used in fishery management. Issues arise due to the complexity of ecosystem models (Fulton *et al.* 2003), but this is not necessarily a barrier to their implementation considering the vast data requirements of single species stock assessment models typically used in management (Aydin and Friday, 2001). Nevertheless, multiplying those data requirements up to the ecosystem level is not feasible, and simplifying assumptions need to be made when considering the entire ecosystem (Christensen and Walters, *in press*). Choosing what variables to omit is critical if we wish to capture only the processes most relevant to ecosystem function (Monte *et al.*, 1996).

Larkin (1996) postulated that the next generation of ecosystem modelers may be “armed to the teeth with computing *savoir faire*”, enabling them apply known aspects of ecosystem ecology more effectively in fisheries management. Ecosystem models have seen much development in the last two decades. If we can trust their output, then they will allow us to see a broader picture of ecosystem functioning. They provide a new capacity to account for species interactions in a way that was impossible 20 years ago and they allow us to analyze changes in trophic structure holistically, in a way that single species models cannot (Trites *et al.*, 1999).

As the ecosystem based approach becomes more widely implemented, driven by shifting attitudes towards conservation and facilitated by advances in computing technologies, ecosystem science will require a better understanding of fisheries and climate effects on marine communities (FAO, 2004). Scientists from diverse disciplines will need to work together in order to breakdown walls that have so far divided marine science into disjointed sectors (Rosenberg and McLeod, 2005). In many cases, the legislation to monitor fisheries impacts on the ecosystem has developed ahead of the science (Fulton *et al.*, 2005), but new approaches like the one introduced in this volume may eventually go some ways towards satisfying the technical requirements of EBM.

8.4 Policy recommendations

The methodology introduced here best serves as a strategic guide to coordinate restoration efforts among interrelated stocks and fisheries. Simulations using EwE cannot provide sufficient advice on 'tactical' management actions by itself, and must be used in conjunction with traditional stock assessment methods (Christensen *et al.*, 2004b). If an ORB restoration policy drafted using the present method requires us to reduce the impact of a particular fishery, there is no practical guide provided here that would help managers decide the best way to do that. Whether through protected areas, time closures, modification of fishing gear, or through application of effort controls or property rights, the options for management are wide open. The specific methods we use to reduce fishing mortality will need to be considered on a case-by-case basis, and will need to be informed by experience gained with the particular stock. These are technical challenges that can be solved using existing tools and techniques.

General policy recommendations can be made regarding current fisheries management in northern BC based on the findings in this study.

1. *Enact policies that curtail further loss of biodiversity.*

With the 1750 and 1900 models, maintenance of biodiversity opposes high economic returns from fisheries. However, with the more depleted models, 1950 and 2000, the aims of conservation and management have converged. Results in Chapter 6 suggest that an economically optimal fishing plan will need to recover biodiversity for the maintenance of healthy predator populations.

2. *Spread out fishing pressure across the ecosystem.*

Whenever the ecology of the ecosystem is taken into account, optimal fishing policies tend to spread out fishing pressure among functional groups. In as much as this can be

done under the present market conditions, this will allow greater fisheries take with less impact on sustainability.

3. *Reduce fishing on low-production functional groups.*

Functional groups with low production rates, particularly inshore rockfish and sablefish, have exploitation rates today that are well above optimal limits - even for the pre-industrial models. Equilibrium analysis of the 2000 model indicates that rockfish exploitation in all groups is currently above safe limits.

4. *Reduce bycatch.*

The profit potential of the marine ecosystem increases dramatically when more selective fishing practices are employed. Reducing discarded and retained bycatch will increase the sustainable take from the ecosystem under an optimal harvest plan, and improve the economic outlook regarding restoration.

5. *Allow recreational fisheries to take a prominent role in conservation.*

Optimal fishing plans tend to rely mainly on recreational fisheries to generate wealth whenever depletion is undesirable. This occurs under ecological optimizations using 1750 and 1900 models, and under economic optimizations using the recent past models, 1950 and 2000. However, the sport sector needs to be better regulated and documented (Chapter 4; community interviews: Erfan, *in press.*). The cultural value of commercial fisheries must also be respected.

8.5 Concluding remarks

Regulation of anthropogenic influences, habitat restoration and species reintroductions: there are many paths that we might take back to the future. It is too late for Steller's Sea Cow (*Hydrodamalis gigas*), a large sirenian mammal of the NE Pacific driven to global extinction in

1768 (Domning, 1976). But it is not too late for sea otters, whose reintroductions on the west coast of Vancouver Island and on the central coast have seen impressive success (Riedman and Estes, 1998). It is not too late for humpback whales, whose resident population once numbering over 200 was extirpated by commercial whaling in the 1920s (Gregar, 2002). Although the resident population is lost, the species is making a slow, natural return to northern British Columbia (Gregar, 2002); anecdotal accounts place the wintering number at around 30 (Erfan, *in press*). It is not too late for Northern abalone, inshore rockfish, eulachon and other depressed stocks that were once significant commercial and cultural assets in northern BC.

If reversing the damage done by people holds any appeal to the public, then presenting the historic ecosystem as a restoration goal may provide us with a policy agenda that most stakeholders can accept. It may also satisfy the stated interests of First Nations groups to restore the natural ecosystem (Pitcher *et al.*, 2002b). Lucas (2004) and Jones and Williams-Davidson (2000) describe how the cultural and spiritual health of coastal First Nations is closely linked to the marine environment. Ecosystem restoration then could potentially have positive social effects for these communities that extend well beyond fishery employment numbers or dollar value. Choosing the past to inform our restoration goal may also combat the shifting baselines syndrome (Pauly, 1995) and help people understand that that marine ecosystem once supported an incredible diversity and abundance of animals. It can again.

Before capture fisheries become an icon of the past, and before we squander our childrens' inheritance, we will have the opportunity to correct some past mistakes. Depletion on the west coast of Canada is less severe than in many places of the world. If our privileged society cannot free up the resources needed to restore the natural environment, it is difficult to imagine where that plan stands a better chance of success.

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APPENDICES

Appendix 2.1 The Effect of Discounting on Fisheries

Intergenerational valuation of fisheries resources can justify long-term conservation:

A case study in Atlantic Northern cod (*Gadus morhua*)

Introduction

In a departure from what he called the 'standard economic argument for overexploitation', Clark (1973) proposed that depletion of the Grand Banks demersal fisheries may not be entirely due to problems of open access, or open competition among impoverished fishermen. It may be partly the result of discounting practices applied by fishing companies. For economic reasons, Clark suggests, a corporate owner of property rights might prefer extermination to conservation. If true, then a cost-benefit analysis of the harvest record may find depletion justified over a more conservative harvest strategy at a discount rate equal to the market interest rate.

Among other factors, the discount rate used in CBA reflects investors' time-preference for early consumption and delayed payment, and reflects uncertainties associated with the investment (Brennan, 1997). The human tendency to prefer early benefits can be called impatience, a trait possibly stemming from our own mortality (Fearnside, 2002). The time preference people choose to discount the flow of future benefits may act as a function of some distance measure that relates the investor to the recipient (Schelling, 1995; Azar and Sterner, 1996). Whether we are speaking of geographic, ethnic, cultural or temporal distance, investors would prefer to bestow benefits to those whom they consider more closely related, than to strangers. In this respect, discounting emulates human behavior, providing investors with an analytical means to make a value-based decision.

Yet, a CBA of education in Appendix 2.2 demonstrates that conventional discounting does not wholly capture human tendency. If an alternate investment were to promise a greater return than

a person's increased earning potential through education, we would expect children to be rarely educated to the highest levels. However, parents and society chose to do so. People, perhaps unwittingly, apply some form of intergenerational valuation – where assured benefit to one's children carries significant value in the present. For long-term environmental conservation to work economically, similar intergenerational consideration may be required.

A very low discount rate has been suggested as a means to protect the environment (Hasselmann *et al.*, 1997; UN-FCCC, 1997), though others have cautioned against this on the grounds that it may result in wasteful use of resources if the social rate of discounting is set arbitrarily (Fisher and Krutilla, 2002). Moreover, using a low social discount rate may blur the distinction between potential Pareto improvement, a measure of policy efficiency, and other legitimate policy-evaluation criteria such as distribution equity (Goulder and Stavins, 2002). Recently, alternatives to standard discounting have been proposed that limit discounting of future benefits (e.g., Heal, 1998; Weitzman, 2001; Nielsen, 2001). A method by Sumaila and Walters (2005) allows us to separate the discount rates we use to value benefits to ourselves (i.e., standard discount rate - δ) and benefits destined for future generations (δ_{fg}). Their approach is applied here.

The current study tests the ability of three intergenerational discount rates to preserve the cod resource from a pre-collapse perspective: one that is less than, equal to, and greater than the standard discount rate. The standard discount rate used here is equivalent to the market interest rate. A CBA of education informs us as to what may be considered an upper and lower estimate of δ_{fg} that society may be willing to apply in order to value benefits destined for future generations. The education of children is therefore taken as an example of a well-established multigenerational investment that people often make. By finding the apparent discount rate that parents and society use to value a child's education, I account for a variety of non-monetary benefits that may be considered by those investors. Similar benefits, I argue, could also apply to resource conservation where future generations are the recipients of today's investment.

Comparing conventional and intergenerational discounting approaches, I perform cost benefit analyses on five cod harvest profiles: the actual historic trend since 1985 followed by projected

post-collapse earnings, the conventional optimum estimated using an ecosystem model, and three intergenerational optima - maximizing net present value (NPV) when $\delta > \delta_{fg}$, $\delta = \delta_{fg}$ and $\delta < \delta_{fg}$. Although optimal scenarios generate less immediate benefit than the historic fishing pattern, they maintain higher resource abundance at equilibrium and so permit greater sustained yields over time. This case study is published in Ainsworth and Sumaila (2003; 2005).

Methods

Selecting the conventional discount rate

The conventional discount rate used to represent the rate of return for an alternate investment ($\delta = 10\%$) approximately corresponds to the average annual rate of return for Bank of Canada long-term (10+ years) marketable bonds from 1981 to 2001, which was 9.3% (Bank of Canada, Department of Monetary and Financial Analysis). Since the early 1980s saw anomalously high interest rates (e.g., 15.2% in 1981) I have instead used the average rate during this 20 year period.

Selecting the intergenerational discount rate

The intergenerational rates chosen for this evaluation were 5%, 10% and 15% (for cases $\delta > \delta_{fg}$, $\delta = \delta_{fg}$ and $\delta < \delta_{fg}$ respectively). These represent discount rates that society may be willing to apply in order to value benefits received by future generations. They are based on a CBA of education in BC (Appendix 2.2). The low value (5%) approximately corresponds to the IRR needed to make a PhD education worthwhile, and the high value (15%) approximately equals the IRR needed to make a grade 10 education worthwhile. Although the CBA of education in Appendix 2.2 ignores non-monetary benefits, I ignore other non-monetary benefits in the following analysis of the Atlantic cod fishery (e.g., ecosystem health, aesthetics). Ultimately, I argue that commercial considerations center foremost when deciding fishing policy.

Model parameterization

The EwE model used for this evaluation is based on Heymans (2003). It represents the ecosystem of the Grand Banks off Newfoundland as it appeared in 1985, prior to the 1992 Northern cod collapse (Canadian Department of Fisheries and Oceans (DFO) statistical areas 2J3KLNO). Minor modifications to the diet matrix and basic parameters were made in the static Ecopath base model to improve dynamic predictions, particularly concerning cod. When driven by historic fishing mortalities, the sum of squares between the predicted and observed biomass trends were reduced slightly for the ecosystem overall (~2%; based on residuals from 17 species groups), and much more for cod alone (~50%). See Heymans (2003) for data sources regarding fishing mortality and biomass time series.

In the dynamic model Ecosim, a global setting of 0.3 for the vulnerability parameters is used to emulate mixed trophic control¹⁸. This intermediate value is default in Ecosim, and it has been used often where better estimates are lacking. Following the methodology of Okey and Wright (2004), a stability analysis confirms that this value results in reasonably stable and conservative ecosystem dynamics (Fig. A2.1.1).

Determining optimal harvest patterns

The optimal fishing mortalities (F) per gear type that would maximize cod value over a 16-year harvest simulation are determined using the policy search routine in Ecosim (Christensen and Walters, 2004b).

To have the fisheries optimization routine return a harvest strategy geared for maximum cod catch, the search criterion is restricted to an economic optimization and cod are assigned a high price compared to other target groups in the model. This configuration causes the search routine to increase the cod fishery at the expense of other sectors. The procedure does not merely

¹⁸ The Ecosim model used in this section is based on EwE V5.0, which defines the vulnerability parameter as a flux rate linking vulnerable and invulnerable biomass pools. The global value used, 0.3, is roughly equivalent to 2 under a revised definition of the vulnerability parameter incorporated in EwE V5.1. The vulnerability parameter now indicates the maximum increase in predation mortality (Christensen and Walters, 2004).

regulate cod biomass to achieve maximum sustainable yield (MSY), it tailors the entire ecosystem to augment cod production, redefining MSY at an increased level. Through selective fishing, predators and competitors of cod are eliminated, while other species groups are preserved only in so much as they may support a large cod population.

The fishing mortalities per gear type maximizing NPV under conventional and intergenerational discounting approaches are identified by the optimization and applied to a 16-year dynamic simulation, using the 1985 model as a starting point. End-state catch equilibrium was maintained for another 24 years, totaling 40 years (consisting of two generations of an assumed 20 years each).

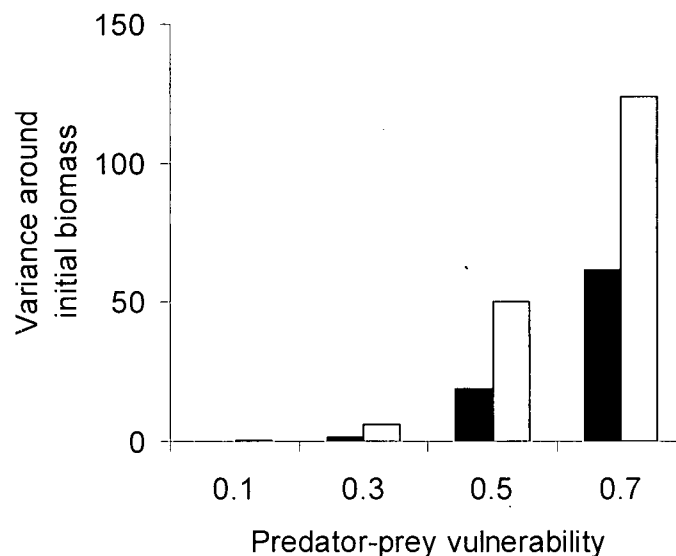


Figure A2.1.1 Stability analysis of dynamic ecosystem model. Mean variance of biomass change per species group is shown for a 16-year simulation. There is a non-linear increase in the directional responsiveness of biomass change with increasing predator-prey vulnerabilities. Black bars show simulations optimized for intergenerational benefit ($\delta = \delta_{ig}$); white bars show conventional discounting optima ($\delta = 10\%$). A global vulnerability setting of 0.3 for all species groups should provide a stable and conservative dynamic response; model dynamics become unstable when vulnerabilities exceed 0.5. Model adapted from Heymans (2003). Stability analysis after Okey and Wright (2004).

Calculating fisheries profits

The policy search routine maximizes the NPV of future earnings. If one applies a large discount rate, the optimal fishing mortalities are high, the population is aggressively harvested and landings increase to the point where future productivity is finally compromised. However, since greater harvests leave less standing stock biomass, which in turn increases harvesting cost, I introduce the following linear cost-abundance relationship to capture this effect (eq. A2.1).

$$NB_t = GB_t \cdot \left\{ 1 - C \cdot \left(1 + \left[\frac{B_{\delta > \delta_{fg}} - B_t}{B_{\delta > \delta_{fg}}} \right] \right) \right\} \quad \text{Equation A2.1}$$

Where GB is gross benefit (i.e., landed value) in year t ; C is base cost of fishing (assumed 60% of landed value based on *Anon.*, 1994); $B_{\delta > \delta_{fg}}$ is equilibrium biomass resulting from the optimum solution when $\delta > \delta_{fg}$ ($\delta_{fg} = 5\%$) and B is biomass in year t . Net benefit is therefore standardized so that costs equal 60% of gross benefit at the stock density left by the most conservative strategy, with cost of fishing increasing linearly as standing biomass is reduced from that level.

For all projections I assume a steadily increasing price for cod that reflects the trend from 1972-1992 (Fig. A2.1.2), based on historical value and landings data obtained from the DFO Department of Statistical Services, Ottawa (available online at <http://www.dfo-mpo.gc.ca/communic/statistics/commercial/landings/historical>). The trend is adjusted for inflation using the Bank of Canada consumer price index. I did not consider prices after 1992 in calculating the trend since the collapse of the cod fishery may have contributed to the jump in price after that year, whereas optimal solutions would have averted the collapse. I think this assumption is reasonable for the purpose at hand, even though it is worth noting that price formation and determination is a broad area of research activity, and that prices are determined by various variables. For instance, the price of whitefish is determined by variables such as the

income level of consumers, the availability of substitutes, the specific market in which the fish is sold, etc.

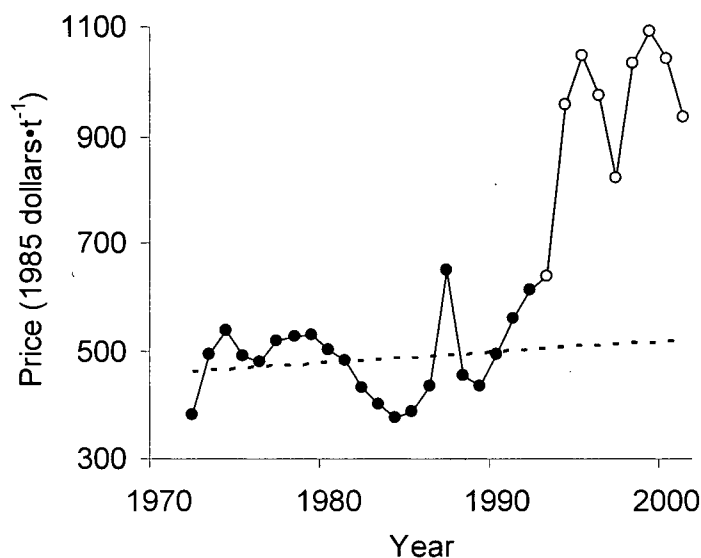


Figure A2.1.2 Real price of cod based on harvest from Atlantic Canada. Trend line (broken) shows linear price projection used in the analysis based on data from 1972 – 1992. The price increase after 1992 may have been influenced by the cod collapse, whereas optimal solutions would have averted the collapse. Source: Fisheries and Oceans Department of Statistical Services, Ottawa.

Results

Optimal policies

Harvest profiles optimized under intergenerational valuation return more conservative exploitation rates than those optimized under conventional valuation ($F = 0.172 \text{ yr}^{-1}$ [$\delta > \delta_{fg}$]; $F = 0.202 \text{ yr}^{-1}$ [$\delta = \delta_{fg}$]; $F = 0.229 \text{ yr}^{-1}$ [$\delta < \delta_{fg}$]; $F = 0.415 \text{ yr}^{-1}$ [conv.]), so, as might be expected, intergenerational policies maintain a greater standing stock abundance. Fig. A2.1.3 compares the stock biomass profiles optimized under these valuation schemes with the real-world trend. A Monte Carlo procedure was used to vary basic Ecopath parameters for biomass, consumption and production ($n = 20$; $c.v. = 0.2$); error bars show the effect of parameter uncertainty on the

model's biomass estimates. The real-world time series data is taken from VPA stock assessment (Ahrens 1999); 1985 (baseline) model estimate is based on Heymans (2003). The difference in end-state biomass between the conventional optimum and the intergenerational optimum may be blamed on the application of conventional discounting (A in Fig. A2.1.3). If we can assume that fishing consortiums operate at a discount rate equal to the market interest rate, then the difference in end-state biomass between the real-world profile and the conventional optimum may be blamed on ineffective management and/or environmental factors (B in Fig. A2.1.3).

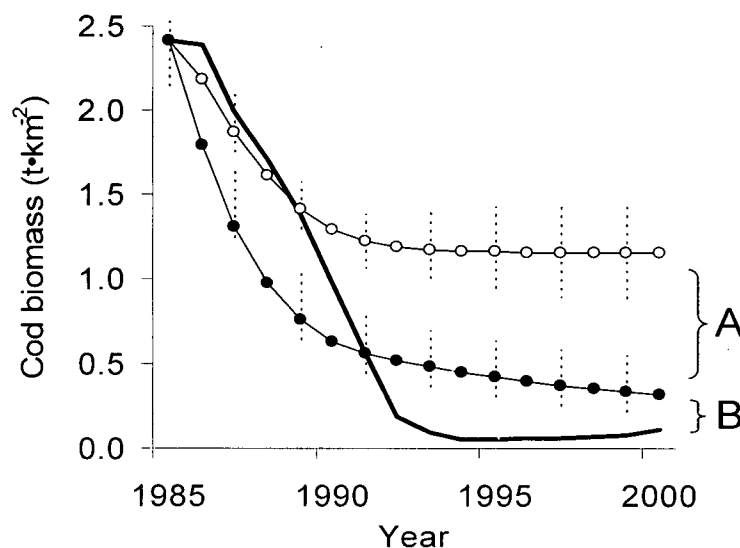
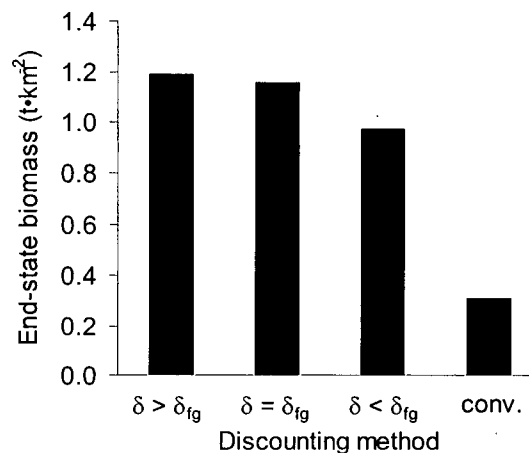


Figure A2.1.3 Historic cod biomass trajectory estimated from VPA versus EwE optimal trajectories. Solid line shows VPA biomass, open circles show intergenerational optimum ($\delta = \delta_{fg}$) and closed circles show conventional optimum ($\delta = 10\%$). Difference between end biomasses: (A) represents depletion that may be blamed on the application of conventional discounting, (B) represents depletion that may be blamed on environment factors or ineffective management. Error bars show one SD around the mean from a Monte Carlo procedure varying basic Ecopath parameters (biomass, production and consumption) for all species groups ($n = 20$; c.v. = 0.2). Model adapted from Heymans *et al.* (2003), VPA from Ahrens (1999).

I then show end-state biomass (year 16) of all harvest simulations (Fig. A2.1.4). Intergenerational solutions preserve the resource better than the conventional solution, and far better than the real-world harvest profile. Not shown, real-world biomass estimate in 2000 is $0.116 \text{ t}\cdot\text{km}^{-2}$ based on Ahrens (1999).



Comparing end-state catch rates between optimal solutions (Fig. A2.1.5), we see a curve analogous to a surplus production plot. At the lowest intergenerational discount rate ($\delta > \delta_{fg}$; $\delta_{fg} = 5\%$), the conservative optimal harvest plan does not fully take advantage of available production. The maximum long-term harvest rate is reached by the intermediate intergenerational solution (δ

$= \delta_{fg}$; $\delta_{fg} = 10\%$). At greater exploitation rates, such as those advocated by the most aggressive intergenerational solution ($\delta < \delta_{fg}$; $\delta_{fg} = 15\%$) and the conventional solution ($\delta = 10\%$), the standing stock is depleted to the point where productivity is compromised. Not shown, the actual end-state cod catch ($0.013 \text{ t}\cdot\text{km}^{-2}$) is based on year 2000 landings and estimated unreported discards from unpublished DFO and Northwest Atlantic Fisheries Organization (NAFO) records (Watson *et al.*, 2000).

Figure A2.1.4 Optimal end-state biomasses after 16 years of harvest under various discounting methods. Under harvest profiles optimized for net present value, the conventional discounting method (conv.) leaves a small standing biomass at $\delta = 10\%$ (right). Intergenerational optima ($\delta > \delta_{fg}$, $\delta = \delta_{fg}$ and $\delta < \delta_{fg}$) leave larger standing biomasses, especially at low intergenerational discount rates (left) ($\delta_{fg} = 5\%$, 10% and 15% respectively). Real-world biomass in 2001 is $0.116 \text{ t}\cdot\text{km}^{-2}$ (not shown).

Economic results

The NPV of harvest profiles based on the real-world cod dataset and optimal solutions is shown under conventional and intergenerational valuation (Fig. A2.1.6). Under intergenerational valuation the conservative strategies are worth more than the depleting real-world harvest profile. However, under conventional valuation, the real-world pillage outperforms the more conservative optimal plan. NB: All optimizations use a fixed exploitation rate throughout the 16 year harvest simulation, providing a long-term (equilibrium level) optimal solution.

The share of catch enjoyed by the first and second generations under intergenerational and conventional discounting models (generation time = 20 years) is compared with the real-world harvest profile, including projected catches until 2025 at current levels (Fig. A2.1.7). Benefit to the second generation carries considerable weight under intergenerational valuation, so a higher catch rate is maintained into the future.

Sensitivity analysis

The relationship between the discount rate and the equilibrium biomass for conventional and intergenerational valuation ($\delta = \delta_{fg}$) is shown (Fig. A2.1.8). At all levels of discounting, intergenerational optima leave a larger biomass at end-state than conventional optima.

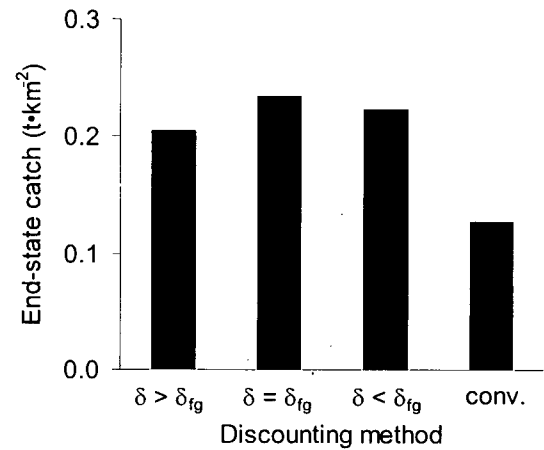


Figure A2.1.5 Optimal end-state catches after 16 years of harvest under various discounting methods. Under harvest profiles optimized for net present value, the conventional discounting method (conv.) depletes the system early, leaving only small catches by end-time (right), while intergenerational optima ($\delta > \delta_{fg}$, $\delta = \delta_{fg}$ and $\delta < \delta_{fg}$) maintain larger harvests to simulation's end ($\delta_{fg} = 5\%$, 10% and 15% respectively). Real-world catch in 2001 is 0.013 t·km^{-2} (not shown).

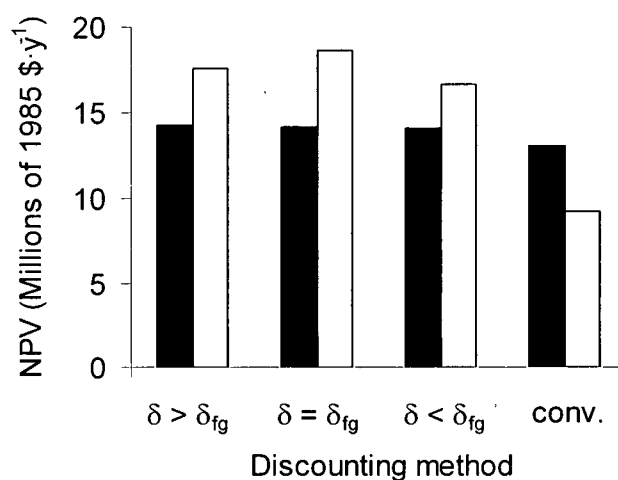


Figure A2.1.6 Net present value of 40-year harvest profile based on real-world data and optimum solutions. Black bars show actual NPV of fisheries; white bars show optimum solutions. X-axis varies discounting method. Under intergenerational discounting, the conservative optimal solutions outperform the depletory real-world harvest profile. However, under conventional valuation, early benefits make the exploitative fishing pattern observed in reality more valuable than an optimal (long-term) solution.

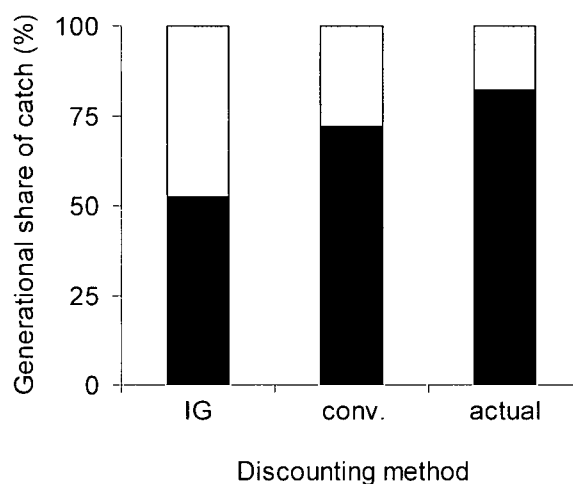


Figure A2.1.7 Generational share of catch after 40 years for three harvest profiles. Black bars show catch taken by first generation; white bars show catch taken by second generation (generation time = 20 years). The harvest profile optimized under intergenerational discounting (IG; $\delta = \delta_{fg}$; $\delta_{fg} = 10\%$) leaves almost an even split to each generation, while the conventional discounting optimum (conv.; $\delta = 10\%$) and the real-world harvest profile (actual) grant most catch to the first generation. Actual harvest profile includes projected catch to 2025 at current levels.

Discussion

Impatience of the individual is fundamental to cost-benefit decisions at all levels. It is an ingrained human attribute that allows us to instinctively account for uncertainty, lost opportunity costs, and other considerations relevant in resource acquisition. The economist's practice of discounting emulates this tendency; providing an analytical and quantitative guide for us to make value-based judgments. Yet under the conventional model of discounting it may be impossible to conserve a fisheries resource like Atlantic cod, or implement an environmental recovery strategy whose benefits may be years or decades in the coming.

Using education as an example of an existing and well-established multigenerational investment, parents and society seemingly disregard conventional financial wisdom and educate their children with little promise of return - except the confidence that they have equipped them with the tools needed to prosper. If that confidence carries value in the present, then this new form of intergenerational discounting provides a more accurate model of human valuation. It can be used to justify long-term environmental protection when investors are separated in time from recipients. As management agendas turn toward long-term environmental conservation, proactively or in response to fishery failures, we will need to change the way we value the stream of benefits derived from our living resources. As conventional discounting emulates an instinctive human behaviour, so too may intergenerational discounting emulate our willingness to support future generations.

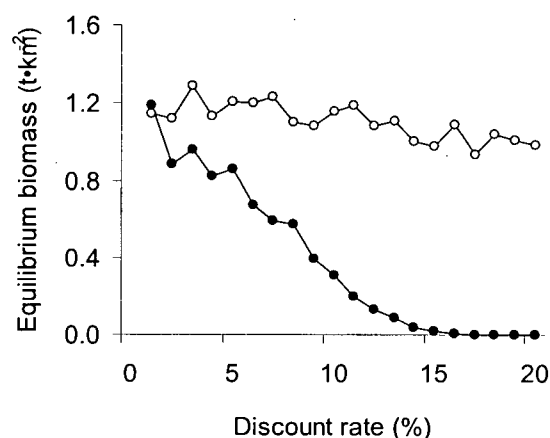


Figure A2.1.8 Sensitivity analysis showing the effect of discount rate on the optimal end-state biomass. Closed circles show optimal biomass under conventional valuation; open circles show optimal biomass under IG valuation ($\delta = \delta_{ig}$). Large discount rates leave less standing biomass in the ecosystem, but the intergenerational solution maintains the stock at higher levels than the conventional solution. The conventional solution advocates stock collapse at discount rates greater than $\delta \approx 15\%$.

Appendix 2.2 Cost-benefit Analysis of Education

As an estimate of the highest and lowest discount rate the public may be prepared to accept in valuing the benefits bestowed on future generations (δ_{ig}), I have chosen to use the internal rates of return needed to make a grade 10, and a PhD education economically worthwhile. Therefore, the following cost-benefit analysis may be what a taxpayer in 1981 would have used to compare the expected benefits from a child's education with that of an alternative investment. The value used to represent the rate of return for an alternate investment ($\delta = 9.3\%$) corresponds to the average annual rate of return for Bank of Canada long-term (10+ years) marketable bonds between 1981-2001 (GOC, 2002).

A child enters grade 1 at a BC public school in 1981. He or she graduates high school in 1992 at a total cost of \$54,307 paid for by the provincial government (BC, 1989; BC, 1990). The four-year arts or science undergraduate degree at the University of British Columbia costs \$50,192 (PAIR, 2002; UBC, 2002), which is paid for by tuition and government grants. Masters and PhD take six more years and costs \$80,430 (PAIR, 2002; UBC, 2002). After 22 years he/she has completed his education at a total cost of \$184,931 and begins earning income. As a national average, someone with a PhD level of training may expect to make \$59,000 per year (HRDC, 2002a), while a high-school dropout earns only \$21,000 (HRDC, 2002b).

Fig. A2.2.1 compares the costs and benefits of education to PhD and grade 10 levels. Each year's cost and benefit has been adjusted to 1981-dollar equivalents, and discounted to reveal the time preference for payment made far from the 1981 perspective. A grade 10 education outperforms the alternate investment, bank interest at $\delta = 9.3\%$ ($NPV = 19.9 \$ \cdot 10^3$), but a PhD education is not worthwhile ($NPV = -17.3 \$ \cdot 10^3$).

Table A2.1.1 shows discount rates where varying levels of education become worthwhile. A PhD becomes uneconomic at all discount rates greater than $\delta = 6.3\%$, but an incomplete high school education (grade 10) is worthwhile at $\delta = 15.0\%$. Although average annual income of the high school drop-out is only 36% that of the PhD, the lower level of education is more advisable

from a conventional cost-benefit perspective at the discount rate set by an alternative rate of return from bank interest. Total career length is also longer for a high school drop-out (the individual retires in 2032 in both cases).

This example represents a conservative estimate of the costs of education. Were this analysis repeated for private grade school the increased cost would weigh heavily in the evaluation because of its immediacy from the 1981 vantage point. Also, the early eighties saw very high yields in long-term marketable bonds (15.2% in 1981). If we had used this rate to represent an alternative investment rather than 9.3%, the average rate between 1981-2001, the returns from even the most modest education level would become less attractive than returns from the alternative investment rate.

This analysis disregards non-monetary benefits to education. However, even if the benefits to education are increased by 50% to approximate these considerations the discount rate required to make a PhD worthwhile is still 8.6% - less than the alternate rate of return.

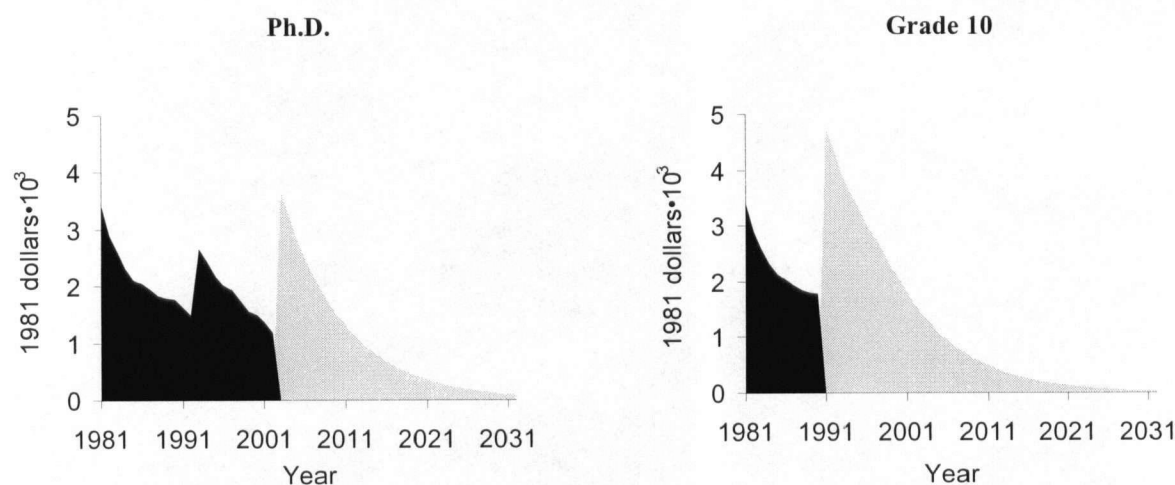


Figure A2.2.1 Costs and benefits of education in BC discounted from a 1981 time perspective. Education to a Ph.D. level takes 22 years, education to grade 10 takes 10 years; total career length is 50 years including education (black area) and employment (grey area). The Ph.D. makes more money once employed, but earlier benefits make grade 10 education a better investment at a discount rate that represents an alternate rate of return (bank interest $\delta = 9.3\%$). Dollar values are adjusted to 1981-dollar equivalents using the consumer price index.

Table A2.1.1 CBA of education.

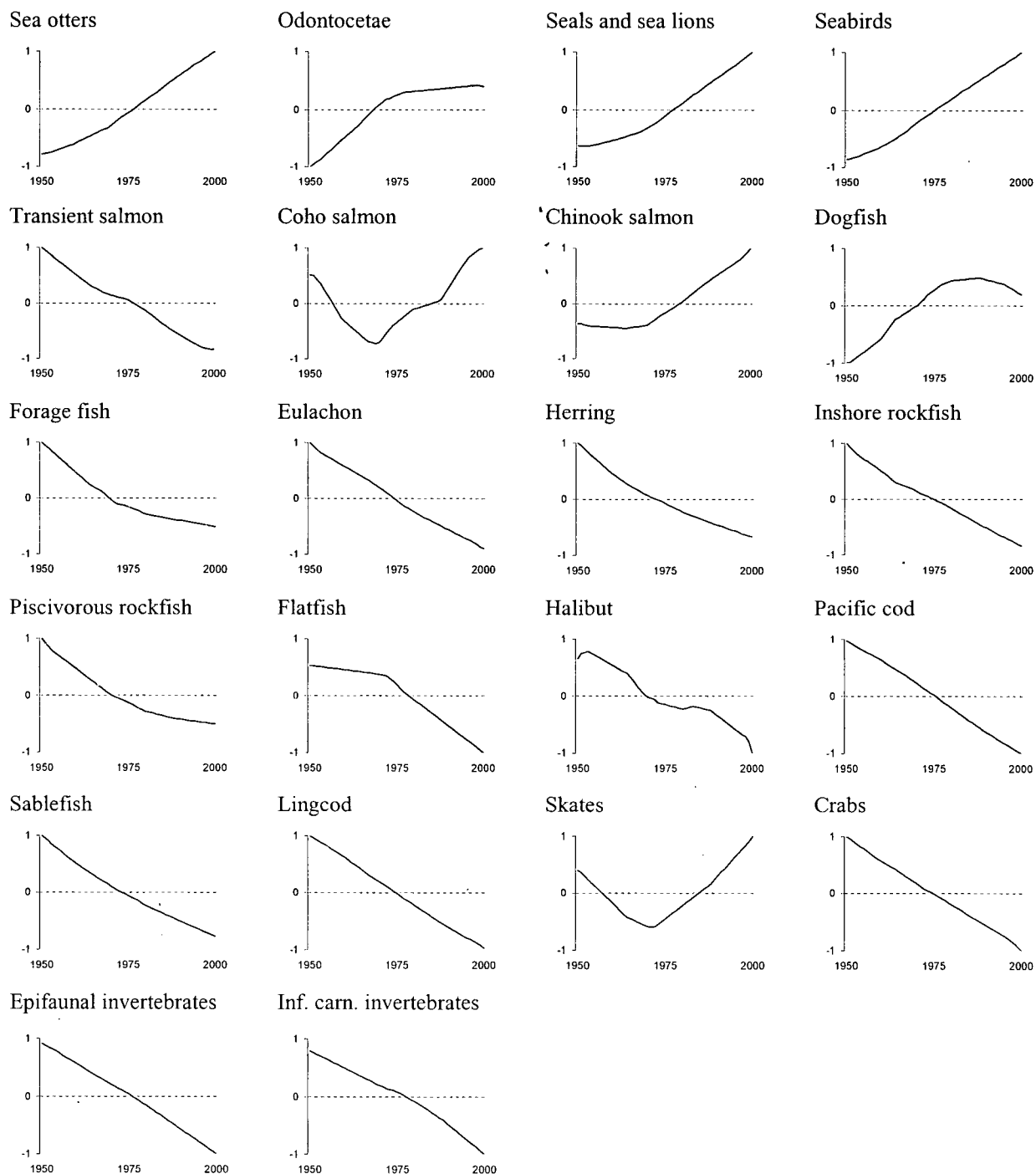
Educational level	Years of education	δ where education is worthwhile (IRR in %) ^a	Discounted constant 1981 dollars ($\cdot 10^3$)		
			Cost	Benefit	NPV ^b
Grade 10	10	15.0	22.2	42.0	19.9
High school diploma	12	10.3	25.1	28.5	3.4
Trade school diploma	13	11.4	27.5	36.3	8.7
College diploma	14	8.6	29.8	26.8	-3.0
Bachelors degree	16	7.5	33.6	25.0	-8.6
Masters degree	18	7.4	36.9	26.5	-10.4
PhD degree	22	6.3	41.8	24.6	-17.3

a. Internal rate of return

b. Net present value based on alternative rate of return from bank interest ($\delta = 9.3\%$)

Appendix 3.1 Local Ecological Knowledge (LEK): Trends of Relative Abundance

Figure A3.1.1 LEK trends of relative abundance



Appendix 4.1 BC Fisheries Timeline

Table A4.1.1 BC fisheries timeline

References listed in Appendix 9.4.1

Period	Event
1950-1954	Otter trawls introduced in BC (c. 1950), previously most vessels were side trawlers [105]. Drum seiners introduced (c. 1950) [126]. Purse seining mechanized using hydraulic systems (e.g., "puretic block") [115, 126]. American fleet enters BC waters in numbers (c. 1950) [127, 5], although US had been operating near S. Vancouver Island previously [82]. International North Pacific Fisheries Commission established ¹⁹ (1952) [119]. Canada/US treaty signed, "Convention for the Preservation of the Halibut Fishery of the Northern Pacific Ocean and Bering Sea" (1953) [32]. Collection of catch statistics improves for Pacific Ocean perch and other rockfish under voluntary trawl logbook program (1954) [82, 57].
1955-1959	Synthetic fibres (nylon, polyester) used in net construction (c. 1955) [15]. Echo-sounders integrated into fleet following World War II. Western Canadian Whaling Ltd. close operations (1959), limited foreign whale harvest continues until late 1960s [126].
1960-1964	Larger trawl vessels in use (c. 1960). Spawning channel construction, flow control projects and hatchery programs initiated to improve salmon production (c. 1960) [126]. Foreign catch now includes sablefish (1961) [82]. Foreign vessels required to carry license (1961) [105]. USSR fleet enters BC waters (c. 1965) [127]. Canadians begin groundfish trawl in earnest [82]. Rockfish fishery begins (c. 1960), but accurate records not kept until 1967 [82].
1965-1969	Japanese fleet enters BC waters (c. 1966) [127]. Rockfish records begin (1967) [82], but with a large degree of aggregation in early years [35]. Final whaling season (1968) [107]. Improved technology to preserve fish at sea (e.g., quick freezing) [12]. Long-range navigation systems (LORAN) in use.
1970-1974	Fishermen begin reporting an increasing variety of rockfish species, misreporting may have occurred to avoid restrictive regulations (c. 1970) [82]. Freezers and 'cold chains' open a market for less valuable species to make processed foods (e.g., Kamaboko, imitation crab meat, faux scallops) (c. 1970) [128]. Large area fishery closure in Queen Charlotte Sound to reduce foreign fishing for Pacific Ocean perch (POP) (1971) [57].

¹⁹ INPFC was responsible for numerous projects from 1952 to 1993 to protect stocks and habitats of anadromous species, to control harvesting and improve productivity..

Table A4.1.1 BC fisheries timeline (cont.)

1975-1979	<p>Polish fleet enters BC waters (c. 1975) [127]. Average vessel size has doubled since 1940 up to 60 tonnes (c. 1975) [105]. Trawl fishery shifts away from POP and towards other rockfish and flatfish species [122]. Limited entry to groundfish trawl established using trip limits (1976) [128]. Canada unilaterally extends its fisheries jurisdiction to 200 nautical miles - Economic Exclusive Zone (EEZ) formed²⁰ (1977) [8, 31]. Other countries follow, and UNCLOS III will eventually formalize EEZ concept in 1982 [125]. Soviet and Japanese rockfish catches cease (1977) [127]. Salmonid Enhancement Program²¹ (SEP) begins (1977) [51]. A change in British Columbia Fishing Regulations of 1888 explicitly prohibits natives from selling their catch commercially (1977) [118]. Aquaculture begins in BC in the late 1970s: first farms were on the sunshine coast and northeast coast of Vancouver Is. raising Pacific species chinook, coho and sockeye (c. 1977) [1]. Annual quotas applied to groundfish fishery (1978) [82]. Department of Fisheries and Oceans (DFO) established (1979) [19]. DFO implements measures to control harvest of groundfish stocks (1979) - license limitations, total allowable catches (TACs), species/area/time closures and trip limits [82].</p>
1980-1984	<p>Global Positioning System (GPS) in use by fishing vessels (1980) [64] – not widespread. Rock-hoppers/tickle chains reduce habitat damage and bycatch in trawl fleet (c. 1980). Most foreign fishing ends in EEZ (c. 1980) [5]. Increasing engine power and the availability of fiberglass hull designs facilitates more “bow pickers” in the salmon gillnet fleet (c. 1980) [9], which use shorter ‘soak’ times, reducing bycatch [3]. BC recognizes Race Rocks as an ecological reserve (1980) [104]. Groundfish trawl advisory committee formed (GTAC) (1980). Trip limits and area quotas imposed on yellowtail rockfish fisheries (1980) [71]; quota applied on rock sole (1980) [68]. Increased misreporting, discards in rockfish fishery in response to new regulations (c. 1980) [122]. Amendment to 1888 BC Fishing Regulations restricts the number and species that may be caught by aboriginals for food, social and ceremonial purposes (1981) [118]. New techniques improve aging of major groundfish species (1981) [114]. US rockfish catch ceases (1982) [127, 57]. Widow rockfish quota implemented (1983) [70].</p>

²⁰ EEZ formed under authority of *Territorial Sea and Fishing Act* of 1964.

²¹ SEP increases salmon production through hatchery programs, spawning channel improvement and lake fertilization; also assists in Community Economic Development Program (CEDP).

Table A4.1.1 BC fisheries timeline (cont.)

1985-1989	SEP integrated into Pacific Region Salmon Resource Management Plan ²² (SRMP) (1985) [126]. Fisheries Act ²³ introduced (1985) [20]. Canada/US Pacific Salmon Treaty ²⁴ (PST) of 1984 is ratified by Canada (1985) [33]. Commercial fishery for widow rockfish begins (1986) [70]. First quota for lingcod introduced (area 3b, W. Vancouver Is.; 1987) [91]. Pacific Regions Fisheries Observer Program ²⁵ initiated by DFO, Archipelago Marine Research Ltd. contracted to provide observer coverage for foreign and domestic vessels operating within EEZ (1987) [10]. General Agreement on Tariffs and Trade (1988). Observer coverage extended offshore to seamount fisheries (1989) [10]. With salmon aquaculture on the rise, and Alaskan salmon fisheries producing unprecedented yields, salmon prices fall throughout late 1980s and 1990s [106]. Commercial size limit for lingcod increased from 58 cm to 65 cm in Strait of Georgia (area 4b; 1989) (original coast-wide limit of 58 cm was established in 1940s) [91].
1990	Sparrow decision (1990) affirms aboriginal right to fish under Sec. 35(1) of 1982 Constitution Act [21]. Individual transferable quota (ITQ) established for sablefish fishery (1990) [93]. Annual quotas for English sole applied (1990) [63]. Vessel trip quotas for petrale sole applied (1990) [66]. Commercial lingcod fishery closed in area Strait of Georgia (area 4b; 1990) - still open for recreational fleet, but 65 cm size limit imposed in 1991 [91]. Dockside monitoring initiated for ITQ sablefish fishery, 100% coverage of landings (1990) [10]. Weedlines used in gillnet fishery to reduce steelhead bycatch (c. 1990). Sorting grids (c. 1990).
1991	Canada-US Free Trade Agreement (FTA) (1991) [34]. Joint Canada/US commitment to reduce halibut bycatch through International Pacific Halibut Commission (IPHC) (1991) [108]. Halibut individual vessel quota system initiated in British Columbia, beginning with a 2 year trial (1991) [108, 10]. Voluntary 140 mm cod-end mesh size suggested by 1991 Pacific groundfish trawl management plan for Pacific cod fishery (78 mm was already legislated) (1991) [65]. Dockside monitoring initiated for halibut IVQ fishery (1991) [10].

²² SRMP includes fleet management, habitat restoration, natural stock rebuilding and salmonid enhancement.

²³ Main provisions of the Fisheries Act deal with habitat protection, pollution prevention, licensing, powers of fishery officers and marine safety; authorizes DFO to implement fishery closures for conservation purposes.

²⁴ Pacific Salmon Commission established by PST; it has power to enact time/area closures, gear restrictions and harvest limitations.

²⁵ Observer program includes several objectives: to monitor trawl offloading events, to collect data on experimental fisheries (e.g. tanner crab traps, squid fishing, seamount fishing), to collect information for research programs (e.g. halibut bycatch), and to monitor for enforcement purposes.

Table A4.1.1 BC fisheries timeline (cont.)

1992	TAC and trip limits imposed on Hecate Strait Pacific cod fishery (1992), previously open access [65]. DFO cooperates with Department of National Defense to conduct air surveillance patrols of North Pacific (primarily looking for high seas drift netting) (1992) [72]. Salmon agreements made by Canada and US under the 1984 Pacific Salmon Treaty expire, competitive harvesting commences under the political limbo (1992) [82]. Aboriginal Fishing Strategy (AFS) established in response to Sparrow decision (1992) [94]. Pilot Sales Program permits limited commercial catch for natives (1992) [118], intended to reduce wide-scale poaching. DFO issues permits for commercial aboriginal salmon fisheries in Port Alberni and Lower Fraser R. (1992) and later, Skeena R. under the Aboriginal Fishing Strategy - controversial fisheries called “race-based” by critics [124, 13]. 1992 Fraser R. sockeye fishery an “environmental disaster”, while opponents to Sparrow decision claim link to AFS [4]; Pearce-Larkin report commissioned in response [120]. Pacific license retirement plan initiated (1992) [118].
1993	North Pacific Fisheries Commission dissolved (1992), replaced by Convention for the Conservation of Anadromous Stocks in the North Pacific Ocean (1993) [119]. Sustainable Fisheries Program launched by DFO to conserve Skeena R. salmonids (1993) – includes steelhead observer program and selective fishing experiments [38]. Widow and canary rockfish receive coast-wide quotas and trip limits (1993), both fisheries previously unrestricted [70, 60]. Quota for lingcod introduced (area 5c/d; 1993) [91]. Aboriginal Communal Fishing Licenses Regulations approved (1993) [106, 2].
1994	User-pay mandatory dock-side monitoring program put in place for majority of groundfish trawl fisheries (1994) [45, 10] – misreporting reduced. United Nations Law of the Sea ²⁶ (UNCLOS) enters into force (ratified by Canada 2003) [28].

²⁶ UNCLOS III sets out general rules to govern oceans: defines territorial sea, contiguous zones and establishes Exclusive Economic Zones (EEZ).

Table A4.1.1 BC fisheries timeline (cont.)

1995	<p>Directed fishery for Petrale sole discontinued coast-wide (1995) [66]. IPHC regulation: halibut must be dressed before offloading (1995) [109]. IPHC regulation: halibut caught as bycatch in Pacific cod and sablefish fisheries can now be kept (1995) [109]. Staged reduction of halibut trawl bycatch mortality (1995) [130]. Mesh size regulation introduced for rock sole fishery to minimize juvenile mortality (1995) [82]. 140 mm mesh size for Pacific cod trawl fishery is legislated (1995), was previously voluntary [65]. Under 1992 Pilot Sales Program, commercial licenses granted to three lower mainland First Nations bands on lower Fraser River. Jack, John and John case²⁷ (1995) [16]. United Nations Food and Agriculture Organization (FAO) draft 'Code of Conduct for Responsible Fisheries', adopted by Canada (1995) [103]. Groundfish trawl fishery suspended mid-season (TAC exceeded), hook and line limited (1995) [39]. Mifflin Plan announced, for voluntary buy-back of commercial fishing licenses (1995) [106]. BC Sport Fishing Regulations now prohibit on-sea canning to improve data collection (1995) [37]. Restrictions introduced on west coast Vancouver Island and Strait of Georgia coho stocks due to conservation concerns (1995) [121]. 1995 – 2000 see budget cuts for DFO.</p>
1996	<p>Van der Peet case²⁸ [23]. New sport fishery limits introduced for many species not previously regulated, including sablefish and yelloweye rockfish; also higher fines for offences under BC Sport Fishing Regulations (1996) [41]. Canada Oceans Act passed²⁹ (1996) [22, 99]. Coastal fisheries protection regulations amended; it is now mandatory for foreign fishing vessels to notify DFO on passage through BC waters (1996) [42]. New aerial surveillance pilot project augments DFO's enforcement capabilities, improves data collection for commercial fisheries effort statistics 1996 [40]. Public consultation begins for upcoming groundfish IVQ allocations (1996) [44]. IPHC increases recreational halibut limit in Strait of Georgia (area 2b), halibut fletching (filleting) prohibited at sea (1996) [110]. Lingcod size limit increased from 58 cm – 65cm near Queen Charlotte Is. (area 5e) and west coast Vancouver Is. (area 3c/d) (1996) [91]. Trawl catch for Pacific cod in Hecate Strait limited to bycatch only, IVQs implemented (1996) [65]. IVQs implemented for dover sole (area quotas previously), and English sole (1996) [82, 62]. New Groundfish Management Plan (1996) includes: mandatory on-board observer program for groundfish trawl (replaces limited observer program in place since 1987), and bycatch caps for trawlers (1996) [82, 43, 46]. Despite additional costs of observer program, IVQ scheme stimulates a large increase in the number of active trawlers; probably increased the occurrence of illegal sales and high grading (1996) [12]. British Columbia Liberal government places 6 year moratorium on new Salmon aquaculture (netpen) operations (1996).</p>

²⁷ Jack, John and John case affirms DFO fiduciary obligation to manage stocks specific fish to a tribal group's traditional area (1995) [123]

²⁸ Van der Peet (1996) conviction was eventually upheld by the Supreme Court of Canada, affirming that aboriginal commercial rights to the resource exist only if sale is a continuation of traditional tribal practices.

²⁹ Major provisions of the Oceans Act (1996) include recognition of Canada's ocean jurisdiction, guidelines for sustainable development, integrated management and precautionary approach, and consolidation of federal fisheries/environmental responsibilities [77].

Table A4.1.1 BC fisheries timeline (cont.)

1997	<p>Major groundfish trawl regulation change: comprehensive IVQ scheme set for 25 groundfish species (1997), IVQ holders allocated 80% of TAC [50] - formalized in 1998 [54]. Pacific halibut management plan reallocates halibut IVQ; bycatch objectives set under 1991 IPHC agreement have been well exceeded (1997) [49]. Kwakiutl Aboriginal Fisheries Guardians join DFO on enforcement patrols on Vancouver Is. and central BC (1997) [48]. Following major regulatory changes in groundfish trawl industry, Groundfish Development Authority (GDA) formed to protect non-quota shareholders (1997) [77]. Limited entry established for groundfish hook and line (1997) [46]. Delgamuukw decision³⁰ (1997) [24]. Quota for lingcod introduced (area 5e; 1997) [91]. Steady decline in world salmon prices due to continued development of aquaculture, particularly in Norway and Chile [96]; in 1997, prices are approximately 1/3 of ten years ago [7]. Retention/possession of coho, chinook and steelhead for all seine fisheries disallowed (1997) [46, 47].</p>
1998	<p>Canadian Code of Conduct for Responsible Fisheries developed through grassroots industry initiative (1998) [30, 95]. Under new Pacific Fisheries Adjustment and Restructuring Program (PFAR) (1998) [79], the Selective Fishing Program (SFP) is initiated by DFO; 5% of salmon TAC allocated to improve fisheries selectivity [69, 76, 80]; experimental fisheries authorized for data collection and aboriginal fisheries: includes fish wheels, weirs, beach seine, live fish traps and Norwegian fish herders [13, 14, 79, 73, 58] NB: widespread use has not materialized as of 2004 [14]. Under Selective Fishing Program, gear modifications³¹ are mandated between 1998 and 2002 through licensing conditions to reduce coho bycatch mortality [14, 81, 56, 75, 78], however time/area closures still principle methods [14]. Selective fishing initiative successfully reduces coho bycatch mortality from 60% to 5% [14, 69]. New Coho Response Team evaluates coho conservation in BC, seeks new methods to reduce bycatch among other projects (1998) [92]. Fisheries Renewal BC established, coordinates resources for habitat enhancement made available under Pacific Salmon Endowment Fund and Pacific Salmon Treaty (1998) – Fisheries Renewal operates until 2002 [89, 83]. IPHC issues new halibut logbooks to improve in-season data collection (1998) [111]. DFO and BC Wildlife Federation (BCWF) create first fishery officer-dog team trained for fisheries enforcement (1998) [55]. Commercial troll fleet in lower Strait of Georgia (area H) gets 10% observer coverage (1998) [10]. DFO releases PFAR report, “A new direction for Pacific Salmon Fisheries” – reiterates commitment to develop selective fishing practices, among other priorities (1998) [52].</p>

³⁰ Delgamuukw decision recognizes aboriginal title to land and resources, affirms aboriginal right to fish commercially [29].

³¹ Revival tanks are made mandatory throughout BC salmon fleet; gillnetters employ shorter-than-regulation nets and shorter soak times; seiners use brailing with dip nets, knot-less mesh to reduce abrasion, and plastic escape holes to release undersized fish; trollers use large and barbless hooks [14].

Table A4.1.1 BC fisheries timeline (cont.)

1999	<p>DFO begins pilot project using satellite technology and psion units to collect real-time catch information, as well as biological and oceanographic data (1999) [84]. BC Fisheries Survival Coalition organizes a large-scale protest fishery in 1998 on the Fraser River following smaller actions in 1996 and 1997, to protest native-only fisheries set under DFO's 1992 Pilot Sales Program. Thomas case affirms that "race-based" fisheries are discriminatory and illegal, stays charges against 24 non-aboriginal protesters from 1996/1997 Fraser River protest fisheries (1998) [6]; an appeal is made to BC supreme court, but rejected in 1999. The 1999 groundfish trawl fleet was composed of 142 licensed vessels, of which approximately 88 recorded landings (1999) [82]. DFO observer program extended to include halibut fisheries (hook and line now, in addition to trawl) (1999) [82]. Observer coverage extended to include rockfish fisheries and Rockfish Protected Areas (RPAs) implemented under recommendation of Groundfish Hook and Line Advisory Committee [87]; rockfish discarding now prohibited (1999) [82, 61]. Size limit for recreational lingcod increased to 65 cm off West Vancouver Island (areas 3c/d) (1999) [91]. BC recreational fishers required to supply catch records (1999) [67]. Formal joint commitment by Canada and US to restore salmon habitat, improve management under 1985 Pacific Salmon Treaty, two endowment funds established totaling \$200 million CDN (1999) [59]. Additional selective fishing experiments in Juan de Fuca Strait (1999) [36]. Supreme Court of Canada Marshall decision affirms the rights of two Maritime aboriginal groups to fish, hunt and gather in pursuit of "moderate livelihood" (1999) [26]. Canadian Environmental Protection Act³² (CEPA) passed (1999) [25].</p>
2000	<p>Nis'ga treaty becomes law, first modern-day land claim - includes exclusive aboriginal fishing rights (2000) [11]. Groundfish trawlers accept voluntary area closures to protect Hexactinellida sponge reefs (2000) [116, 113]. Environment Canada announces National Programme of Action³³ (NPA) (2000) in response to Global Plan of Action initiated by United Nations Environment Programme (UNEP) in 1995. Additional DFO funding strengthens fishery enforcement (2000) [74].</p>

³² CEPA includes marine protection provisions to complement existing measures, gives Minister of the Environment additional powers to enact environmental guidelines. Particularly addresses land-based marine pollution concerns.

³³ NPA highlights conservation, sustainable use and economic diversification to reduce land-based marine pollution [101].

Table A4.1.1 BC fisheries timeline (cont.)

2001	United Nations Fish Stocks Agreement ³⁴ (UNFA) ratified by Canada in 1999, now goes into effect (2001) [85]. DFO extends observer coverage to include all groundfish hook and line fisheries (in addition to trawl) (2001) [82]. IPHC allows limited filleting of halibut while at sea (2001) [112]. To aid in data collection, log books now kept by halibut fishers throughout the season, rather than surrendering at each off-loading (2001) [112]. Race Rocks are established as Canada's first Marine Protected Area under Oceans Act (2001) [117].
2002	After a six-year moratorium on new aquaculture development, permits for new Atlantic salmon net-pens are issued (2002) [100]. DFO announces Rockfish Conservation Strategy ³⁵ (RCS) (2002) [87]. Rockfish Conservation Areas (RCAs) planned under Rockfish Conservation Strategy (i.e., an extension of Rockfish Protected Areas, established 1999) (2003) [87]. A 2001 Pacific Scientific Advice Review Committee (PSARC) document reports unfavorable lingcod recovery – commercial closure maintained for Strait of Georgia (in place since 1990), recreational closure to commence (2002) [90]. Fraser River coho listed as “endangered” by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) (2002) [88, 98]. Coast-wide commercial fishery closure for sablefish (2002) [86]. Areas surrounding Hexactinellida sponge reefs closed (annually) to commercial trawling by annual regulations (2002) – area currently under review for permanent Marine Protected Area designation [113].
2003	Species at Risk Act (SARA) comes into effect (2003) [27]. Green/shortnose sturgeon and Bocaccio rockfish currently listed as “threatened” by SARA [102]. Sauderson decision grants absolute discharge to 40 non-aboriginal fishers involved in Johnstone Strait protest fishery of 2002 [17]. Kitchen decision grants stay of proceedings to 140 non-aboriginal fishermen of charges following large 1998 protest fishery on Fraser River [18]. Pink Salmon Action Plan ³⁶ (PSAP) began in Broughton Archipelago [97].

³⁴ UNFA fills in gaps in the United Nations Law of the Sea (UNCLOS) with regards to migratory and straddling stocks.

³⁵ RCS to focus on four areas of conservation: harvest reduction, establishment of rockfish protected areas, catch monitoring and stock assessment improvement.

³⁶ PSAP to address poor returns at the north end of Vancouver Is., mortality factors to be investigated – particularly sea lice infections as may be caused by proximate salmon net pen operations.

Appendix 4.2 IUU Influences Table

Table A4.2.1 IUU influences table

Period	Event summary ^{a,b}				Refs ^c	Influence ^d			Affected fleet ^e						Rationale ^f	Duration		Termination event	
	Policy	Technology	Political	Supply/market		Discard	Illegal	Unreport	Salmon			Groundfish				Start	End ^g		
									Gillnet	Troll	Seine	Rec	Trawl	H&L	Rec				
1950 - 1954		otter trawls			105	▲							✓		Increased catching power	1950	2003		
		drum seiners			126	▲					✓				Increased catching power	1950	2003		
		hydraulics			115, 126	▲			✓	✓	✓		✓	✓	Increased catching power	1950	2003		
				US fleet enters	127, 5	▲			✓	✓	✓		✓	✓	Increased catching power, heavy American presence	1950	1982		EEZ Formed
			INPFC established		119	△			✓	✓	✓		✓	✓	Functions include harvest controls	1952	1992		CCASNPO established
			Can/US halibut treaty		32	▼	▼						✓	✓	Co-management commitment, power of seizure for illegal activity	1953	2003		
		trawl logbook programme			82, 57			▽					✓		Logbooks only voluntary	1954	1960		Mandatory recording
1955-1959		synthetic fibres			15	▲			✓	✓	✓	✓	✓	✓	✓	Increased catching power	1955	2003	
		echo sounders				▲			✓	✓	✓		✓	✓	Increased catching power	1955	2003		
1960-1964		unrecorded rockfish landings begin			82			▲					✓				1960	1967	EEZ Formed
		larger trawl vessels				▲									Increased catching power	1960	2003		
		foreign sablefish fishery begins			82	▲							✓	✓	Increased catching power, foreign fleet less accountable than domestic	1961	1977		
		foreign vessles licenced			105			▼	✓	✓	✓		✓	✓	Improved data collection	1961	2003		
				USSR fleet enters	127	▲							✓	✓	Increased catching power, heavy Soviet presence; rockfish targeted	1965	1977	EEZ Formed	
		domestic trawling begins			82	△							✓	✓	Increased catching power	1967	2003		
		foreign / domestic rockfish fishery begins			82	▲		▲					✓	✓	Increased catching power; no rockfish records are kept initially	1967	2003		
1965-1969				Japan fleet enters	127	△							✓	✓	Increased catching power, light Japanese presence; rockfish targeted	1966	1977		
		rockfish records begin			82, 35			▽					✓	✓	Records are highly aggregated in early years	1968	2003		
		freezing/preservation			12	▼			✓	✓	✓		✓	✓	Fishing vessels have access to ice	1968	2003		
		ground-based navigation systems				▼			✓	✓	✓		✓	✓	Fisheries become more selective	1969	2003		

a Events listed in chronological order. Refer to Table 1 for description of event.

b Abbreviations used: Strait of Georgia (SOG); Hecate Strait (HS); West Coast Vancouver Is. (WCVI); Queen Charlotte Is. (QCI); Queen Charlotte Sound (QCS).

c References listed in Table 2.

d Closed arrow = major influence; open arrow = minor influence (see rationale).

e Salmon troll includes all commercial hook and line methods.

f See text for conventions used.

g n/a = termination event for previous influence

Table A4.2.1 IUU influences table (cont.)

Period	Event summary				Refs	Influence			Affected fleet						Rationale	Duration		Termination event	
	Policy	Technology	Political	Supply/market		Discard	Illegal	Unreport.	Salmon			Groundfish							
									Gillnet	Troll	Seine	Rec	Trawl	H&L		Rec			
1970-1974	new rockfish regulations				82	▲		▲					✓	✓	Rockfish misreporting increases to contravene restrictions	1970	1980	DFO regulations	
	"cold chains" introduced				128	▼			✓	✓	✓		✓	✓	Cold chains open a market for less valuable species previously discarded	1970	2003		
	POP closure (QCS)				57	▽	△						✓	✓	✓	Affects only a portion of study area (QCS); opportunity for poaching	1971	1974	
1975-1979	Poland fleet enters				127	△							✓	✓	Increased catching power; rockfish targeted; light Polish presence	1975	1977	EEZ Formed	
	larger fishing vessels				105	▲			✓	✓	✓		✓	✓	Increased catching power; longer trips	1975	2003		
	additional rockfish/flatfish spp. targeted				122	▼		△					✓	✓	Bycatch retained and sold; data collection inadequate for rockfish	1975	2003		
	trip limits GF trawl				128	▲		△					✓		Fishers can no longer land everything they catch; incentive to high grade, misreporting as other species	1976	2003		
	EEZ formed				8, 31	▼	△	▼	✓	✓	✓		✓	✓	Reduced catching power (foreign restricted); opportunity for poaching; better monitoring of domestic fleet than foreign fleet	n/a	n/a		
	USSR / Japan rockfish fishery ends				127	▼	△	▼					✓	✓	Reduced catching power (foreign restricted); opportunity for poaching; better monitoring of domestic fleet than foreign fleet	n/a	n/a		
	SEP begins				51	△			✓	✓	✓				CEDP assists fishers (subsidy); increased catching power	1977	2003		
	legislation ends native commercial fishery				118		▲	▲	✓	✓	✓		✓	✓	Opportunity for poaching; catch not reported	1977	1992	Pilot sales program	
	annual quotas set for GF				82	▲		△					✓	✓	Fishers can no longer land everything they catch; incentive to high-grade, misreporting as other species	1978	2003		
	DFO established				19		▼	▽	✓	✓	✓		✓	✓	Enforcement improved; data collection improved	1979	2003		
	GF TAC, trip limits, closures, license limitations				82	▲		△					✓	✓	Fishers can no longer land everything they catch; incentive to high-grade, misreporting as other species	1979	2003		
1980-1984	GPS				42	▼			✓	✓	✓		✓	✓	Fisheries become more selective	1980	2003		
	rockhoppers					▲							✓		Increased catching power	1980	2003		
	tickle chains, cod-end sensors					▼							✓		Bycatch reduction devices	1980	2003		
	foreign fishing ends in EEZ				5	▼	△	▼	✓	✓	✓		✓	✓	Better monitoring of domestic fleet than of foreign fleet; opportunity for poaching	n/a	n/a		
	more bowpickers in salmon fleet				9, 3	▽			✓						Bowpickers use shorter soak times, reduce bycatch	1980	2003		
	Race Rocks ecological reserve				104	▽	△						✓	✓	Minor: affects only a portion of study area; increased opportunity for poaching	1980	2003		
	GTAC formed					△							✓		Helps establish GF regulations; management-industry liaison	1980	2003		

Table A4.2.1 IUU influences table (cont.)

Period	Event summary				Refs	Influence			Affected fleet						Rationale	Duration		Termination event		
	Policy	Technology	Political	Supply/market		Discard	Illegal	Unreport.	Salmon			Groundfish				Start	End			
									Gillnet	Troll	Seine	Rec	Trawl	H&L					Rec	
1980-1984 (continued)	trip limits, area quotas for yellowtail rockfish				71	▲		△						✓	✓	Fishers can no longer land everything they catch; incentive to high-grade, misreporting as other species	1980	2003		
	rock sole quota				68	▲		△						✓	✓	Fishers can no longer land everything they catch; incentive to high-grade, misreporting as other species	1980	2003		
	new rockfish regulations				122	▲		▲						✓	✓	Tighter regulations on rockfish retention; increased misreporting to conceal catch [122]	1980	2003		
			restrictions on aboriginal food fish		118		▲		✓	✓	✓			✓	✓	Opportunity for poaching	1981	1990		Sparrow case
			US rockfish fishery ends		127, 57	▼		▼						✓	✓	Less foreign fishing, fewer unreported catches	n/a	n/a		
	widow rockfish quota				70	▲		△							✓	✓	Fishers can no longer land everything they catch; incentive to high-grade, misreporting as other species	1983		2003
1985-1989			SRMP drafted		126	△			✓	✓	✓					Assists salmon fishers, catching power increased	1985	2003		
			Fisheries Act passed		20	▼	▼	▼	✓	✓	✓			✓	✓	Provisions allow fisheries closures, grants powers to DFO officers	1985	2003		
			Pacific Salmon Treaty ratified		33	▲			✓	✓	✓					Tighter regulations on salmon retention	1985	1992		Expires
	commercial widow rockfish fishery begins				70	▲								✓	✓	Increased catching power	1986	2003		
	lingcod quota				91	▲								✓	✓	Fishers can no longer land everything they catch; incentive to high-grade, misreporting as other species	1987	2003		
			GF at-sea observer program begins		10	▽	▽	▽						✓	✓	Partial coverage only	1987	1996		Trial program ends
			GATT treaty			▼			✓	✓	✓			✓	✓	Market is less restrictive, new markets open.	1988	2003		
	GF observer coverage extended offshore				10	▽	▽	▽						✓	✓	Partial coverage only	1989	2003		
			salmon prices fall throughout 80s/90s		106	▲	▲		✓	✓	✓					Incentive to sell privately, incentive to highgrade	1985	2003		
	lingcod size limit increased (SOG)				91	△										More discarding of undersized individuals. Only a portion of study area.	1989	1990		Commercial retention banned (SOG)
1990-1994	Sparrow decision				21		▼		✓	✓	✓			✓	✓	Aboriginal subsistence fishery is now legal	1990	2003		
	sablefish ITQ				93	▲								✓	✓	Fishers can no longer land everything they catch; incentive to high-grade	1990	2001		Closure
	English sole quota				63	▲		△						✓	✓	Fishers can no longer land everything they catch; incentive to high-grade, misreporting as other species	1990	2003		
	petrale sole trip limits				66	▲		△						✓	✓	Fishers can no longer land everything they catch; incentive to high-grade, misreporting as other species	1990	2003		
	SOG closed to commercial lingcod				91	▽	△							✓	✓	Affects only a portion of study area (SOG); opportunity for poaching	1991	2003		
	sport size limit for lingcod introduced				91	▲									✓	More discarding of undersized individuals	1990	2001		Closure

Table A4.2.1 IUU influences table (cont.)

Period	Event summary				Refs	Influence			Affected fleet						Rationale	Duration		Termination event
	Policy	Technology	Political	Supply/market		Discard	Illegal	Unreport.	Salmon			Groundfish				Start	End	
									Gillnet	Troll	Seine	Rec	Trawl	H&L				
1990-1994 (continued)	dockside monitoring for sablefish (100%)				10		▼	▼					✓	✓	Data collection improved, improved enforcement	1990	2003	
		gillnet weedlines				▼			✓						Bycatch reduction device	1990	2003	
		sorting grids				▼							✓		Bycatch reduction device	1990	2003	
				Free Trade Agreement	34	▼			✓	✓	✓		✓	✓	Market is less restrictive, new markets open.	1991	2003	
			IPHC bycatch reduction agreement		108	▼							✓	✓	Agreement's stated purpose is to reduce halibut bycatch	1991	2003	
	Halibut IVQ				108, 10	▲							✓	✓	Fishers can no longer land everything they catch; incentive to high-grade	1991	1993	
	voluntary mesh size increase for Pacific cod				65	▽							✓	✓	Fewer discards of undersized fish; voluntary (limited effect)	1991	2003	
	dockside monitoring for halibut				10		▼	▼					✓	✓	Data collection improved; improved enforcement	1993	2003	
	TAC/trip limits Pacific cod				65	▲							✓	✓	Fishers can no longer land everything they catch; incentive to high-grade	1992	2003	
		high seas aerial surveillance			72		▽			✓	✓				Minor improvement to enforcement (adjacent to study area)	1992	2003	
			Pacific Salmon Treaty expires		82	▲	▲	▲	✓	✓	✓				Widely publicized "fish wars" begin - grab for fish	n/a	n/a	
	INPFC dissolved				129	▽									Founded in 1953.	n/a	n/a	
	CCASNPO founded				129	△									Replaces INPFC.	1992	2003	
	Aboriginal Fishing Strategy				94	▽	▽		✓	✓	✓		✓	✓	First Nations contribute to management. FN harvest allocations. Gov't must consult FN before restrictions.	1992	2003	
	Pilot sales program				118		▼		✓	✓	✓				Intended to reduce poaching - salmon openings only	1992	2002	
	Pacific license retirement plan				118		▽		✓	✓	✓				reallocation to FN under AFS; subsistence needs satisfied	1992	2003	
			Sustainable Fisheries Program		38	▽	▽	▽	✓	✓	✓				SFP includes steelhead observer program, selective fishing experiments. Only affects part of study area (Skeena)	1993	2003	
	widow rockfish quota				70	▲		△					✓	✓	Fishers can no longer land everything they catch; incentive to high-grade, misreporting as other species	1993	2003	
	canary rockfish quota				60	▲		△					✓	✓	Fishers can no longer land everything they catch; incentive to high-grade, misreporting as other species	1993	2003	
lingcod quota				91	▲		△					✓	✓	Fishers can no longer land everything they catch; incentive to high-grade, misreporting as other species	1993	2003		

Table A4.2.1 IUU influences table (cont.)

Period	Event summary				Refs	Influence			Affected fleet						Rationale	Duration		Termination event			
	Policy	Technology	Political	Supply/market		Discard	Illegal	Unreport	Salmon			Groundfish				Start	End				
									Gillnet	Trawl	Seine	Rec	Trawl	H&L	Rec						
1990-1994 (continued)	dockside monitoring for all GF		Aboriginal communal licenses		106, 2		▼	▼	✓	✓	✓				Allows sale of product. Aids data collection.	1993	2003				
					45, 10			▼	▼				✓	✓	Mandatory user-pay system, improves data collection.	1993	2003				
			UNCLOS ratified		28	▼	△	▼	✓	✓	✓	✓	✓	✓	✓	Formalizes EEZ by international law.	1994		2003		
1995-1999	Petrale sole fishery ends rock sole mesh size regulation Pacific cod mesh size regulation now mandatory GF trawl suspended temporarily Salmon sport fishery bans on-sea canning Coho restrictions (SOG, WCVI) DFO funding cuts begin Van der Peet case new sport regulations coastal aerial surveillance				66	▼							✓	✓	Directed fishery ends, petrale bycatch in other sectors is less likely to be discarded.	1995	1995	Closure			
			several IPHC regulations reduce bycatch		109	▼								✓	✓	Certain bycatch must now be landed	1995		2003		
					82	▼								✓		Reduced capture of undersized individuals.	1995		2003		
					65	▼								✓		Trawl only	1995		2003		
			Jack, John and John case		16		▼			✓	✓	✓		✓	✓	DFO under obligation to manage traditional tribal areas. Improved enforcement, monitoring.	1995		2003		
			UN FAO Code of Conduct adopted		103	▼	▼	▼	✓	✓	✓	✓	✓	✓	✓	Contains numerous guidelines for responsible fishing	1995		2003		
					39	▽	▽	▽						✓		Remainder of season closed to GF trawl.	1995		1996		
			vessel buy-back (Mifflin plan)		106		▽			✓	✓	✓		✓	✓	Reallocation to FN under AFS; subsistence needs satisfied	1995		2003		
					37			▼					✓			Regulation meant to improve data collection	1995		2003		
					121	▽	△			✓	✓	✓				Selective fishing measures to protect coho (localized); commercial catch is now illegal	1995		2003		
			DFO funding cuts begin				△			✓	✓	✓	✓	✓	✓	Gradual decline in funding, beginning in mid 1990s. Until 2000 when enforcement is strengthened.	1995		2000	Additional enforcement funding	
			Van der Peet case		23		△	△	✓	✓	✓					Sale is illegal unless a continuation of traditional tribal practices. Illegal catch goes unreported. Most often refers to salmon.	1996		2003		
					41	▲	▼	▲							✓	Sablefish, yelloweye sport now regulated, fishers can no longer land everything they catch. Higher fines reduce illegal.	1996		2003		
			Oceans Act		20, 99	▼	▼	▼	✓	✓	✓	✓	✓	✓	✓	Improved federal/provincial management, improved monitoring and data collection.	1996		2003		
					42		▼			▼	✓	✓	✓		✓	✓	Greater accountability to foreign vessels within EEZ. Less opportunity for illegal practices.		1996	2003	
					40		▼	▼	✓	✓	✓	✓	✓	✓	✓	✓	Reduces illegal activity, improves data collection.		1996	2003	

Table A4.2.1 IUU influences table (cont.)

Period	Event summary				Refs	Influence			Affected fleet						Rationale	Duration		Termination event	
	Policy	Technology	Political	Supply/market		Discard	Illegal	Unreport.	Salmon			Groundfish				Start	End		
									Gillnet	Troll	Seine	Rec	Trawl	H&L	Rec				
1995-1999 (continued)	halibut size limit (SOG)				110	△							✓	✓	✓	Undersized fish now discarded. Affects only portion of study area.	1996	2003	IPHC regulation
	halibut fletching prohibited				110			▼					✓	✓		To improve data collection	1996	2001	
	lingcod size limit increased (QC, WCVI)				91	△							✓	✓	✓	Undersized fish now discarded. Affects only portion of study area.	1996	2003	
	Pacific cod directed trawl fishery ends (HS)				65	▽							✓	✓		Pacific cod bycatch in other sectors is less likely to be discarded. Affects only portion of study area.	1996	2003	
	Pacific cod IVQ				65	▲		△					✓	✓		Fishers can no longer land everything they catch; incentive to high grade, misreporting as other species	1996	2003	
	dover sole IVQ				62	▲		△					✓	✓		Fishers can no longer land everything they catch; incentive to high grade, misreporting as other species	1996	2003	
	English sole IVQ				82	▲		△					✓	✓		Fishers can no longer land everything they catch; incentive to high grade, misreporting as other species	1996	2003	
	on-board 100% GF observer mandatory				82, 43	▼	▼	▼					✓			Groundfish management plan extends trawl observer coverage to 100%.	n/a	n/a	
	trawler GF bycatch caps				82, 43	▼							✓			Groundfish management plan reduces bycatch.	1996	2003	
	GF IVQ increases illegal sales/high grading				12	▲	▲	△					✓			Increases illegal sales and high grading [12]	1996	2003	
				Aquaculture moratorium		△				✓	✓	✓				May increase local salmon prices. More incentive to high grade?	1996	2002	Moratorium lifted
	Comprehensive GF IVQ				54	▲		△					✓	✓		Fishers can no longer land everything they catch; incentive to high grade, misreporting as other species	1997	2003	
			halibut bycatch reduced by IPHC		49	▼							✓	✓		IPHC announces success in controlling bycatch.	1997	2003	
			First Nation aid enforcement		48		▼			✓	✓	✓	✓	✓	✓	Improved enforcement.	1997	2003	
	GF hook and line limited entry				46	▲		△						✓		Fishers can no longer land everything they catch; incentive to high grade, misreporting as other species	1997	2003	
			Delgamuukw decision		24		▼			✓	✓	✓		✓	✓	Affirms aboriginal right to fish commercially.	1997	2003	
	lingcod quota (QCS)				91	△		△					✓	✓		Fishers can no longer land everything they catch; incentive to high grade, misreporting as other species. Affects portion of study area	1997	2003	
				decline in salmon prices	96, 7	▽				✓	✓	✓				Less incentive to high grade	1997	2003	
	seine retention prohibited: coho, chinook, steelhead				46, 47	▲						✓				Must release these species.	1997	2003	

Table A4.2.1 IUU influences table (cont.)

Period	Event summary				Refs	Influence			Affected fleet						Rationale	Duration		Termination event	
	Policy	Technology	Political	Supply/market		Discard	Illegal	Unreport.	Salmon			Groundfish				Start	End		
									Gillnet	Troll	Seine	Rec	Trawl	H&L					Rec
1995-1999 (continued)			Canadian Code of Conduct		30, 95	▼	▼	▼	✓	✓	✓	✓	✓	✓	✓	Contains numerous guidelines for responsible fishing	1998	2003	
			Selective Fishing Program		69, 76, 80	▼			✓	✓	✓					SFP under Pacific Fisheries Adjustment and Restructuring. Designed to reduce bycatch. Salmon only.	1998	2003	
		experimental fisheries			13, 14, 19, 73, 58	▽		▲	✓	✓	✓					Selective methods under SFP not widespread. Mainly used to aid data collection.	1998	2003	
		gear modification to reduce coho mortality			14, 81, 56, 75, 78	▼			✓	✓	✓					Less coho bycatch in salmon fisheries, greater release survival.	1998	2003	
			Coho Response Team		92	▼			✓	✓	✓					Active monitoring and response to incidental coho mortality.	1998	2003	
			IPHC logbook regulation		111			▼					✓	✓		To improve in-season data collection	1998	2001	IPHC regulation
		dog team for enforcement			55		▽		✓	✓	✓		✓	✓		Dog program not widely used.	1998	2003	
	10% troll observer coverage (SOG)				10	▽	▽	▽		✓						Limited coverage, limited portion of study area.	1998	2003	
		Psion/satellite pilot project			84			▽	✓	✓	✓					Real time catch information recorded. Pilot project only.	1999	2001	Trial program ends
			Protest fishery (Fraser R.)		6		△		✓							Largest of four protest fisheries, over 100 boats (smaller protests 96, 97, 02 not included). Limited duration/area.	1999	1999	Single opening
	halibut hook and line observers				82	▼	▼	▼						✓		GF observer program extended to include hook and line.	1999	2000	Becomes mandatory
	rockfish observers				87	▼	▼	▼					✓	✓		GF observer program extended to include rockfish.	1999	2000	Becomes mandatory
			Rockfish Protected Areas		87		△						✓	✓	✓	Opportunity for poaching. Permanent area closure.	1999	2003	
	rockfish discards prohibited				87	▼							✓	✓		Discarding is illegal	1999	2003	
	sport lingcod size limit increased (WCVI)				91	△								✓		More discarding of undersized individuals. Only a portion of study area.	1999	2003	
	sport salmon catch records begin				57			▼				✓				Improved data collection.	1999	2003	
			Can/US commitment to improve salmon management		36		▼		✓	✓	✓					Improved cross-border enforcement	1999	2003	
			Marshall decision		26		▼		✓	✓	✓					Permits limited commercial aboriginal fishery.	1999	2003	

Table A4.2.1 IUU influences table (cont.)

Period	Event summary				Refs	Influence			Affected fleet						Rationale	Duration		Termination event
	Policy	Technology	Political	Supply/market		Discard	Illegal	Unreport.	Salmon			Groundfish						
									Gillnet	Troll	Seine	Rec	Trawl	H&L		Rec		
2000-2004			Nis'ga treaty		11		▼		✓	✓	✓				Establishes exclusive aboriginal fishing rights.	2000	2003	Becomes mandatory
	GF trawl voluntary area closure				116, 113	▽							✓		To protect sponge reefs (limited effect on discards, small area), only voluntary (illegal catch not affected).	2000	2002	
	DFO enforcement gets more funding				74		▼		✓	✓	✓	✓	✓	✓	Improved enforcement.	n/a	n/a	
			UNFA in effect		85			▼	✓	✓	✓	✓	✓	✓	Migrating stocks (salmon) management improves. Improved data collection.	2001	2003	
	GF hook and line observer mandatory				82	▼	▼	▼						✓	Mandatory observer coverage extended to include hook and line.	n/a	n/a	
			IPHC allows at-sea limited filleting of halibut		112			△					✓	✓	Regulation in response to industry petition, data collection compromised.	2001	2003	
			IPHC logbook regulations changed		112			▼					✓	✓	Data collection improved	2001	2003	
			Race Rocks MPA established		117			△	✓	✓	✓	✓	✓	✓	Opportunity for poaching. Permanent area closure.	2002	2003	
				aquaculture moratorium lifted	100	▽			✓	✓	✓				Continued decline in salmon prices as farmed product finds market. Less incentive to high grade?	2002	2003	
			Rockfish Conservation Strategy		87	▲		▼							Improved data collection, harvests reductions increase incentive to high grade.	2002	2003	
	recreational lingcod closure (SOG)				90	▽	△							✓	Only portion of study area, opportunity for poaching.	2002	2003	
	commercial sablefish fishery closed coast-wide				86	▲							✓	✓	Landings decrease, all catch is discarded.	2002	2003	
	GF trawl closure around sponge reefs				113	▽	△						✓		To protect sponge reefs (limited effect on discards, small area), mandatory (opportunity for poaching).	2002	2003	

Appendix 4.3 BC Reported Landings

Table A4.3.1 BC reported landings

Year	Salmon ^{1,2}						Groundfish ^{3,4,5}			Year	Salmon				Groundfish ^{3,4,5}		
	Chinook	Sockeye	Coho	Pink	Chum	Total Salmon	Halibut	Other GF	Total GF		Gillnet	Seine	Troll	Sport ^{1a}	Trawl	Hook & Line	Sport ^{1c}
1951	5851	13517	15987	27307	28799	91522	9052	19873	28926	1951	37043	41237	13236	4576	17545	2328	-
1952	6536	14003	10065	23261	14456	68552	10655	22447	33101	1952	29570	26381	12601	3428	19817	2630	-
1953	7108	16029	10505	28005	24687	86551	11286	17785	29072	1953	34765	39726	11766	4328	15702	2083	-
1954	6119	21328	9385	11685	33748	82518	11431	18847	30277	1954	34953	38335	9226	4126	16639	2208	-
1955	5684	7552	10682	28713	8246	60986	8926	16242	25168	1955	23687	26882	10424	3049	14339	1903	-
1956	6214	9751	11408	13141	12442	53062	10576	18863	29439	1956	23800	18838	10424	2653	16654	2210	-
1957	5738	7135	10333	25987	12355	61621	10225	20072	30297	1957	26212	23463	11939	3081	17721	2351	-
1958	6446	33621	11204	15382	17287	84052	13087	21584	34671	1958	34869	37148	12039	4203	19055	2528	-
1959	6128	8192	8877	15894	10483	49633	13969	20817	34786	1959	20825	17488	11322	2482	18379	2439	-
1960	4681	7022	6455	7707	9217	35195	15362	23760	39122	1960	16910	10918	7366	3519	20977	2783	-
1961	4125	12075	11217	22707	6627	56845	13373	21931	35304	1961	25415	20289	11140	5684	19362	2569	-
1962	4112	9120	12070	42453	8189	76053	15683	22339	38022	1962	30500	34184	11369	7605	19722	2617	-
1963	4611	5386	11558	27478	6997	56095	16907	21475	38382	1963	20754	22195	13147	5610	18959	2516	-
1964	6064	10411	14359	16677	10856	58472	15101	28644	43746	1964	25952	18283	14237	5847	25289	3356	-
1965	5747	7358	18638	10402	3019	43225	14957	32769	47726	1965	16975	10814	15436	4323	28930	3839	-
1966	6951	11660	17546	33325	6968	76589	14515	40364	54879	1966	29451	29007	18131	7659	35636	4728	-
1967	6965	16816	10214	23425	63043	63043	11894	11111	23005	1967	23588	21887	17568	6304	9809	1302	-
1968	6911	18774	15142	25220	16561	82709	13331	16582	29913	1968	33554	31058	18097	8271	14639	1942	-
1969	6452	10949	7991	6272	6077	37806	15332	17079	32411	1969	17800	6969	13037	3781	15078	2001	-
1970	6565	11424	13649	24022	16770	72489	13381	14421	27802	1970	30856	25498	16135	7249	12732	1689	-
1971	8702	17329	14092	17630	5419	63252	11476	15001	26477	1971	22138	19658	21456	6325	13244	1757	-
1972	8352	9517	10533	18141	30192	76832	10025	19103	29128	1972	28633	33053	15146	7683	16865	2238	-
1973	7556	21526	11250	13306	32762	86458	6577	20251	26828	1973	36751	33339	16368	8646	17879	2372	-
1974	7637	21694	10378	11207	12479	63444	3402	17569	20971	1974	24367	22034	17044	6344	15511	2058	-
1975	7289	5681	7737	10239	5389	36384	5126	20272	25398	1975	12014	12128	12241	3638	17897	2375	-
1976	7776	12339	9322	17056	10922	57462	5444	22854	28298	1976	17746	24925	14790	5746	20177	2677	-
1977	7522	17388	9857	24723	6032	65582	3951	19453	23403	1977	22028	24661	18893	6558	18467	985	-
1978	7887	22321	9152	15331	15855	70604	3864	25310	29174	1978	19102	34337	17165	7060	24028	1282	-
1979	6845	14532	10342	24696	4751	61214	2935	31402	34337	1979	13430	24717	23067	6121	29812	1580	-
1980	6540	7727	9025	13718	16809	53871	3305	32966	36271	1980	14084	24796	14991	5387	31296	1670	-
1981	5916	21000	7514	38253	6157	78921	2456	58735	61191	1981	18153	40994	19774	2537	55760	2975	552
1982	7092	30143	9297	15091	4899	65704	2382	31635	34017	1982	17742	26952	21010	4095	30032	1602	1218
1983	5378	14326	10461	39539	74659	74659	2398	72047	74445	1983	12100	43225	19334	5150	68398	3649	1635
1984	6254	12877	10089	12058	9003	50431	4033	33357	37390	1984	12761	20514	17156	8457	31668	1690	1976
1985	5470	31568	8977	37701	23646	107565	4704	56508	61212	1985	25219	58676	23669	7947	53646	2862	1360
1986	5007	30833	13238	29505	25197	103936	5390	48483	53873	1986	26130	53156	24652	6360	46027	2456	1394
1987	5249	15035	8415	26921	11000	66695	5444	43895	49339	1987	16026	29465	21203	6240	41671	2223	1374
1988	5922	11943	7077	32217	30297	87548	5866	62656	68522	1988	19281	50401	17866	7940	55751	6905	1523
1989	5235	34383	8752	31004	9322	88728	4659	57977	62636	1989	20818	42936	25174	7264	51588	6389	1333
1990	5228	37134	10569	26240	17181	96396	3783	62118	65901	1990	23257	47338	25801	7133	55272	6846	726
1991	5058	25211	10053	35096	10236	85680	3241	81405	84646	1991	19410	42543	23725	5638	72434	8971	567
1992	5336	20938	7328	14913	17964	66497	3441	86646	90087	1992	23183	28641	14674	7508	77097	9549	661
1993	4817	42529	4316	16046	17274	84989	4796	79735	84531	1993	28549	38566	17874	8182	70948	8787	677
1994	3574	30828	7713	3383	20323	65827	4498	85704	90202	1994	21186	27427	17214	5151	76259	9445	772
1995	3574	10533	4866	19767	12115	48794	4320	79057	83377	1995	15081	21339	17214	3557	70345	8712	860
1996	456	15647	3869	8620	6555	35147	5453	81304	86757	1996	15922	14129	5131	3208	72344	8960	1004
1997	1658	25305	750	12217	8676	48606	6901	95708	102608	1997	14339	25567	8700	3855	85160	10547	1041
1998	1401	5170	25	3921	19913	30429	7573	93986	101560	1998	7820	19657	2952	1959	83829	10358	1079
1999	768	1811	28	9515	4981	17112	7106	121584	128700	1999	4945	11465	702	2328	108194	13400	504
2000	521	8860	34	7219	2843	19476	6213	56822	63035	2000	8180	10556	740	2259	50560	6262	1000
2001	666	7184	53	10970	5850	24724	4937	88985	93823	2001	9568	14142	1014	2438	79179	9807	1196
2002	1675	10065	460	8609	12341	33151	6495	106836	113331	2002	12356	17761	3033	2722	95062	11774	606
2003	2099	6181	753	15321	13430	37785	5370	110054	115424	2003	12417	21872	3496	949	97926	12128	364

Caveats

- a Salmon sport landings assumed 5% of total catch prior to 1960, 10% of total catch prior to 1981 when data series begins.
b Trawl/hook and line data estimated from total groundfish landings; relative landings per gear type estimated from example years in data series.
c Groundfish sport catch assumed negligible prior to beginning of time series.

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Appendix 4.4 Average Species Weight

Table A4.4.1 Average species weight

References listed in Appendix Table A9.1.2

Species	Weight (kg)	Ref #
Sockeye	2	131
Coho	4	132
Pink	1.85	133
Chum	4.5	134
Chinook	14	135
Halibut	10	136
Lingcod	5.4	137
Rockfish ¹	1.5	138

Reproduced from [139]

¹Average of roughey (*Sebastes aleutianus*)
and yellowmouth rockfish (*S. reedi*).

Appendix 5.1 Ecopath Parameters

Table A5.1.1 Species aggregation by functional group.

Functional group names are given in bold.

Common name	Scientific name
Sea Otters	
Sea otter	<i>Enhydra lutra</i>
Mysticetae	
blue whale	<i>Balaenoptera musculus</i>
fin whale	<i>Balaenoptera physalus</i>
sei whale	<i>Balaenoptera borealis</i>
humpback whale	<i>Megaptera novaeangliae</i>
right whale	<i>Balaena glacialis</i>
gray whale	<i>Eschrichtius robustus</i>
Odontocetae	
sperm whale	<i>Physeter macrocephalus</i>
Baird's beaked whale	<i>Berardius bairdii</i>
northern right whale dolphin	<i>Lissodelphis borealis</i>
Pacific white-sided dolphin	<i>Lagenorhynchus obliquidens</i>
Dall's porpoise	<i>Phocoenoides dalli</i>
harbour porpoise	<i>Phocoena phocoena</i>
killer whale	<i>Orcinus orca</i>
Seals and sea lions	
Steller sea lion	<i>Eumetopias jubatus</i>
harbour seal	<i>Phoca vitulina</i>
northern fur seal	<i>Callorhinus ursinus</i>
northern elephant seal	<i>Mirounga angustirostris</i>
California sea lion	<i>Zalophus californianus</i>
Seabirds	
gulls	<i>Laridae</i>
grebes	<i>Podicipedidae</i>
Cassin's auklet	<i>Ptychoramphus aleuticus</i>
tufted puffin	<i>Fratercula comiculata</i>
common murre	<i>Uria aalge</i>
rhinoceros auklet	<i>Cerorhinca monocerata</i>
marbled murrelet	<i>Brachyramphus marmoratus</i>
pigeon guillemot	<i>Cepphus columba</i>
merganser spp.	<i>Mergus serrator</i> , <i>M. merganser</i>
pelagic cormorants	<i>Phalacrocorax pelagicus</i>
sooty shearwater	<i>Puffinus griseus</i>
northern fulmar	<i>Fulmarus glacialis</i>
double-crested cormorant	<i>Phalacrocorax auritus</i>
common loon	<i>Gavia immer</i>
Transient salmon	
sockeye	<i>Oncorhynchus nerka</i>
chum	<i>Oncorhynchus keta</i>
pink salmon	<i>Oncorhynchus gorbuscha</i>

Table A5.1.1 Species aggregation by functional group (cont.)

Common name	Scientific name
Coho and chinook salmon	
coho salmon	<i>Oncorhynchus kisutch</i>
chinook salmon	<i>Oncorhynchus tshawytscha</i>
Large and small squid	
common squid	<i>Loligo opalescens</i>
Ratfish	
ratfish	<i>Hydrolagus collei</i>
Dogfish	
dogfish	<i>Squalus acanthias</i>
Juvenile and adult pollock	
walleye pollock	<i>Theragra chalcogramma</i>
Forage fish and eulachon	
sandlance	<i>Ammodytes hexapterus</i>
pilchards	<i>Sardinops sagax</i>
anchovy	<i>Engraulis mordax</i>
capelin	<i>Mallotus villosus</i>
chub mackerel	<i>Scomber japonicus</i>
shad	<i>Alosa sapidissima</i>
smelts	<i>Osmeridae</i>
eulachon	<i>Thaleichthys pacificus</i>
Juvenile and adult herring	
Pacific herring	<i>Clupea pallasii</i>
Juvenile and adult Pacific ocean perch	
Pacific Ocean perch	<i>Sebastes alutus</i>
Inshore rockfish	
copper rockfish	<i>Sebastes caurinus</i>
quillback rockfish	<i>Sebastes maliger</i>
tiger rockfish	<i>Sebastes nigrocinctus</i>
China rockfish	<i>Sebastes nebulosus</i>
yelloweye rockfish	<i>Sebastes ruberrimus</i>
Juvenile and adult piscivorous rockfish	
roughey rockfish	<i>Sebastes aleutianus</i>
shortraker rockfish	<i>Sebastes borealis</i>
shortspine thornyhead	<i>Sebastolobus altivelis</i>
longspine thornyhead	<i>Sebastolobus alascanus</i>
black rockfish	<i>Sebastes melanops</i>
blue rockfish	<i>Sebastes mystinus</i>
chillipepper	<i>Sebastes goodei</i>
dusky rockfish	<i>Sebastes ciliatus</i>

Table A5.1.1 Species aggregation by functional group (cont.)

Common name	Scientific name
Juvenile and adult planktivorous rockfish	
yellowmouth rockfish	<i>Sebastes reedi</i>
red-stripe rockfish	<i>Sebastes proriger</i>
widow rockfish	<i>Sebastes entomelas</i>
yellowtail rockfish	<i>Sebastes flavidus</i>
darkblotch rockfish	<i>Sebastes cremeri</i>
canary rockfish	<i>Sebastes pinniger</i>
splitnose rockfish	<i>Sebastes diploproa</i>
sharpchin rockfish	<i>Sebastes zacentrus</i>
Puget sound rockfish	<i>Sebastes emphaeus</i>
bocaccio	<i>Sebastes paucispinis</i>
shortbelly rockfish	<i>Sebastes jordani</i>
Juvenile and adult turbot	
arrowtooth flounder	<i>Atheresthes stomias</i>
Juvenile and adult flatfish	
rock sole	<i>Lepidosetta bilineata</i>
English sole	<i>Parophrys vetulus</i>
dover sole	<i>Microstomas pacificus</i>
Juvenile and adult halibut	
halibut	<i>Hippoglossus stenolepis</i>
Juvenile and adult Pacific cod	
Pacific cod	<i>Gadus macrocephalus</i>
Juvenile and adult sablefish	
Sablefish	<i>Anoplopoma fimbria</i>
Juvenile and adult lingcod	
lingcod	<i>Ophiodon elongatus</i>
Shallow water benthic fish	
sculpins	<i>Cottidae</i>
blennies	<i>Blennidae</i>
poachers	<i>Agonidae</i>
gobies	<i>Gobiedae</i>
greenlings	<i>Hexagramidae</i>
rock greenling	<i>Hexagrammos lagocephalus</i>
eelpouts	<i>Zoarcidae</i>
northern clingfish	<i>Gobiesox maeandricus</i>
red irish lords	<i>Hemilepidotus hemilepidotus</i>
cabezon	<i>Scorpaenichthys marmoratus</i>
cutthroat trout	<i>Oncorhynchus clarki clarki</i>
white sturgeon	<i>Acipenser transmontanus</i>

Table A5.1.1 Species aggregation by functional group (cont.)

Common name	Scientific name
Skates	
big skate	<i>Raja binoculata</i>
longnose skate	<i>Raja rhina</i>
starry skate	<i>Raja stellulata</i>
black skate	<i>Raja kincaidi</i>
deep-sea skate	<i>Raja abyssicola</i>
tope shark	<i>Galeorhinus galeus</i>
great white shark	<i>Carcharodon carcharias</i>
broadnose sevengill shark	<i>Notorynchus cepedianus</i>
bluntnose sixgill shark	<i>Hexanchus griseus</i>
blue shark	<i>Prionace glauca</i>
basking shark	<i>Cetorhinus maximus</i>
diamond stingray	<i>Dasyatis dipterura</i>
Pelagic stingray	<i>Pteroplatytrygon violacea</i>
Large and small crabs	
Dungeness crab	<i>Cancer magister</i>
red rock crab	<i>Cancer productus</i>
tanner crab	<i>Chionecetes sp.</i>
king crab	<i>Paralithodes sp.</i>
kelp crab	<i>Pugettia producta</i>
Commercial shrimp	
smooth shrimp	<i>Pandalus jordani</i>
spiny shrimp	<i>Pandalus borealis eous</i>
pink shrimp	<i>Pandalus goniurus</i>
coonstripe shrimp	<i>Pandalus danae</i>
humpback shrimp	<i>Pandalus hypsinotus</i>
sidestripe shrimp	<i>Pandalopsis disbar</i>
prawn	<i>Pandalus platycterus</i>
Macrophytes	
bull kelp	<i>Nereocystis leutkeana</i>
giant kelp	<i>Macrocystis integrifolia</i>

Table A5.1.2 Basic parameters for all periods

#	Groups	Biomass (t•km ⁻²)				Biomass accumulation (t•km ⁻² •yr ⁻¹)				Production/Biomass ratio (yr ⁻¹)				Consumption/Biomass ratio (yr ⁻¹)			
		1750	1900	1950	2000	1750	1900	1950	2000	1750	1900	1950	2000	1750	1900	1950	2000
1	Sea otters	0.0016	0.0001	0.00005	0.0001	0.000	0.000	1.E-06	0.000	0.130	0.130	0.130	0.130	101.50	101.50	101.50	101.50
2	Mysticetae	2.672	1.541	1.607	1.339	0.000	0.028	-0.014	0.004	0.020	0.060	0.040	0.020	8.00	8.00	9.10	9.10
3	Odontocetae	0.066	0.066	0.061	0.061	0.000	0.000	0.001	0.000	0.040	0.040	0.020	0.040	15.50	15.50	15.50	15.50
4	Seals, sea lions	0.080	0.069	0.130	0.258	0.000	0.001	-0.001	0.000	0.060	0.100	0.060	0.060	15.10	15.10	15.10	15.10
5	Seabirds	0.007	0.015	0.007	0.007	0.000	0.000	0.000	0.0000	0.100	0.100	0.100	0.100	105.20	105.20	105.20	105.20
6	Transient salmon	1.008	0.840	0.500	0.208	0.000	0.000	-0.025	-0.028	0.517	2.480	2.480	2.480	3.72	3.72	8.33	8.33
7	Coho salmon	0.096	0.160	0.100	0.024	0.000	0.000	-0.010	0.000	1.157	2.760	2.760	2.760	4.00	10.00	13.80	13.80
8	Chinook salmon	0.144	0.160	0.090	0.036	0.000	0.000	0.000	0.000	0.366	2.160	2.160	2.160	2.82	10.00	10.80	10.80
9	Small squid	1.207	0.795	1.090	1.090	0.000	0.000	-0.109	-0.006	6.023	6.023	6.023	6.023	34.68	34.68	34.67	34.68
10	Squid	0.399	0.259	0.765	0.765	0.000	0.026	-0.038	0.027	6.023	6.023	6.023	6.023	34.68	34.68	34.67	34.68
11	Ratfish	0.262	0.183	0.517	0.517	0.000	0.000	0.003	-0.011	0.199	0.199	0.099	0.099	1.40	1.40	1.40	1.40
12	Dogfish	1.364	0.600	0.417	0.909	0.000	0.030	0.004	-0.060	0.110	0.200	0.150	0.099	3.33	3.33	2.72	2.72
13	Juvenile pollock	1.318	0.926	0.132	0.132	0.000	0.000	0.000	0.002	0.230	0.230	1.060	1.061	5.05	5.05	5.31	5.31
14	Pollock	0.622	0.479	0.359	0.359	0.000	0.000	0.022	0.011	0.153	0.154	0.450	0.263	3.36	3.36	1.17	1.17
15	Forage fish	32.499	24.603	7.600	8.478	0.000	0.000	-0.076	0.130	0.595	0.588	1.743	1.432	6.61	6.61	8.40	8.40
16	Eulachon	7.315	5.033	1.893	1.660	0.000	0.000	0.000	0.049	0.600	0.700	1.432	1.432	6.61	6.61	8.40	8.40
17	Juvenile herring	5.446	3.729	1.317	2.265	0.000	0.000	0.000	0.061	1.173	2.000	2.190	2.190	11.26	11.26	10.95	10.95
18	Adult herring	7.503	2.480	1.001	0.658	0.000	0.000	-0.050	-0.003	0.792	0.900	1.540	0.683	7.51	7.51	5.84	5.84
19	Juvenile POP	0.213	0.153	0.036	0.062	0.000	0.000	-0.004	0.001	0.338	0.338	0.500	0.672	6.12	6.12	3.21	3.21
20	Adult POP	1.404	1.011	1.019	0.561	0.000	0.000	0.000	0.000	0.227	0.227	0.144	0.144	4.08	4.08	2.14	2.14
21	Inshore rockfish	0.096	0.081	0.095	0.100	0.000	0.000	-0.002	0.001	0.182	0.182	0.190	0.190	3.70	5.54	5.69	5.69
22	Juvenile piscivorous rockfish	0.020	0.016	0.008	0.007	0.000	0.000	0.000	0.000	0.261	0.261	0.261	0.261	1.89	1.89	1.89	1.89
23	Adult piscivorous rockfish	0.137	0.119	0.541	0.654	0.000	0.000	0.004	-0.006	0.037	0.037	0.037	0.037	1.26	1.26	1.26	1.26
24	Juvenile planktivorous rockfish	0.207	0.134	0.189	0.136	0.000	0.000	-0.009	0.005	0.261	0.261	0.261	0.261	3.21	3.21	3.21	3.21
25	Adult planktivorous rockfish	2.086	1.286	1.213	1.207	0.000	0.000	0.006	-0.027	0.068	0.068	0.068	0.068	2.14	2.14	2.14	2.14

Table A5.1.2 Basic parameters for all periods (cont.)

#	Groups	Biomass (t·km ⁻²)				Biomass accumulation (t·km ⁻² ·yr ⁻¹)				Production/Biomass ratio (yr ⁻¹)				Consumption/Biomass ratio (yr ⁻¹)			
		1750	1900	1950	2000	1750	1900	1950	2000	1750	1900	1950	2000	1750	1900	1950	2000
26	Juvenile turbot	0.248	0.170	0.218	0.218	0.000	0.000	-0.013	-0.003	0.330	0.330	0.559	0.330	2.17	2.17	2.17	2.17
27	Adult turbot	2.196	1.530	1.530	1.530	0.000	0.000	0.000	-0.050	0.220	0.300	0.220	0.220	1.98	1.98	1.98	1.98
28	Juvenile flatfish	2.583	1.606	0.259	0.259	0.000	0.000	0.000	0.007	0.382	0.382	1.000	1.935	6.31	6.31	6.02	6.02
29	Adult flatfish	1.765	1.014	0.535	0.236	0.000	0.000	-0.080	0.006	0.257	0.257	0.450	0.949	4.21	4.21	4.27	4.27
30	Juvenile halibut	0.445	0.296	0.406	0.628	0.000	0.000	0.000	-0.011	0.099	0.450	0.550	0.600	2.56	2.56	1.46	1.46
31	Adult halibut	1.000	0.608	0.429	0.628	0.000	0.000	0.000	0.000	0.067	0.220	0.450	0.400	1.70	1.70	1.10	1.10
32	Juvenile Pacific cod	0.464	0.307	0.185	0.089	0.000	0.000	-0.009	0.005	0.258	0.258	1.980	1.980	3.43	3.43	7.50	7.50
33	Adult Pacific cod	2.039	1.219	0.348	0.163	0.000	0.000	0.000	-0.020	0.174	0.174	1.320	1.320	2.29	2.29	4.00	4.00
34	Juvenile sablefish	0.180	0.108	0.238	0.119	0.000	0.000	-0.002	0.002	0.273	0.273	0.600	0.600	7.00	7.00	7.00	7.00
35	Adult sablefish	0.191	0.602	0.600	0.269	0.000	0.000	-0.030	0.000	0.183	0.184	0.276	0.276	3.73	3.73	3.73	3.73
36	Juvenile lingcod	0.006	0.005	0.078	0.031	0.000	0.000	-0.016	0.002	0.389	0.389	1.400	1.200	3.94	3.94	3.50	3.30
37	Adult lingcod	0.148	0.119	0.104	0.039	0.000	0.000	0.001	-0.001	0.262	0.300	0.800	0.800	2.80	2.80	3.00	3.30
38	Shallow-water benthic fish	7.506	4.464	0.509	0.509	0.000	0.000	0.025	0.006	0.266	0.266	1.500	1.500	2.10	2.10	5.26	5.26
39	Skates	0.239	0.167	0.300	0.335	0.000	0.000	0.006	-0.003	0.150	0.150	0.310	0.310	1.20	1.20	1.24	1.24
40	Large crabs	0.652	0.388	0.506	0.456	0.000	0.000	-0.051	-0.002	1.500	1.500	1.500	1.500	5.00	5.00	5.00	5.00
41	Small crabs	2.407	1.458	0.599	0.650	0.000	0.000	0.000	-0.010	3.500	3.500	3.500	3.500	14.00	14.00	8.75	14.00
42	Commercial shrimp	0.070	0.047	0.150	0.200	0.000	0.000	0.000	0.002	5.700	5.700	11.480	11.475	22.80	22.80	45.90	45.90
43	Epifaunal invertebrates	42.833	28.604	11.191	13.448	0.000	0.000	-0.224	-0.194	1.448	1.448	1.448	1.448	16.09	16.09	4.05	16.09
44	Infauanal camivorous invertebrates	8.205	13.245	13.245	13.245	0.000	0.000	0.000	0.270	2.000	2.000	2.000	2.000	22.22	22.22	22.22	22.22
45	Infauanal invertebrate detritivores	39.279	34.305	34.305	34.305	0.000	0.000	0.000	-0.377	1.296	1.300	2.848	1.349	14.40	14.44	14.99	14.99
46	Carnivorous jellyfish	4.625	3.363	3.000	3.000	0.000	0.000	0.000	-0.008	18.00	18.00	18.00	18.00	60.00	60.00	60.00	60.00
47	Euphausiids	22.660	12.606	8.700	8.700	0.000	0.000	0.000	-0.042	6.00	6.00	6.00	6.10	24.82	24.82	24.82	24.82
48	Copepods	13.127	8.671	4.667	4.667	0.000	0.000	0.000	-0.018	27.000	27.000	27.000	27.000	99.00	99.00	90.00	90.00
49	Corals and sponges	1.929	1.929	1.929	1.929	0.000	0.000	0.000	0.000	0.010	0.010	0.010	0.010	2.00	2.00	2.00	2.00
50	Macrophytes	10.560	5.280	5.280	5.280	0.000	0.000	0.000	0.000	5.256	5.256	5.256	5.256	-	-	-	-
51	Phytoplankton	15.406	15.406	15.406	15.406	0.000	0.000	0.000	26.118	178.5	178.5	178.5	178.5	-	-	-	-
52	Discards	0.000	0.001	0.072	0.072	-	-	-	-	-	-	-	-	-	-	-	-
Biodiversity (Q-90)		7.44	5.84	4.35	3.54												

Table A5.1.3 Diet composition

# Group	Period			
1 Sea Otters	1750	1900	1950	2000
Small squid	0.045	0.045	0.045	0.045
Squid	0.045	0.045	0.045	0.045
J. pollock	0.100	0.100	0.100	0.100
S. benthic fish	0.100	0.100	0.100	0.100
Large crabs	0.010	0.010	0.010	0.010
Small crabs	0.200	0.200	0.200	0.200
Epifaunal inv.	0.500	0.500	0.500	0.500
2 Mysticetae	Mysticetae	1900	1950	2000
	1750			
Forage fish	0.014	0.372	0.014	0.014
A. herring	0.101	0.001	0.001	0.001
Epifaunal inv.	0.035	0.035	0.035	0.035
Inf. cam. inv.	0.045	0.045	0.045	0.045
Inf. det. inv.	0.258	0.300	0.658	0.658
Euphausiids	0.420	0.226	0.226	0.226
Copepods	0.127	0.021	0.021	0.021
3 Odontocetae	1750	1900	1950	2000
Sea otters	<0.001	<0.001	<0.001	<0.001
Seals, sea lions	<0.001	<0.001	<0.001	<0.001
Transient salmon	0.041	0.046	0.041	0.041
Coho salmon	0.011	0.031	0.011	0.004
Chinook salmon	0.005	0.050	0.005	0.004
Small squid	0.213	0.283	0.213	0.213
Squid	0.213	0.094	0.213	0.213
Ratfish	0.016	0.001	0.026	0.026
Forage fish	0.162	0.162	0.162	0.162
Eulachon	0.032	0.032	0.032	0.032
J. herring	0.026	0.026	0.026	0.056
A. herring	0.140	0.065	0.040	0.010
J. POP	0.005	<0.001	0.005	0.005
J. plank. rockfish	0.008	0.008	0.008	0.002
A. plank. rockfish	0.011	0.011	0.011	0.011
J. turbot	0.015	0.015	0.015	0.015
A. halibut	<0.001	0.015	0.011	0.028
Large crabs	0.042	0.021	0.021	0.028
Euphausiids	0.052	0.132	0.152	0.142
Import	0.008	0.008	0.008	0.008
4 Seals, sea lions	1750	1900	1950	2000
Transient salmon	0.100	0.110	0.100	0.010
Coho salmon	0.002	0.002	0.002	0.001
Chinook salmon	0.015	0.005	0.015	0.004
Small squid	0.045	0.059	0.045	0.045
Squid	0.045	0.045	0.045	0.052
Dogfish	0.002	0.002	0.002	0.021
Pollock	0.040	0.040	0.040	0.001
Forage fish	0.070	0.070	0.070	0.100
J. herring	0.200	0.100	0.200	0.210
A. herring	0.127	0.179	0.027	0.010
A. POP	0.031	0.021	0.031	0.001
Inshore rockfish	0.002	0.002	0.002	<0.001
J. pisc. rockfish	<0.001	<0.001	<0.001	<0.001
A. pisc. rockfish	0.001	0.004	0.001	<0.001
J. plank. rockfish	0.004	0.001	0.004	<0.001
A. plank. rockfish	0.005	0.001	0.005	0.002
A. turbot	0.062	0.062	0.062	0.022
A. flatfish	0.049	0.049	0.049	0.004
J. halibut	0.010	0.010	0.010	0.061
A. halibut	0.005	0.009	0.005	0.014
A. Pacific cod	0.091	0.061	0.121	0.007
J. sablefish	0.005	0.005	0.005	0.005
A. sablefish	0.001	<0.001	0.052	0.003
A. lingcod	0.010	0.010	0.010	<0.001
S. benthic fish	0.010	0.010	0.010	0.036
Epifaunal inv.	0.041	0.116	0.060	0.194
Inf. cam. inv.	0.028	0.028	0.028	0.195
5 Seabirds	1750	1900	1950	2000
Transient salmon	0.054	0.054	0.054	0.054
Small squid	0.035	0.035	0.035	0.035
Squid	0.055	0.055	0.055	0.055
Forage fish	0.243	0.243	0.243	0.243
Eulachon	0.079	0.005	0.079	0.079
J. herring	0.105	0.025	0.105	0.105
A. herring	0.113	0.083	0.003	0.003
Small crabs	0.041	0.041	0.041	0.041
Epifaunal inv.	0.041	0.115	0.041	0.041
Cam. jellyfish	0.036	0.036	0.036	0.036
Euphausiids	0.112	0.112	0.112	0.112
Copepods	0.046	0.156	0.156	0.156
Import	0.041	0.041	0.041	0.041
6 Transient salmon	1750	1900	1950	2000
Cam. jellyfish	0.100	0.100	0.100	0.100
Euphausiids	0.150	0.150	0.150	0.150
Copepods	0.150	0.150	0.150	0.150
Import	0.600	0.600	0.600	0.600
7 Coho salmon	1750	1900	1950	2000
Small squid	0.200	0.200	0.200	0.200
Squid	0.190	0.175	0.190	0.190
Forage fish	0.167	0.167	0.167	0.167
Eulachon	0.033	0.063	0.033	0.033
A. herring	0.200	0.040	0.040	0.040
J. Pacific cod	0.010	0.005	0.010	0.010
Euphausiids	0.200	0.350	0.360	0.360
8 Chinook salmon	1750	1900	1950	2000
Forage fish	0.353	0.353	0.353	0.433
Eulachon	0.067	0.067	0.067	0.067
A. herring	0.180	0.180	0.180	0.050
Euphausiids	0.400	0.400	0.400	0.450
9 Small squid	1750	1900	1950	2000
Forage fish	0.131	0.011	0.081	0.081
Eulachon	0.014	0.001	0.014	0.014
J. herring	0.035	0.001	0.030	0.050
Cam. jellyfish	0.085	0.142	0.060	0.060
Euphausiids	0.350	0.430	0.430	0.380
Copepods	0.099	0.129	0.099	0.099
Detritus	0.286	0.286	0.286	0.316
10 Squid	1750	1900	1950	2000
Small squid	0.174	0.174	0.174	0.174
Squid	0.001	0.010	0.100	0.100
Forage fish	0.187	0.151	0.187	0.187
Eulachon	0.025	0.001	0.025	0.025
Cam. jellyfish	0.060	0.210	0.060	0.060
Euphausiids	0.187	0.107	0.107	0.107
Copepods	0.060	0.041	0.041	0.041
Detritus	0.307	0.307	0.307	0.307
11 Ratfish	1750	1900	1950	2000
Forage fish	0.278	0.278	0.278	0.278
Eulachon	0.056	0.056	0.056	0.056
Epifaunal inv.	0.183	0.183	0.183	0.183
Inf. cam. inv.	0.070	0.070	0.070	0.200
Inf. det. inv.	0.080	0.080	0.080	0.080
Euphausiids	0.333	0.333	0.333	0.203

Table A5.1.3 Diet composition (cont.)

# Group	Period			
12 Dogfish	1750	1900	1950	2000
Transient salmon	0.054	0.024	0.054	0.014
Coho salmon	0.015	0.050	0.015	0.003
Chinook salmon	0.001	0.020	0.010	0.004
Small squid	0.077	0.125	0.064	0.065
Squid	0.102	0.032	0.102	0.117
Ratfish	0.006	0.004	0.006	0.006
Forage fish	0.072	0.072	0.072	0.072
Eulachon	0.015	0.015	0.015	0.015
J. herring	0.023	0.023	0.023	0.023
A. herring	0.249	0.010	0.010	0.010
J. POP	0.001	0.001	0.001	0.001
A. POP	0.003	0.003	0.003	0.003
J. plank. rockfish	<0.001	<0.001	0.003	0.003
A. plank. rockfish	0.005	0.012	0.005	0.002
J. turbot	0.005	0.001	0.005	0.005
A. turbot	0.020	0.020	0.020	0.020
J. flatfish	0.010	0.010	0.010	0.020
A. flatfish	0.020	0.020	0.020	0.020
A. Pacific cod	0.001	0.010	0.010	0.010
J. sablefish	0.001	0.001	0.014	0.005
A. sablefish	0.001	0.005	0.020	0.004
S. benthic fish	0.017	0.017	0.017	0.062
Large crabs	0.037	0.017	0.037	0.037
Small crabs	0.026	0.026	0.026	0.026
Epifaunal inv.	0.092	0.062	0.052	0.052
Inf. cam. inv.	0.019	0.024	0.019	0.019
Inf. det. inv.	0.008	0.008	0.008	0.008
Carn. jellyfish	0.029	0.029	0.029	0.029
Euphausiids	0.020	0.160	0.149	0.139
Copepods	0.020	0.070	0.130	0.126
Detritus	0.051	0.129	0.051	0.080
13 J. pollock	1750	1900	1950	2000
J. pollock	0.015	0.001	0.025	0.025
Forage fish	0.072	0.096	0.072	0.072
Eulachon	0.014	0.014	0.014	0.014
Epifaunal inv.	0.081	0.071	0.071	0.071
Inf. cam. inv.	0.020	0.020	0.020	0.020
Euphausiids	0.369	0.369	0.369	0.300
Copepods	0.429	0.429	0.429	0.498
14 Pollock	1750	1900	1950	2000
Small squid	0.016	0.016	0.016	0.016
Squid	0.016	0.016	0.016	0.016
J. pollock	0.050	0.001	0.100	0.100
Forage fish	0.198	0.288	0.198	0.198
Eulachon	0.028	0.037	0.028	0.028
Epifaunal inv.	0.072	0.022	0.022	0.022
Euphausiids	0.461	0.461	0.461	0.461
Copepods	0.160	0.160	0.160	0.160
15 Forage fish	1750	1900	1950	2000
Epifaunal inv.	0.100	0.100	0.100	0.100
Carn. jellyfish	0.270	0.205	0.270	0.270
Euphausiids	0.080	0.115	0.080	0.200
Copepods	0.500	0.520	0.500	0.380
Detritus	0.050	0.060	0.050	0.050
16 Eulachon	1750	1900	1950	2000
Epifaunal inv.	0.100	0.100	0.100	0.100
Carn. jellyfish	0.200	0.200	0.200	0.200
Euphausiids	0.100	0.100	0.100	0.100
Copepods	0.600	0.600	0.600	0.600
17 J. herring	1750	1900	1950	2000
Euphausiids	0.100	0.100	0.100	0.100
Copepods	0.900	0.900	0.900	0.900
18 A. herring	1750	1900	1950	2000
Euphausiids	0.900	0.900	0.900	0.900
Copepods	0.100	0.100	0.100	0.100
19 J. POP	1750	1900	1950	2000
Euphausiids	0.200	0.200	0.200	0.200
Copepods	0.800	0.800	0.800	0.800
20 A. POP	1750	1900	1950	2000
Small squid	0.012	0.012	0.012	0.012
Squid	0.012	0.012	0.012	0.012
Euphausiids	0.893	0.893	0.893	0.893
Copepods	0.083	0.083	0.083	0.083
21 Inshore rockfish	1750	1900	1950	2000
Forage fish	0.150	0.060	0.060	0.060
Eulachon	0.012	0.012	0.012	0.012
J. herring	0.300	0.100	0.300	0.330
A. herring	0.050	0.250	0.050	0.020
S. benthic fish	0.045	0.065	0.045	0.045
Large crabs	0.005	0.005	0.005	0.005
Small crabs	0.107	0.107	0.107	0.107
Comm. shrimp	0.005	0.075	0.095	0.095
Epifaunal inv.	0.050	0.050	0.050	0.050
Inf. cam. inv.	0.124	0.124	0.124	0.124
Euphausiids	0.152	0.152	0.152	0.152
22 J. pisc. rockfish	1750	1900	1950	2000
Small crabs	0.100	0.100	0.100	0.100
Comm. shrimp	0.100	0.100	0.100	0.100
Epifaunal inv.	0.100	0.100	0.100	0.100
Euphausiids	0.400	0.400	0.400	0.400
Copepods	0.300	0.300	0.300	0.300
23 A. pisc. rockfish	1750	1900	1950	2000
Small squid	0.070	0.070	0.070	0.070
Squid	0.070	0.070	0.070	0.080
Forage fish	0.027	0.027	0.027	0.027
Eulachon	0.005	0.005	0.005	0.005
J. POP	0.005	0.005	0.005	0.020
J. pisc. rockfish	0.002	0.002	0.002	<0.001
Skates	0.068	0.068	0.068	0.028
Small crabs	0.130	0.130	0.130	0.130
Epifaunal inv.	0.254	0.254	0.254	0.254
Inf. det. inv.	0.040	0.040	0.040	0.150
Carn. jellyfish	0.038	0.038	0.038	0.038
Euphausiids	0.196	0.196	0.196	0.086
Detritus	0.096	0.096	0.096	0.112
24 J. plank. rockfish	1750	1900	1950	2000
Euphausiids	0.500	0.500	0.500	0.500
Copepods	0.500	0.500	0.500	0.500

Table A5.1.3 Diet composition (cont.)

# Group	Period			
25 A. plank. rockfish	1750	1900	1950	2000
Small squid	0.101	0.101	0.101	0.101
Squid	0.101	0.061	0.101	0.101
Forage fish	0.028	0.028	0.028	0.028
Eulachon	0.006	0.006	0.006	0.006
J. herring	0.109	0.109	0.109	0.109
Cam. jellyfish	0.006	0.006	0.006	0.006
Euphausiids	0.574	0.574	0.574	0.574
Copepods	0.075	0.115	0.075	0.075
26 J. turbot	1750	1900	1950	2000
Small squid	0.087	0.087	0.087	0.087
Squid	0.087	0.087	0.087	0.087
J. pollock	0.030	0.030	0.030	0.030
Pollock	0.070	0.070	0.070	0.070
Forage fish	0.111	0.111	0.111	0.111
Eulachon	0.022	0.022	0.022	0.022
J. herring	0.150	0.150	0.150	0.180
A. herring	0.067	0.048	0.048	0.018
Inshore rockfish	0.007	0.007	0.007	0.007
J. pisc. rockfish	0.001	<0.001	0.001	0.001
J. plank. rockfish	0.017	0.007	0.017	0.017
J. flatfish	0.033	0.044	0.033	0.033
Small crabs	0.100	0.100	0.100	0.100
Comm. shrimp	0.001	0.100	0.100	0.100
Epifaunal inv.	0.113	0.033	0.033	0.033
Euphausiids	0.074	0.074	0.074	0.074
Detritus	0.030	0.030	0.030	0.030
27 A. turbot	1750	1900	1950	2000
Small squid	0.161	0.221	0.161	0.161
Squid	0.161	0.101	0.161	0.161
J. pollock	0.009	0.009	0.009	0.009
Pollock	0.001	0.001	0.001	0.001
Forage fish	0.107	0.107	0.107	0.107
Eulachon	0.051	0.051	0.051	0.051
J. herring	0.099	0.001	0.099	0.099
A. herring	0.050	0.010	0.010	<0.001
J. POP	0.001	0.001	0.001	0.001
A. POP	0.015	0.015	0.015	0.001
J. turbot	0.001	0.001	0.001	0.001
J. flatfish	0.005	0.005	0.005	0.005
A. flatfish	0.005	0.005	0.005	0.005
A. Pacific cod	0.003	0.003	0.003	0.003
J. lingcod	<0.001	<0.001	0.020	0.002
S. benthic fish	0.079	0.079	0.078	0.088
Large crabs	0.034	0.015	0.015	0.015
Comm. shrimp	0.050	0.001	0.050	0.050
Epifaunal inv.	0.024	0.024	0.024	0.028
Euphausiids	0.050	0.118	0.050	0.050
Detritus	0.095	0.233	0.135	0.163
28 J. flatfish	1750	1900	1950	2000
Forage fish	0.050	0.050	0.050	0.050
Eulachon	0.010	0.010	0.010	0.010
Small crabs	0.068	0.068	0.068	0.068
Epifaunal inv.	0.046	0.046	0.046	0.046
Inf. cam. inv.	0.373	0.373	0.373	0.373
Inf. det. inv.	0.453	0.453	0.453	0.453
29 A. flatfish	1750	1900	1950	2000
Forage fish	0.154	0.154	0.154	0.154
Eulachon	0.031	0.031	0.031	0.031
J. lingcod	<0.001	<0.001	0.005	0.005
Small crabs	0.038	0.039	0.038	0.038
Epifaunal inv.	0.052	0.051	0.051	0.051
Inf. cam. inv.	0.272	0.272	0.272	0.272
Inf. det. inv.	0.445	0.445	0.445	0.445
Euphausiids	0.008	0.008	0.004	0.004
# Group	Period			
30 J. halibut	1750	1900	1950	2000
Small squid	0.032	0.032	0.032	0.032
Squid	0.032	0.032	0.032	0.032
Forage fish	0.055	0.055	0.055	0.055
Eulachon	0.011	0.011	0.011	0.011
J. herring	0.050	0.050	0.050	0.050
J. POP	0.003	0.003	0.003	0.003
J. turbot	0.001	<0.001	0.100	0.010
J. flatfish	0.020	0.020	0.020	0.020
J. Pacific cod	0.008	0.008	0.008	0.008
S. benthic fish	0.105	0.055	0.055	0.055
Large crabs	0.200	0.280	0.200	0.240
Small crabs	0.309	0.300	0.300	0.300
Comm. shrimp	0.001	0.060	0.060	0.060
Epifaunal inv.	0.050	0.070	0.050	0.050
Detritus	0.124	0.025	0.025	0.075
31 A. halibut	1750	1900	1950	2000
Small squid	0.064	0.064	0.064	0.064
Squid	0.064	0.064	0.064	0.064
Forage fish	0.014	0.014	0.014	0.014
Eulachon	0.003	0.003	0.003	0.003
A. herring	0.072	0.020	0.020	0.020
A. POP	0.020	0.020	0.020	0.005
J. plank. rockfish	0.005	0.005	0.005	0.005
A. plank. rockfish	0.001	0.001	0.001	0.001
A. turbot	0.074	0.074	0.074	0.090
A. flatfish	0.123	0.053	0.123	0.100
J. halibut	0.001	0.001	0.080	0.080
A. Pacific cod	0.005	0.055	0.100	0.050
A. lingcod	<0.001	0.005	0.040	0.005
S. benthic fish	0.095	-	-	-
Skates	0.001	0.003	0.053	0.053
Large crabs	0.155	0.155	0.155	0.155
Small crabs	0.110	0.189	0.110	0.125
Epifaunal inv.	0.159	0.110	0.040	0.090
Detritus	0.035	0.165	0.035	0.077
32 J. Pacific cod	1750	1900	1950	2000
Small crabs	0.027	0.027	0.027	0.027
Epifaunal inv.	0.210	0.210	0.210	0.210
Inf. cam. inv.	0.079	0.079	0.079	0.079
Inf. det. inv.	0.363	0.363	0.363	0.363
Euphausiids	0.115	0.115	0.115	0.115
Copepods	0.057	0.057	0.057	0.057
Detritus	0.149	0.149	0.149	0.149
33 A. Pacific cod	1750	1900	1950	2000
Forage fish	0.393	0.393	0.393	0.393
Eulachon	0.079	0.009	0.079	0.079
J. herring	0.009	0.009	0.009	0.030
A. herring	0.112	0.100	0.103	0.023
J. turbot	0.001	0.001	0.001	0.001
A. turbot	0.009	0.001	0.059	0.059
J. halibut	<0.001	0.020	0.020	0.020
J. Pacific cod	0.001	0.010	0.001	0.001
A. Pacific cod	0.020	0.001	0.020	0.020
J. sablefish	0.005	0.001	0.005	0.005
J. lingcod	<0.001	<0.001	0.023	0.013
A. lingcod	0.005	0.001	0.005	0.001
S. benthic fish	0.175	0.082	0.122	0.122
Skates	0.001	0.003	0.010	0.010
Epifaunal inv.	0.064	0.053	0.024	0.074
Inf. cam. inv.	0.070	0.079	0.070	0.070
Detritus	0.056	0.237	0.056	0.079

Table A5.1.3 Diet composition (cont.)

# Group	Period				# Group	Period			
34 J. sablefish	1750	1900	1950	2000	38 Shallowwater benthic fish	1750	1900	1950	2000
Small squid	0.009	0.009	0.009	0.009	Small squid	<0.001	<0.001	<0.001	<0.001
Squid	0.009	0.009	0.009	0.009	Squid	<0.001	<0.001	<0.001	<0.001
Forage fish	0.025	<0.001	0.025	0.025	Forage fish	0.198	0.198	0.198	0.198
Eulachon	0.005	0.005	0.005	0.005	Eulachon	0.053	0.001	0.040	0.040
J. herring	0.020	0.020	0.020	-	A. lingcod	<0.001	<0.001	0.010	<0.001
A. herring	0.010	0.035	0.010	0.010	S. benthic fish	0.010	0.010	0.010	0.010
A. POP	0.001	0.001	0.001	0.001	Small crabs	0.333	0.283	0.333	0.333
J. plank. rockfish	0.001	0.001	0.001	0.001	Comm. shrimp	0.001	0.004	0.004	0.004
J. turbot	0.010	<0.001	0.010	0.010	Epifaunal inv.	0.037	0.037	0.037	0.037
S. benthic fish	0.049	-	-	-	Inf. carn. inv.	0.181	0.181	0.181	0.181
Comm. shrimp	0.001	0.001	0.050	0.050	Inf. det. inv.	0.096	0.096	0.096	0.096
Epifaunal inv.	0.020	0.079	0.020	0.020	Euphausiids	0.025	0.073	0.025	0.025
Carn. jellyfish	0.010	0.010	0.010	0.010	Copepods	0.048	0.048	0.048	0.048
Euphausiids	0.830	0.830	0.830	0.850	Detritus	0.017	0.068	0.017	0.027
35 A. sablefish	1750	1900	1950	2000	39 Skates	1750	1900	1950	2000
Small squid	0.025	0.025	0.025	0.025	Dogfish	0.030	0.001	0.001	0.001
Squid	0.025	0.025	0.025	0.025	Forage fish	0.238	0.238	0.238	0.238
J. pollock	0.010	0.010	0.010	0.010	Eulachon	0.048	0.048	0.048	0.048
Forage fish	0.313	0.313	0.313	0.313	A. halibut	0.060	0.060	0.060	0.060
Eulachon	0.067	0.067	0.067	0.067	S. benthic fish	0.019	0.019	0.019	0.019
A. herring	0.250	0.010	0.010	0.010	Large crabs	0.154	0.140	0.140	0.140
J. plank. rockfish	0.004	0.004	0.004	0.004	Small crabs	0.259	0.180	0.180	0.080
J. turbot	0.010	0.010	0.010	0.010	Comm. shrimp	0.001	0.100	0.100	0.100
J. flatfish	0.010	0.010	0.010	0.010	Epifaunal inv.	0.090	0.090	0.090	0.090
J. halibut	0.005	0.005	0.005	0.005	Inf. carn. inv.	0.050	0.050	0.050	0.150
J. Pacific cod	0.005	0.005	0.005	0.005	Inf. det. inv.	0.050	0.050	0.050	0.050
J. sablefish	0.001	0.001	0.030	0.030	Detritus	0.001	0.024	0.024	0.024
Small crabs	0.040	0.011	0.011	0.011	40 Large crabs	1750	1900	1950	2000
Comm. shrimp	0.105	0.015	0.105	0.105	J. flatfish	0.024	0.010	0.010	0.010
Epifaunal inv.	0.010	0.100	0.010	0.010	Small crabs	0.100	0.100	0.100	0.100
Carn. jellyfish	0.100	0.340	0.340	0.340	Comm. shrimp	0.001	0.015	0.015	0.015
Detritus	0.020	0.049	0.020	0.020	Epifaunal inv.	0.435	0.435	0.435	0.395
36 J. lingcod	1750	1900	1950	2000	Inf. carn. inv.	0.065	0.065	0.065	0.105
Forage fish	0.227	0.227	0.227	0.227	Inf. det. inv.	0.075	0.075	0.075	0.075
Eulachon	0.063	0.063	0.063	0.063	Detritus	0.300	0.300	0.300	0.300
J. herring	0.170	0.120	0.170	0.170	41 Small crabs	1750	1900	1950	2000
J. flatfish	0.070	0.120	0.070	0.070	Epifaunal inv.	0.800	0.800	0.800	0.600
J. halibut	0.160	0.080	0.160	0.160	Inf. carn. inv.	0.100	0.100	0.100	0.300
J. Pacific cod	0.100	0.100	0.100	0.100	Inf. det. inv.	0.100	0.100	0.100	0.100
Small crabs	0.150	0.150	0.150	0.150	42 Comm. shrimp	1750	1900	1950	2000
Comm. shrimp	0.050	0.050	0.050	0.050	Inf. det. inv.	0.100	0.100	0.100	0.100
Detritus	0.010	0.090	0.010	0.010	Euphausiids	0.300	0.300	0.300	0.300
37 A. lingcod	1750	1900	1950	2000	Copepods	0.200	0.200	0.200	0.200
J. pollock	0.020	0.020	0.020	0.020	Detritus	0.400	0.400	0.400	0.400
Forage fish	0.117	0.143	0.117	0.117	43 Epifaunal inv.	1750	1900	1950	2000
Eulachon	0.063	0.063	0.063	0.063	Inf. det. inv.	0.010	0.005	0.200	0.060
J. herring	0.050	0.050	0.050	0.099	Macrophytes	0.041	0.010	0.001	0.001
A. herring	0.309	0.069	0.069	0.020	Detritus	0.949	0.985	0.799	0.939
J. POP	0.021	<0.001	0.021	0.021	44 Inf. carn. inv.	1750	1900	1950	2000
Inshore rockfish	0.010	0.010	0.010	0.010	Inf. det. inv.	0.100	0.075	0.250	0.060
J. pisc. rockfish	0.001	<0.001	0.001	0.001	Detritus	0.900	0.925	0.750	0.940
J. plank. rockfish	0.020	<0.001	0.020	0.020	46 Carn. jellyfish	1750	1900	1950	2000
J. turbot	0.020	0.001	0.020	0.020	J. herring	0.011	0.001	0.001	0.001
J. flatfish	0.011	0.032	0.011	0.111	Carn. jellyfish	0.010	0.060	0.060	0.060
J. halibut	0.001	0.050	0.050	0.050	Euphausiids	0.050	0.010	0.010	0.010
A. halibut	0.005	0.150	0.150	0.050	Copepods	0.080	0.080	0.080	0.080
J. Pacific cod	0.109	0.050	0.011	0.011	Detritus	0.849	0.849	0.849	0.849
A. Pacific cod	0.070	0.020	0.070	0.070					
J. lingcod	<0.001	0.001	0.050	0.050					
A. lingcod	<0.001	0.010	0.010	0.010					
Comm. shrimp	0.001	0.060	0.100	0.100					
Epifaunal inv.	0.115	0.141	0.100	0.100					
Detritus	0.057	0.130	0.057	0.057					

Table A5.1.4 Landings data for all time periods (t·km⁻²)

1750 landings									
# Group	Fleet								
	Sea otters	Halibut H+L	Salmon seine	Eulachon	Herring	Whaling	Cod, etc.	Invertebrates	Total
1 Sea otters	<0.001								<0.001
2 Mysticetae						<0.001			<0.001
4 Seals, sea lions						<0.001			<0.001
6 Transient salmon			0.046						0.046
7 Coho salmon			0.023						0.023
8 Chinook salmon			0.023						0.023
16 Eulachon				0.043					0.043
18 A. herring					0.002				0.002
30 J. halibut		0.010							0.010
31 A. halibut		0.010							0.010
33 A. Pacific cod							0.001		0.001
37 A. lingcod							<0.001		<0.001
43 Epifaunal inv.								<0.001	<0.001
45 Inf. det. inv.								<0.001	<0.001
Sum	<0.001	0.01944	0.09149	0.04286	0.00162	<0.001	0.0015	0.001	0.15871

1900 landings									
# Group	Fleet								
	Herring	Hook + Line	Salmon seine	Eulachon	Whaling	Longline	FN - Halibut	FN Inverts.	Total
2 Mysticetae					0.061				0.061
3 Odontocetae					0.002				0.002
4 Seals, sea lions					0.005				0.005
6 Transient salmon			0.126						0.126
7 Coho salmon			0.012						0.012
8 Chinook salmon			0.020						0.020
12 Dogfish						0.040			0.040
16 Eulachon				0.043					0.043
18 A. herring	<0.001								0.001
31 A. halibut		0.014					0.005		0.018
33 A. Pacific cod							0.001		0.001
37 A. lingcod		0.005							0.005
43 Epifaunal inv.								<0.001	<0.001
45 Inf. det. inv.								<0.001	<0.001
Sum	0.001	0.019	0.186	0.043	0.069	0.040	0.006	<0.001	0.336

Table A5.1.4 Landings data for all time periods (cont.)

1950 landings		Fleet																		
# Group	GF trawl	Sablefish	Herr. gillnet	GF H+L	Salm. gillnet	Crab trap	Prawn trap	Other Inv.	Halibut H+L	Salm. troll	Salm. seine	Salm. freezer	Herr. seine	Shrimp trawl	Eulachon	Longline	Seine nets	Rec.	Sea lion cull	Total
2 Mysticetæ																			0.069	0.069
4 Seals, sea lions																			0.004	0.004
6 Transient salmon					0.271					0.007	0.312	0.007						<0.001		0.598
7 Coho salmon					0.046					0.033	0.027	0.033						0.001		0.140
8 Chinook salmon					0.014					0.016	0.003	0.016						0.005		0.054
12 Dogfish																0.032				0.032
14 Pollock	0.007																			0.007
15 Forage fish																	0.006			0.006
16 Eulachon															<0.001					<0.001
18 A. herring			<0.001										0.461				0.461			0.922
20 A. POP	0.002			<0.001												<0.001				0.003
21 Inshore rockfish				0.002					0.002									<0.001		0.004
23 A. pisc. Rockfish	0.010			0.001														<0.001		0.011
25 A. plank. rockfish	0.036																	<0.001		0.036
27 A. turbot	0.004																			0.004
29 A. flatfish	0.073																			0.073
30 J. halibut									<0.001											<0.001
31 A. tulibut	0.071								<0.001											0.072
33 A. Pacific cod	0.019															<0.001	0.033			0.052
35 A. sablefish	0.002	0.004																		0.006
36 J. lingcod																		0.003		0.003
37 A. lingcod	0.004			0.001														<0.001		0.006
39 Skates	<0.001															<0.001				<0.001
40 Large crabs	<0.001					0.005												<0.001		0.005
42 Comm. shrimp							0.001								<0.001			<0.001		0.002
43 Epifaunal inv.								0.029										<0.001		0.029
Sum	0.227	0.004	<0.001	0.004	0.332	0.005	0.001		0.003	0.056	0.342	0.056	0.461	<0.001	<0.001	0.033	0.500	0.012	0.073	2.138
2000 landings		Fleet																		
# Group	GF trawl	Sablefish	Herr. gillnet	GF H+L	Salm. gillnet	Crab trap	Prawn trap	Other Inv.	Halibut H+L	Salm. troll	Salm. seine	Salm. freezer	Herr. seine	Shrimp trawl	Eulachon	Longline	Rec.	Total		
6 Transient salmon					0.150					0.003	0.067	0.010					<0.001	0.230		
7 Coho salmon					0.045					<0.001	<0.001	0.001					0.002	0.050		
8 Chinook salmon					0.036					<0.001	<0.001	<0.001					0.009	0.046		
11 Ratfish	<0.001																	<0.001		
12 Dogfish	<0.001															0.030		0.031		
14 Pollock	0.007																	0.007		
16 Eulachon																		0.003		
18 A. herring			0.155										0.087		0.003			0.241		
20 A. POP	0.060																	0.060		
21 Inshore rockfish	<0.001			0.003					0.004								0.003	0.010		
23 A. pisc. Rockfish	0.023			0.002													0.002	0.027		
25 A. plank. rockfish	0.077																	0.077		
27 A. turbot	0.067																	0.067		
29 A. flatfish	0.058																	0.058		
30 J. halibut	<0.001								0.028								0.001	0.029		
31 A. halibut	<0.001								0.031								0.016	0.046		
32 J. Pacific cod																		<0.001		
33 A. Pacific cod	0.051								0.003									0.054		
35 A. sablefish	<0.001	0.038							0.002									0.041		
36 J. lingcod																	0.002	0.002		
37 A. lingcod	0.019																0.001	0.020		
39 Skates	0.029																	0.029		
40 Large crabs						0.026												0.001	0.027	
42 Comm. shrimp							0.004								0.033			<0.001	0.037	
43 Epifaunal inv.	<0.001							0.078										<0.001	0.078	
Sum	0.391	0.038	0.155	0.005	0.231	0.026	0.004	0.078	0.067	0.004	0.067	0.012	0.087	0.033	0.003	0.030	0.038	1.267		

Table A5.1.5 Discard data for 1950 and 2000 (t·km⁻²)

1950 discards					
# Group	Fleet				Total
	GF trawl	Salm. gillnet	Salm. seine	Shrimp trawl	
4 Seals, sea lions			<0.001		<0.001
11 Ratfish	0.010				0.010
12 Dogfish	0.009	0.010			0.019
14 Pollock	0.002			0.002	0.004
20 A. POP	0.002				0.002
25 A. plank. rockfish	0.005	0.001			0.006
31 A. halibut	0.003				0.003
33 A. Pacific cod	0.002				0.002
35 A. sablefish	0.003				0.003
37 A. lingcod	0.001				0.001
39 Skates	<0.001				<0.001
43 Epifaunal inv.	0.002				0.002
46 Carn. jellyfish		<0.001			<0.001
Sum	0.039	0.011	<0.001	0.002	0.052

2000 discards					
# Group	Fleet				Total
	GF trawl	Salm. gillnet	Halibut H+L	Shrimp trawl	
4 Seals, sea lions		<0.001			<0.001
7 Coho salmon	<0.001				<0.001
8 Chinook salmon	<0.001				<0.001
10 Squid	<0.001				<0.001
11 Ratfish	0.010			0.001	0.011
12 Dogfish	0.009	0.008		<0.001	0.018
14 Pollock	0.002			<0.001	0.003
15 Forage fish	<0.001				<0.001
16 Eulachon	<0.001			0.001	0.001
18 A. herring	<0.001				<0.001
20 A. POP	0.002				0.002
21 Inshore rockfish	<0.001				<0.001
23 A. pisc. Rockfish	<0.001				<0.001
25 A. plank. rockfish	0.005	0.001			0.006
27 A. turbot	0.026		0.002	<0.001	0.028
29 A. flatfish				0.002	0.002
30 J. halibut	0.003				0.003
31 A. halibut	0.003				0.003
33 A. Pacific cod	0.002				0.002
35 A. sablefish	0.003				0.003
37 A. lingcod	0.001				0.001
38 S. benthic fish	<0.001			0.001	0.001
39 Skates	0.007			<0.001	0.007
40 Large crabs	<0.001				<0.001
41 Small crabs	<0.001				<0.001
42 Comm. shrimp	<0.001				<0.001
43 Epifaunal inv.	0.002				0.002
45 Inf. det. inv.	<0.001				<0.001
46 Carn. jellyfish	<0.001	<0.001			<0.001
49 Corals and sponges	0.001			0.001	0.002
Sum	0.078	0.010	0.002	0.009	0.098

Table A5.1.6 Market prices (\$·kg⁻¹) for 2000 BC fleet**Groundfish trawl**

Ratfish	3.32
Dogfish	0.55
Pollock	0.49
A. POP	1.28
Inshore rockfish	0.95
A. pisc. rockfish	0.95
A. plank. rockfish	0.95
A. turbot	1.28
A. flatfish	1.16
A. Pacific cod	1.07
A. lingcod	0.95
Skates	0.8
Epifaunal inv.	2.26

Sablefish

A. sablefish	6.17
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Herring gillnet

A. herring	0.46
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Halibut hook & line

Inshore rockfish	13
A. pisc. rockfish	13

Salmon gillnet

Transient salmon	2.62
Coho salmon	1.52
Chinook salmon	3.92

Crab traps

Large crabs	7.21
-------------	------

Shrimp & prawn traps

Commercial shrimp	9.05
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Other invertebrates

Epifaunal inv.	2.26
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Halibut hook & line

Inshore rockfish	13
J. halibut	6
A. halibut	6
A. Pacific cod	1.07
A. sablefish	6.17

Salmon troll

Transient salmon	8
Coho salmon	2
Chinook salmon	4.5

Salmon seine

Transient salmon	2.62
Coho salmon	1.52

Salmon troll freezer

Transient salmon	6
Coho salmon	1.75
Chinook salmon	4

Herring seine

A. herring	0.46
------------	------

Shrimp trawl

Comm. shrimp	4.88
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Eulachon

Eulachon	3.33
----------	------

Recreational

Transient salmon	2.62
Coho salmon	1.52
Chinook salmon	3.92
Inshore rockfish	1.28
A. pisc. rockfish	1.28
J. halibut	0
A. halibut	4.06
J. lingcod	1.69
A. lingcod	1.69
Large crabs	7.21
Comm. shrimp	4.88
Epifaunal inv.	2.26

Table A5.2.2 Feeding parameters for 1950

	Max. rel. feeding time	Feeding time adjust rate [0,1]	Fraction of 'other' mortality sens. to changes in feeding time	Predator effect on feeding time [0,1]	Density-dependent catchability: Q_{\max}/Q_0	$Q_{b\max}/Q_{B0}$
Sea otters	2	0.5	0	0	1	1000
Mysticetae	1	0.2	1	0	1	1000
Odontocetae	2	0.5	1	0	1	1000
Seals, sea lions	2	1	1	0	1	1000
Seabirds	2	0.5	1	0	1	1000
Transient salmon	2	0.5	1	0	1	1000
Coho salmon	2	0.5	1	0	1	1000
Chinook salmon	2	0.5	1	0	1	1000
Small squid	2	0.5	1	0	1	1000
Squid	2	0.5	1	0	1	1000
Ratfish	2	0.5	1	0	1	1000
Dogfish	1.4	0.5	1	0	1	1000
J. pollock	2	0.5	1	0	1	1000
Pollock	2	0.5	1	0	1	1000
Forage fish	2	0.5	1	0	1	1000
Eulachon	2	0.5	1	0	1	1000
J. herring	2	0.9	0	0	1	1000
A. herring	2	0.5	1	0	1	1000
J. POP	2	0.5	1	0	1	1000
A. POP	2	0.5	1	0	1	1000
Inshore rockfish	2	0.5	1	0	1	1000
J. pisc. rockfish	2	0.5	1	0	1	1000
A. pisc. rockfish	2	0.5	1	0	1	1000
J. plank. rockfish	2	0.5	1	0	1	1000
A. plank. rockfish	2	0.5	1	0	1	1000
J. turbot	2	0.5	1	0	1	1000
A. turbot	2	0.5	1	0	1	1000
J. flatfish	2	0.3	1	0	1	1000
A. flatfish	2	0.1	1	0	1	1000
J. halibut	2	0.1	1	0	1	1000
A. halibut	2	0.1	1	0	1	1000
J. Pacific cod	2	0.6	1	0	1	1000
A. Pacific cod	1.1	0.1	1	0	1	1000
J. sablefish	2	0.05	1	0	1	1000
A. sablefish	2	0.05	1	0	1	1000
J. lingcod	2	0	1	0	1	1000
A. lingcod	2	0.5	1	0	1	1000
S. benthic fish	1	0.5	1	0	1	1000
Skates	2	0.5	1	0	1	1000
Large crabs	2	0	1	0	1	1000
Small crabs	2	0	1	0	1	1000
Comm. shrimp	2	0	1	0	1	1000
Epifaunal inv.	2	0	1	0	1	1000
Inf. carn. inv.	2	0	1	0	1	1000
Inf. det. inv.	2	0	1	0	1	1000
Carn. jellyfish	2	0	1	0	1	1000
Euphausiids	2	0	1	0	1	1000
Copepods	2	0	1	0	1	1000
Corals and sponges	2	0	1	0	1	1000

Table A5.2.3 Trophic flow parameters for 1950

Prey	Predator																
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17
1 Sea otters			1														
2 Mysticetae																	
3 Odontocetae																	
4 Seals, sea lions			1														
5 Seabirds																	
6 Transient salmon			1	1	1							1					
7 Coho salmon			1	10 ²¹								1					
8 Chinook salmon			1	10 ²¹								1					
9 Small squid	10 ¹⁴		1	10 ⁹	365		1			10 ⁶		1					
10 Squid	10 ¹⁴		1	10 ¹²	1		1			10 ⁶		1			10 ¹⁸		
11 Ratfish			1									1					
12 Dogfish				10 ²¹													
13 J. pollock	10 ⁸												483	32068			
14 Pollock				1													
15 Forage fish		719	1	10 ¹⁴	15300		1	1	10 ⁹	10 ⁵	10 ²¹	1	1	50.9			
16 Eulachon			1		1		1	1	38158	10 ⁵	10 ²¹	1	1	14260			
17 J. herring			1	1	813				29521			1					
18 A. herring		8902	1	1	1		1	1				1					
19 J. POP			1									1					
20 A. POP				1								1					
21 Inshore rockfish				1													
22 J. pisc. Rockfish				418													
23 A. pisc. Rockfish				1													
24 J. plank. rockfish			1	47								1					
25 A. plank. rockfish			1	10 ¹²								1					
26 J. turbot			1									1					
27 A. turbot				10 ²¹								1					
28 J. flatfish												1					
29 A. flatfish				1								1					
30 J. halibut				1													
31 A. halibut			1	1													
32 J. Pacific cod							1										
33 A. Pacific cod				10 ²¹								1					
34 J. sablefish				1305								1					
35 A. sablefish				3.18								1					
36 J. lingcod																	
37 A. lingcod				10 ¹⁵													
38 S. benthic fish	10 ¹¹			30								1					
39 Skates																	
40 Large crabs	10 ⁶		1									1					
41 Small crabs	10 ⁶				85.2							1					
42 Comm. shrimp																	
43 Epifaunal inv.	1	3737		1	1266						1	1	20429	10 ¹⁴	1	1873	
44 Inf. carn. inv.		28457		1							1	1	1.03				
45 Inf. det. inv.		31930									1	1					
46 Carn. jellyfish					1	1			501	10 ⁶		1			1	2.7	
47 Euphausiids		1249	1		10 ⁵	1	1	1	10 ⁵	6857	1	1	1	1	1	1	1.91
48 Copepods		9053			277	1			1	10 ⁵		1	1	1	1	1	2.98
49 Corals and sponges																	
50 Macrophytes																	
51 Phytoplankton																	
52 Discards			1		3354												
53 Detritus									1	10 ⁷		1			1		

Table A5.2.3 Trophic flow parameters for 1950 (cont.)

		Predator																	
Prey		18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	34	
1	Sea otters																		
2	Mysticetae																		
3	Odontocetae																		
4	Seals, sea lions																		
5	Seabirds																		
6	Transient salmon																		
7	Coho salmon																		
8	Chinook salmon																		
9	Small squid			1			1		10 ²¹	10 ¹²	10 ²¹			1	1			1	
10	Squid			1			10 ²¹		10 ²¹	10 ²¹	10 ²¹			1	1			1	
11	Ratfish																		
12	Dogfish																		
13	J. pollock									10 ¹³	1.01								
14	Pollock									10 ¹²	10 ²¹								
15	Forage fish				3292		10 ²¹		10 ²¹	10 ¹⁴	10 ²¹	1	1	1	1		1	1	
16	Eulachon				10 ²¹		10 ²¹		10 ²¹	10 ¹²	10 ²¹	249	1	1	1		1	1	
17	J. herring				1				10 ²¹	10 ¹²	10 ²¹			1			1	1	
18	A. herring				1					10 ¹²	10 ²¹				1		1	1	
19	J. POP						1				10 ²¹			1					
20	A. POP										10 ²¹				1			1	
21	Inshore rockfish									16.2									
22	J. pisc. Rockfish						1092			10 ¹²									
23	A. pisc. Rockfish																		
24	J. plank. rockfish									10 ¹²					1			1	
25	A. plank. rockfish														1				
26	J. turbot										10 ²¹			1			1	1	
27	A. turbot														1		1		
28	J. flatfish								10 ¹²	10 ²¹	10 ²¹			1			1		
29	A. flatfish										10 ²¹				1				
30	J. halibut														1		10 ²¹		
31	A. halibut																		
32	J. Pacific cod													1			1		
33	A. Pacific cod									10 ²¹					10 ²¹		1		
34	J. sablefish																1		
35	A. sablefish																		
36	J. lingcod										2		1				1		
37	A. lingcod														1		1		
38	S. benthic fish				1						10 ²¹			1			1		
39	Skates						10 ²¹								1		1		
40	Large crabs				1.01						10 ²¹			1	1				
41	Small crabs				6545	10 ⁵	10 ²¹			10 ¹²		1.1	1	1	1	1			
42	Comm. shrimp				438	965				10 ¹²	10 ²¹			1				46.8	
43	Epifaunal inv.				1	10 ⁵	10 ²¹			10 ¹²	10 ²¹	1	10 ¹⁰	22686	1	76.3	1	1	
44	Inf. carn. inv.				1							10 ²¹	10 ¹⁸			1	1		
45	Inf. det. inv.						10 ²¹					457	1			1			
46	Carn. jellyfish						10 ²¹		10 ²¹									1	
47	Euphausiids	1.29	1	1	1.08	10 ²¹	10 ²¹	10 ⁷	10 ²¹	10 ¹²	10 ²¹		1			1		1	
48	Copepods	1	1	1		10 ²¹		1	10 ²¹							1			
49	Corals and sponges																		
50	Macrophytes																		
51	Phytoplankton																		
52	Discards																		
53	Detritus					10 ²¹				>10 ⁵	>10 ⁵			1	1	1	1		

Table A5.2.3 Trophic flow parameters for 1950 (cont.)

Prey	Predator															
	35	36	37	38	39	40	41	42	43	44	45	46	47	48	49	50
1 Sea otters																
2 Mysticetae																
3 Odontocetae																
4 Seals, sea lions																
5 Seabirds																
6 Transient salmon																
7 Coho salmon																
8 Chinook salmon																
9 Small squid	10 ²¹			23.3												
10 Squid	1			1												
11 Ratfish																
12 Dogfish					1											
13 J. pollock	10 ²¹		1													
14 Pollock																
15 Forage fish	10 ²¹	1	1	1	1											
16 Eulachon	10 ²¹	1	1	10 ²⁰	1											
17 J. herring		1	1									1131				
18 A. herring	10 ²¹		1													
19 J. POP			1													
20 A. POP																
21 Inshore rockfish			1													
22 J. pisc. Rockfish			1													
23 A. pisc. Rockfish																
24 J. plank. rockfish	10 ²¹		1													
25 A. plank. rockfish																
26 J. turbot	10 ²¹		1													
27 A. turbot																
28 J. flatfish	10 ²¹	1	1			1.02										
29 A. flatfish																
30 J. halibut	10 ²¹	1	1.01													
31 A. halibut			10 ²¹		1											
32 J. Pacific cod	10 ²¹	1	1													
33 A. Pacific cod			1													
34 J. sablefish	10 ²¹															
35 A. sablefish																
36 J. lingcod			1													
37 A. lingcod			1	10 ²¹												
38 S. benthic fish				1	25.3											
39 Skates																
40 Large crabs						1										
41 Small crabs	10 ²¹	1		1	1	1										
42 Comm. shrimp	10 ²¹	1	1.01	10 ²¹	1	10 ¹⁰										
43 Epifaunal inv.	10 ²¹		1	1	1	1	1									
44 Inf. carn. inv.				10 ²¹	1	1	10 ²¹									
45 Inf. det. inv.				10 ²¹	1	10 ¹⁹	1	10 ¹²	10 ⁵	9.26						
46 Carn. jellyfish	10 ²¹											10 ⁵				
47 Euphausiids				10043				1				1				
48 Copepods				22489				1				264	1			
49 Corals and sponges																
50 Macrophytes									853							2
51 Phytoplankton													18	1		
52 Discards																
53 Detritus	1.28	1	1	10 ⁹	1	1		1	919	10 ²¹	1	2.36			10 ²¹	

Appendix 5.3 Time Series

Table A5.3.1 Biomass time series data (t·km⁻²): 1900-1950

Absolute estimates (A); Relative estimates (R). Relative estimates are scaled to start/end points from 1950 and 2000 models. Dark cell colour indicates high data quality. Data quality key (bottom) refers to all tables in Appendix 5.3. References are listed in Appendix 9.5.3.

Group #	Sea otters	Sea birds	Trans.salm.	Coho	Chinook	Dogfish	Forage fish	Insh. rock.	Pisc. rock	Flatfish	Halibut	Lingcod	Skates	Crabs	Epi. inverts
Abs./Rel.	1	5	6	7	8	12	15	21	23	29	31	37	39	40	43
	R	R	A	A	A	R	R	R	R	R	A	A	R	R	R
1925			1.145	0.190	0.136										
1926			1.396	0.175	0.105										
1927			0.733	0.174	0.094										
1928			1.235	0.197	0.081										
1929			2.190	0.196	0.071										
1930			1.059	0.197	0.099										
1931			0.603	0.117	0.069										
1932			0.568	0.137	0.082										
1933			0.621	0.147	0.071	0.372		0.095	0.536				0.313	0.507	11.584
1934		6.6E-03	0.732	0.171	0.086	0.372		0.095	0.536				0.313	0.507	11.584
1935		6.6E-03	0.757	0.233	0.083	0.372		0.095	0.536		0.212		0.313	0.507	11.584
1936		6.6E-03	0.904	0.198	0.086	0.372		0.095	0.536		0.207		0.313	0.507	11.584
1937		6.6E-03	0.777	0.144	0.079	0.387		0.095	0.536		0.202		0.313	0.507	11.556
1938	4.6E-05	6.6E-03	0.756	0.215	0.069	0.387		0.095	0.536		0.200		0.313	0.507	11.556
1939	5.0E-05	6.6E-03	0.633	0.181	0.069	0.417		0.095	0.536		0.202	0.106	0.307	0.507	11.555
1940	5.0E-05	6.6E-03	0.612	0.196	0.052	0.417	7.535	0.095	0.536		0.210	0.106	0.307	0.507	11.555
1941	5.0E-05	6.6E-03	0.791	0.269	0.084	0.417	7.535	0.095	0.536		0.224	0.106	0.307	0.507	11.555
1942	5.0E-05	6.6E-03	0.726	0.177	0.063	0.417	7.535	0.095	0.536		0.246	0.106	0.307	0.507	11.555
1943	5.0E-05	6.6E-03	0.526	0.152	0.050	0.417	7.535	0.095	0.536		0.278	0.106	0.307	0.507	11.555
1944	5.0E-05	6.6E-03	0.424	0.171	0.058	0.417	7.535	0.095	0.536	0.707	0.320	0.106	0.307	0.507	11.555
1945	5.0E-05	6.6E-03	0.707	0.244	0.065	0.417	7.561	0.095	0.536	0.675	0.366	0.106	0.303	0.507	11.549
1946	4.9E-05	6.6E-03	0.590	0.168	0.082	0.417	7.561	0.095	0.536	0.396	0.404	0.106	0.303	0.507	11.549
1947	4.9E-05	6.6E-03	0.750	0.143	0.066	0.417	7.557	0.095	0.536	0.617	0.426	0.106	0.303	0.507	11.546
1948	4.9E-05	6.6E-03	0.611	0.163	0.077	0.417	7.557	0.095	0.536	0.426	0.433	0.106	0.303	0.507	11.544
1949	4.9E-05	6.6E-03	0.636	0.154	0.083	0.404	7.570	0.095	0.536	0.450	0.431	0.105	0.302	0.507	11.545
1950	5.0E-05	6.6E-03	0.645	0.140	0.077	0.417	7.600	0.095	0.541	0.535	0.429	0.104	0.300	0.506	11.584

References

2 2 21,32 21,32,37 21,32 2 2 2 2 22 40 2 2 2 2

Data quality key

	Detailed information for correct area
	Detailed information from similar or wider area
	Local Ecological Knowledge
	Inferred from other data (fishing mortality = catch / biomass)
	No data available. Guessimate.

Table A5.3.2 Biomass time series data (t·km⁻²): 1950-2000

Values given in t·km⁻². Absolute estimates (A); Relative estimates (R). Relative estimates are scaled to start/end points from 1950 and 2000 models. References are listed in Appendix 9.5.3.

Group	Sea otters	Odont	Seals & SL	Sea birds	Trans.salm.	Coho	Chinook	Dogfish	Forage fish	Eulachon	Herring	J. POP	Ad. POP	Insh. rock	Pisc. rock	Turbot ¹	Flatfish	Halibut	Pac. cod	Sablefish	Lingcod	Skates	Crabs	Shrimp	Epi. inverts	Inf. inverts	Euphasid	Phytoplank.
Abs./Rel	R	R	A	R	A	A	A	R	R	R	A	R	R	R	R	R	R	A	A	A	A	R	R	R	R	R	R	R
1950	5.0E-05			6.6E-03	0.545	0.140	0.077	0.417	7.600	1.893	1.001				0.095	0.541	0.535	0.429	0.348		0.104	0.300	0.506			11.584	34.305	
1951	5.1E-05			6.6E-03	0.540	0.092	0.085	0.430	7.630	1.868	1.001				0.095	0.546	0.762	0.433	0.348		0.104	0.296	0.505			11.623	34.305	
1952	5.1E-05			6.6E-03	0.620	0.096	0.092	0.453	7.659	1.879	0.985				0.095	0.551	0.865	0.450	0.348		0.104	0.295	0.504			11.660	34.305	
1953	5.2E-05			6.6E-03	0.481	0.068	0.100	0.475	7.689	1.872	0.796				0.096	0.557	0.520	0.478	0.348		0.104	0.291	0.503			11.696	34.305	
1954	5.3E-05			6.7E-03	0.384	0.065	0.088	0.498	7.719	1.867	0.705				0.096	0.560	0.673	0.502	0.348		0.104	0.288	0.501			11.735	34.305	
1955	5.4E-05			6.7E-03	0.342	0.102	0.079	0.521	7.748	1.863	0.675				0.096	0.564	0.536	0.509	0.348		0.104	0.285	0.500			11.774	34.305	
1956	5.4E-05	86	0.108	6.7E-03	0.484	0.092	0.084	0.543	7.778	1.858	0.825				0.096	0.567	0.436	0.505	0.348		0.103	0.281	0.499			11.813	34.305	
1957	5.5E-05	86		6.7E-03	0.352	0.099	0.079	0.568	7.806	1.854	0.340				0.096	0.571	0.330	0.493	0.370		0.103	0.278	0.498			11.852	34.305	
1958	5.6E-05	90		6.7E-03	0.476	0.061	0.090	0.589	7.837	1.849	0.844				0.096	0.574	0.431	0.482	0.450		0.103	0.275	0.497			11.891	34.305	
1959	5.7E-05	90		6.7E-03	0.304	0.074	0.083	0.612	7.867	1.845	0.796				0.096	0.578	0.299	0.479	0.540		0.103	0.271	0.495			11.930	34.305	
1960	5.8E-05	92		6.7E-03	0.356	0.113	0.071	0.642	7.896	1.841	1.058				0.096	0.581	0.412	0.480	0.594		0.102	0.268	0.494			11.969	34.305	
1961	5.9E-05	94	0.052	6.7E-03	0.519	0.121	0.066	0.672	7.926	1.836	0.843				0.097	0.585	0.076	0.467	0.482		0.102	0.265	0.494			12.009	34.305	
1962	6.0E-05	97		6.8E-03	1.113	0.117	0.071	0.702	7.956	1.832	1.051				0.097	0.588	0.079	0.430	0.479		0.101	0.261	0.493			12.048	34.305	
1963	6.1E-05	98		6.8E-03	0.595	0.141	0.072	0.733	7.985	1.827	0.869	0.254	1.002		0.097	0.592	0.090	0.446	0.453		0.100	0.256	0.492			12.087	34.305	
1964	6.2E-05	91		6.8E-03	0.659	0.172	0.068	0.763	8.015	1.823	0.722	0.192	1.045		0.097	0.595	0.060	0.350	0.432		0.099	0.255	0.491			12.126	34.305	
1965	6.3E-05	94		6.8E-03	0.324	0.170	0.085	0.781	8.037	1.818	0.343	0.155	1.069		0.097	0.598	0.060	0.382	0.390	0.708	0.097	0.253	0.489			12.165	34.305	
1966	6.4E-05	94		6.8E-03	0.633	0.106	0.102	0.799	8.059	1.814	0.182	0.122	1.009		0.097	0.602	0.060	0.473	0.344		0.096	0.252	0.488			12.204	34.305	
1967	6.5E-05	76		6.9E-03	0.375	0.147	0.100	0.817	8.060	1.810	0.160	0.097	0.771		0.097	0.605	0.060	0.431	0.303		0.095	0.250	0.487			12.243	34.305	
1968	6.6E-05	71		6.9E-03	0.947	0.090	0.103	0.835	8.106	1.804	0.179	0.108	0.648		0.097	0.609	0.079	0.340	0.272		0.090	0.249	0.486			12.281	34.305	
1969	6.7E-05	73		6.9E-03	0.251	0.138	0.094	0.854	8.132	1.799	0.114	0.124	0.556		0.097	0.612	0.079	0.269	0.252		0.090	0.247	0.485			12.319	34.305	
1970	6.8E-05	72	0.046	6.9E-03	0.501	0.144	0.068	0.870	8.159	1.794	0.528	0.123	0.517		0.097	0.616	0.079	0.374	0.242		0.089	0.246	0.484			12.358	34.305	
1971	7.0E-05	72	0.049	6.9E-03	0.206	0.111	0.109	0.881	8.182	1.789	0.725	0.116	0.483		0.097	0.618	0.060	0.317	0.235		0.088	0.245	0.483			12.396	34.305	
1972	7.1E-05	73	0.050	7.0E-03	0.516	0.117	0.106	0.912	8.205	1.783	0.529	0.113	0.485		0.098	0.620	0.064	0.382	0.229		0.088	0.246	0.482			12.435	34.305	
1973	7.2E-05	79	0.052	7.0E-03	0.385	0.110	0.100	0.936	8.214	1.778	0.832	0.103	0.457		0.098	0.622	0.060	0.479	0.221		0.089	0.249	0.481			12.474	34.305	
1974	7.4E-05	81	0.058	7.0E-03	0.407	0.087	0.098	0.963	8.223	1.772	1.129	0.089	0.447		0.098	0.624	0.101	0.512	0.221		0.093	0.251	0.480			12.513	34.305	
1975	7.5E-05	80	0.063	7.0E-03	0.161	0.087	0.095	0.965	8.236	1.766	1.300	0.076	0.388		0.098	0.626	0.118	0.430	0.234		0.090	0.254	0.479			12.552	34.305	
1976	7.6E-05	84	0.067	7.0E-03	0.353	0.102	0.095	0.977	8.249	1.760	1.043	0.075	0.362		0.098	0.629	0.132	0.351	0.235		0.091	0.256	0.478			12.591	34.305	
1977	7.7E-05	82	0.072	7.0E-03	0.385	0.095	0.094	0.990	8.262	1.754	0.923	0.073	0.355		0.098	0.631	0.146	0.293	0.229		0.091	0.259	0.477			12.630	34.305	
1978	7.8E-05	73	0.077	7.1E-03	0.449	0.106	0.098	0.995	8.278	1.749	0.813	0.094	0.364		0.098	0.633	0.161	0.327	0.258		0.093	0.262	0.476			12.669	34.305	
1979	7.9E-05	77	0.082	7.1E-03	0.457	0.093	0.087	1.000	8.293	1.745	0.472	0.093	0.373		0.098	0.635	0.172	0.318	0.268		0.093	0.264	0.476			12.708	34.305	
1980	8.0E-05	75	0.087	7.1E-03	0.361	0.069	0.083	1.006	8.309	1.741	1.536	0.043	0.378		0.098	0.638	0.183	0.311	0.307		0.093	0.267	0.475			12.747	34.305	
1981	8.1E-05	76	0.093	7.1E-03	0.450	0.069	0.083	1.006	8.317	1.737	1.620	0.033	0.368		0.098	0.639	0.196	0.300	0.329		0.092	0.270	0.474			12.786	34.305	
1982	8.2E-05	87	0.095	7.1E-03	0.323	0.100	0.082	1.006	8.326	1.732	1.472	0.033	0.362		0.098	0.640	0.213	0.298	0.351		0.091	0.272	0.473			12.825	34.305	
1983	8.3E-05	85	0.104	7.1E-03	0.592	0.110	0.074	1.006	8.335	1.728	1.250	0.061	0.372		0.098	0.641	0.225	0.365	0.367		0.091	0.275	0.472			12.864	34.305	
1984	8.4E-05	85	0.115	7.1E-03	0.359	0.093	0.087	1.006	8.343	1.724	1.596	0.101	0.388		0.099	0.642	0.236	0.347	0.409		0.091	0.278	0.471			12.903	34.305	
1985	8.5E-05	86	0.125	7.2E-03	0.647	0.126	0.090	1.006	8.352	1.720	1.065	0.101	0.410		0.099	0.643	0.252	0.352	0.420		0.090	0.281	0.470			12.942	34.305	
1986	8.6E-05	86	0.138	7.2E-03	0.813	0.091	0.075	1.006	8.360	1.716	0.907	0.095	0.443		0.099	0.644	0.272	0.401	0.452		0.089	0.283	0.470			12.981	34.305	
1987	8.7E-05	91	0.158	7.2E-03	0.609	0.079	0.085	1.006	8.369	1.713	0.764	0.108	0.473		0.099	0.646	0.295	0.401	0.466		0.089	0.286	0.469			13.020	34.305	
1988	8.8E-05	32	0.180	7.2E-03	0.952	0.092	0.093	1.006	8.377	1.709	1.059	0.108	0.495		0.099	0.647	0.310	0.450	0.512		0.089	0.289	0.468			13.059	34.305	
1989	8.9E-05	87	0.192	7.2E-03	0.449	0.106	0.084	1.000	8.386	1.705	1.213	0.106	0.523		0.099	0.647	0.325	0.363	0.515		0.089	0.292	0.467			13.098	34.305	
1990	9.0E-05	95	0.217	7.2E-03	0.698	0.095	0.079	0.994	8.394	1.701	0.938	0.096	0.548		0.099	0.648	0.341	0.300	0.556		0.089	0.295	0.466			13.137	34.305	
1991	9.1E-05	97	0.230	7.2E-03	0.691	0.082	0.078	0.989	8.403	1.697	0.793	0.106	0.572		0.099	0.649	0.352	0.309	0.584		0.089	0.298	0.465			13.176	34.305	
1992	9.2E-05	96	0.216	7.3E-03	0.636	0.055	0.090	0.983	8.411	1.693	1.008	0.100	0.598		0.099	0.649	0.364	0.301	0.617		0.089	0.301	0.464			13.215	34.305	
1993	9.3E-05	92	0.258	7.3E-03	0.403	0.061	0.081	0.977	8.420	1.689	0.921	0.096	0.619		0.099	0.650	0.383	0.316	0.668		0.089	0.304	0.463			13.254	34.305	
1994	9.4E-05	90	0.300	7.3E-03	0.246	0.045	0.057	0.972	8.428	1.685	0.670	0.074	0.633		0.100	0.651	0.399	0.308	0.729		0.089	0.307	0.462			13.293	34.305	
1995	9.5E-05	85	0.283	7.3E-03	0.626	0.037	0.058	0.966	8.437	1.681	0.474	0.064	0.623		0.100	0.651	0.408	0.303	0.790		0.089	0.310	0.461			13.3		

Table A5.3.3 Catch time series data (t·km⁻²): 1900-1950

Values given in t·km⁻². Composite functional groups (Y/N) combine catch records for multiple species. References listed in Appendix 9.5.3.

Group #	Mysticetae 2	Seals & SL 4	Trans.salm. 6	Coho 7	Chinook 8	Dogfish 12	Herring 18	Flatfish 29	Halibut 31	Sablefish 35	Lingcod 37
Composite?	Y	Y	Y	N	N	N	N	Y	N	N	N
1900	0.061	0.005	0.126	0.012	0.018	0.039	0.001	0	0.018	0	4.94E-03
1901	0.061	0.005	0.126	0.012	0.018	0.038	0.000	0	0.024	0	3.52E-05
1902	0.061	0.005	0.126	0.012	0.018	0.046	0.001	0	0.034	0	3.16E-05
1903	0.061	0.005	0.126	0.012	0.018	0.062	0.001	0	0.040	0	3.46E-05
1904	0.061	0.005	0.126	0.012	0.018	0.053	0.001	0	0.050	0	3.93E-05
1905	0.061	0.005	0.126	0.012	0.018	0.054	0.001	0	0.033	0	4.67E-05
1906	0.061	0.005	0.190	0.025	0.025	0.055	0.001	0	0.043	0	4.29E-05
1907	0.061	0.005	0.190	0.025	0.025	0.051	0.001	0	0.053	0	4.56E-05
1908	0.061	0.005	0.190	0.025	0.025	0.060	0.009	0	0.065	0	4.78E-05
1909	0.061	0.005	0.254	0.038	0.032	0.072	0.002	0	0.081	0	4.04E-05
1910	0.063	0.005	0.254	0.038	0.032	0.030	0.008	0	0.082	0	3.19E-05
1911	0.381	0.005	0.254	0.038	0.032	0.025	0.017	0	0.073	0	3.80E-05
1912	0.288	0.018	0.317	0.051	0.039	0.028	0.108	0	0.085	0	8.63E-05
1913	0.125	0.006	0.317	0.051	0.039	0.018	0.040	0	0.083	0.018	8.99E-05
1914	0.130	0.007	0.317	0.051	0.039	0.000	0.044	0	0.080	0.028	9.33E-05
1915	0.061	0.036	0.381	0.065	0.047	0.004	0.019	0	0.073	0.022	1.37E-04
1916	0.154	0.008	0.381	0.065	0.047	0.015	0.020	0	0.046	0.038	1.39E-04
1917	0.037	0.008	0.381	0.065	0.047	0.009	0.019	0	0.042	0.053	1.30E-04
1918	0.121	0.008	0.445	0.078	0.054	0.038	0.016	0	0.070	0.018	1.40E-04
1919	0.109	0.008	0.445	0.078	0.054	0.033	0.015	0	0.079	0.006	3.05E-04
1920	0.128	0.008	0.445	0.078	0.054	0.013	0.019	0	0.089	0.016	3.26E-04
1921	0.093	0.008	0.509	0.091	0.061	0.034	0.027	0	0.122	0.012	2.35E-04
1922	0.057	0.002	0.509	0.091	0.061	0.026	0.026	0	0.109	0.011	2.01E-04
1923	0.109	0.017	0.509	0.091	0.061	0.042	0.038	0	0.125	0.010	1.94E-04
1924	0.081	0.024	0.573	0.104	0.068	0.066	0.029	0	0.124	0.011	2.01E-04
1925	0.080	0.025	0.573	0.104	0.068	0.084	0.028	0	0.119	0.009	2.84E-04
1926	0.109	0.017	0.698	0.096	0.052	0.065	0.027	0	0.118	0.006	2.22E-04
1927	0.119	0.015	0.440	0.095	0.056	0.093	0.057	0	0.101	0.010	1.86E-04
1928	0.110	0.010	0.741	0.108	0.049	0.174	0.089	0	0.113	0.008	3.71E-04
1929	0.137	0.012	1.314	0.107	0.043	0.208	0.082	0	0.114	0.009	5.00E-04
1930	0.054	0.009	0.742	0.108	0.070	0.077	0.103	0	0.095	0.010	2.57E-04
1931	0.032	0.012	0.422	0.064	0.049	0.103	0.112	0	0.068	0.004	1.43E-05
1932	0.032	0.010	0.397	0.075	0.057	0.023	0.039	0	0.063	0.004	4.29E-05
1933	0.010	0.008	0.435	0.081	0.050	0.072	0.035	0	0.064	0.004	5.71E-05
1934	0.061	0.007	0.512	0.094	0.060	0.117	0.070	0	0.066	0.004	1.43E-05
1935	0.025	0.006	0.530	0.127	0.058	0.073	0.058	0	0.064	0.006	1.29E-04
1936	0.044	0.034	0.633	0.108	0.060	0.102	0.393	0	0.063	0.004	1.00E-04
1937	0.038	0.023	0.544	0.078	0.055	0.090	0.552	0	0.070	0.008	1.00E-04
1938	0.041	0.029	0.529	0.118	0.048	0.143	0.854	0	0.072	0.005	1.14E-04
1939	0.057	0.011	0.443	0.099	0.048	0.122	1.120	0	0.085	0.005	1.57E-04
1940	0.073	0.001	0.428	0.107	0.037	0.125	0.293	0	0.089	0.008	2.00E-04
1941	0.071	0.001	0.554	0.147	0.059	0.199	0.443	0	0.048	0.011	6.71E-04
1942	0.020	0.002	0.508	0.097	0.044	0.243	0.064	0	0.041	0.007	1.09E-03
1943	0.016	0.000	0.368	0.083	0.035	0.293	0.495	0	0.047	0.013	2.26E-03
1944	0.020	0.001	0.296	0.094	0.041	0.445	0.527	0.007	0.049	0.013	3.86E-03
1945	0.024	0.003	0.495	0.134	0.046	0.334	0.189	0.034	0.056	0.013	6.21E-03
1946	0.028	0.003	0.413	0.092	0.057	0.163	0.170	0.041	0.067	0.014	1.03E-02
1947	0.032	0.002	0.525	0.078	0.046	0.215	0.875	0.052	0.090	0.008	9.17E-03
1948	0.036	0.001	0.427	0.089	0.054	0.174	1.043	0.091	0.070	0.013	8.01E-03
1949	0.057	0.003	0.445	0.084	0.058	0.228	1.060	0.055	0.067	0.017	6.86E-03
1950	0.069	0.019	0.598	0.083	0.054	0.032	0.922	0.074	0.071	0.006	5.70E-03

References

• Unshaded entries (guesses) are based on trend or average of adjacent values.

Table A5.3.4 Catch time series data (t·km⁻²): 1950-2000

Values given in t·km². Composite functional groups (Y/N) combine catch records for multiple species. References listed in Appendix 9.5.3.

Group	Mysticetae	Seals & SL	Trans.salm.	Coho	Chinook	Squid	Dogfish	Herring	Ad. POP	Turbot	Flatfish	Halibut	Pac. cod	Sablefish	Lingcod	Skates	Crabs	Shrimp	Epi. inverts
#	2	4	6	7	8	10	12	18	20	27	29	31	33	35	37	39	40	42	43
Composite?	Y	Y	Y	N	N	Y	N	N	N	N	Y	N	N	N	N	N	Y	Y	Y
1950	0.069	0.019	0.598	0.083	0.054		0.032	0.922	0.003	0.004	0.074	0.071	0.052	0.006	0.006	2.4E-03			
1951	0.106	0.002	0.569	0.138	0.060		0.057	1.337	0.003	0.004	0.048	0.065	0.052	0.008	0.005	2.4E-03			
1952	0.114	0.002	0.438	0.091	0.064		0.044	0.805	0.003	0.004	0.071	0.076	0.052	0.005	0.003	2.3E-03			
1953	0.123	0.003	0.565	0.094	0.070		0.045	0.573	0.003	0.004	0.034	0.081	0.052	0.004	0.001	1.2E-03			
1954	0.127	0.002	0.551	0.085	0.062		0.036	1.008	0.018	0.006	0.035	0.082	0.052	0.005	0.005	2.3E-03			
1955	0.129	0.002	0.369	0.094	0.055		0.037	0.888	0.003	0.014	0.054	0.064	0.052	0.005	0.007	2.3E-03			
1956	0.090	0.003	0.299	0.100	0.059		0.016	1.841	0.012	0.022	0.057	0.076	0.055	0.004	0.010	2.3E-03			
1957	0.146	0.005	0.377	0.091	0.055		0.035	1.295	0.008	0.006	0.048	0.073	0.046	0.006	0.011	2.4E-03			
1958	0.198	0.010	0.531	0.097	0.063		0.023	0.577	0.027	0.004	0.052	0.093	0.162	0.004	0.009	2.3E-03			
1959	0.199	0.030	0.295	0.080	0.058		0.091	0.768	0.027	0.008	0.039	0.100	0.108	0.006	0.011	2.4E-03			
1960	0.201	0.018	0.243	0.072	0.057		0.062	0.543	0.025	0.012	0.058	0.110	0.132	0.008	0.012	4.6E-03			
1961	0.203	0.007	0.382	0.112	0.053		0.085	1.149	0.016	0.023	0.043	0.096	0.091	0.008	0.013	4.3E-03			
1962	0.205	0.012	0.523	0.119	0.057		0.006	1.106	0.026	0.016	0.042	0.112	0.095	0.007	0.017	4.3E-03			
1963	0.131	0.009	0.369	0.115	0.057		0.003	1.808	0.053	0.009	0.043	0.121	0.164	0.009	0.013	4.1E-03			
1964	0.139	0.009	0.349	0.140	0.071		0.014	1.691	0.050	0.011	0.041	0.108	0.292	0.009	0.014	4.2E-03			
1965	0.130	0.004	0.210	0.170	0.068		0.004	1.602	0.114	0.010	0.044	0.107	0.509	0.013	0.018	4.4E-03			
1966	0.128	0.002	0.461	0.167	0.081		0.015	0.865	0.318	0.010	0.077	0.104	0.532	0.019	0.027	4.2E-03			
1967	0.103	0.001	0.419	0.104	0.080		0.022	0.401	0.249	0.019	0.071	0.085	0.287	0.031	0.030	4.1E-03			
1968	0	0.000	0.530	0.146	0.082		0.017	0.034	0.183	0.012	0.070	0.095	0.290	0.050	0.039	4.1E-03			
1969	0	0	0.241	0.088	0.075		0.011	0.000	0.141	0.022	0.055	0.110	0.189	0.054	0.022	5.2E-03			
1970	0	0	0.481	0.137	0.079		0.007	0.022	0.111	0.005	0.063	0.096	0.076	0.035	0.020	4.1E-03			
1971	0	0	0.384	0.143	0.098		0.006	0.115	0.061	0.002	0.058	0.082	0.086	0.056	0.017	3.6E-03			
1972	0	0	0.527	0.109	0.096		0.008	0.283	0.094	0.003	0.035	0.072	0.153	0.039	0.015	3.9E-03			
1973	0	0	0.602	0.116	0.090		0.005	0.291	0.086	0.008	0.046	0.047	0.225	0.046	0.012	3.9E-03			
1974	0	0	0.430	0.109	0.088		0.005	0.301	0.131	0.006	0.044	0.024	0.268	0.059	0.017	3.7E-03			
1975	0	0	0.221	0.085	0.083		0.005	0.297	0.078	0.011	0.060	0.037	0.282	0.067	0.013	6.8E-03			
1976	0	0	0.373	0.096	0.086		0.004	0.462	0.043	0.017	0.071	0.039	0.280	0.047	0.013	7.0E-03			
1977	0	0	0.434	0.100	0.084		0.003	0.533	0.028	0.017	0.050	0.029	0.197	0.039	0.009	8.6E-03			
1978	0	0	0.476	0.094	0.088		0.015	0.510	0.018	0.026	0.039	0.028	0.119	0.044	0.006	7.7E-03			
1979	0	0	0.402	0.105	0.078		0.023	0.204	0.041	0.025	0.059	0.022	0.263	0.044	0.009	8.2E-03			
1980	0	0	0.352	0.092	0.074		0.107	0.161	0.072	0.023	0.061	0.024	0.255	0.043	0.011	1.4E-02			
1981	0	0	0.551	0.088	0.057		0.033	0.212	0.065	0.015	0.056	0.018	0.180	0.045	0.016	1.2E-02			
1982	0	0	0.431	0.089	0.073		0.002	0.199	0.079	0.010	0.032	0.020	0.113	0.049	0.020	9.3E-03		0.006	
1983	0	0	0.505	0.099	0.060		0.001	0.230	0.070	0.009	0.036	0.022	0.154	0.044	0.024	1.0E-02		0.007	
1984	0	0	0.324	0.109	0.081	1.5E-04	0.001	0.259	0.086	0.010	0.038	0.034	0.100	0.047	0.016	9.7E-03		0.008	
1985	0	0	0.794	0.092	0.068	1.2E-04	0.003	0.294	0.079	0.013	0.034	0.062	0.065	0.048	0.019	8.8E-03	0.003	0.011	
1986	0	0	0.733	0.125	0.060	8.4E-04	0.004	0.254	0.067	0.011	0.035	0.065	0.122	0.048	0.030	9.8E-03	0.004	0.012	
1987	0	0	0.496	0.090	0.086	9.1E-04	0.009	0.201	0.077	0.021	0.040	0.071	0.499	0.055	0.033	1.2E-02	0.005	0.029	
1988	0	0	0.661	0.078	0.074	9.4E-04	0.013	0.215	0.088	0.008	0.059	0.067	0.350	0.056	0.031	1.1E-02	0.008	0.029	
1989	0	0	0.660	0.091	0.066	7.4E-04	0.003	0.305	0.069	0.011	0.069	0.064	0.272	0.052	0.035	9.1E-03	0.007	0.028	
1990	0	0	0.690	0.104	0.064	7.7E-04	0.018	0.345	0.066	0.029	0.076	0.065	0.205	0.062	0.044	1.3E-02	0.016	0.024	
1991	0	0	0.614	0.098	0.059	1.2E-03	0.016	0.314	0.051	0.026	0.090	0.055	0.430	0.062	0.044	1.6E-02	0.009	0.037	
1992	0	0	0.497	0.082	0.069	9.8E-04	0.015	0.282	0.044	0.029	0.088	0.025	0.289	0.061	0.034	1.5E-02	0.032	0.034	
1993	0	0	0.666	0.054	0.066	1.4E-04	0.002	0.332	0.043	0.030	0.104	0.036	0.225	0.061	0.035	1.5E-02	0.096	0.040	
1994	0	0	0.498	0.080	0.047	1.9E-03	0.002	0.279	0.060	0.033	0.079	0.034	0.092	0.052	0.034	1.8E-02	0.085	0.040	
1995	0	0	0.327	0.044	0.037	8.1E-04	0.005	0.195	0.077	0.020	0.083	0.034	0.084	0.039	0.033	1.6E-02	0.055	0.076	
1996	0	0	0.244	0.037	0.013	9.0E-04	0.040	0.139	0.061	0.072	0.052	0.042	0.032	0.047	0.017	2.2E-02	0.066	0.083	0.087
1997	0	0	0.357	0.014	0.024	7.6E-05	0.023	0.168	0.057	0.047	0.045	0.051	0.079	0.049	0.011	2.8E-02	0.031	0.045	0.080
1998	0	0	0.230	0.004	0.017	9.0E-04	0.027	0.239	0.080	0.061	0.051	0.056	0.071	0.051	0.013	1.5E-02	0.022	0.046	0.089
1999	0	0	0.140	0.005	0.013	4.2E-04	0.034	0.227	0.057	0.057	0.056	0.053	0.043	0.045	0.017	2.5E-02	0.027	0.036	0.080
2000	0	0	0.145	0.005	0.010	4.2E-04	0.072	0.241	0.060	0.067	0.058	0.046	0.034	0.041	0.026	2.9E-02	0.027	0.037	0.078
2001	0	0	0.182	0.006	0.012		0.071	0.165	0.057	0.123	0.054	0.040	0.017	0.035	0.021	2.8E-02	0.051	0.039	0.072
2002	0	0	0.233	0.009	0.020		0.046	0.165	0.057	0.070	0.065	0.049	0.012	0.029	0.024	2.8E-02	0.037	0.017	0.074
2003	0	0	0.257	0.006	0.018		0.054	0.139	0.060	0.067	0.054	0.041	0.013	0.028	0.024	3.8E-02	0.063	0.011	0.077
2004	0	0	0.102	0.007	0.018		0.077	0.146	0.031	0.047	0.031	0.063	0.013	0.024	0.023	1.8E-02	0.019	0.004	0.030
Reference	24	7	1,16,19,20,21	1,16,19,20,21	1,16,19,20,21	14,15	18,19,30	12	1,15,43	1,3,16,19,36	1,16,19	1,5,17,28,29	3,19,45	1,19,26	1,19,31	1,6,16,19	13,19	19,42	19

Table A5.3.5 Fishing mortality time series data (yr⁻¹): 1900-1950

References listed in Appendix 9.5.3.

Group #	Mysticetae 2	Seals & SL 4	Trans.salm. 6	Coho 7	Chinook 8	Dogfish 12	Herring 18	Halibut 31	Sablefish 35	Lingcod 37
1900	0.040	0.072	0.150	0.150	0.2	0.095	0.000	0.030	0	0.048
1901	0.040	0.072	0.150	0.150	0.2	0.090	0.000	0.040	0	0.000
1902	0.040	0.072	0.150	0.150	0.2	0.109	0.000	0.058	0	0.000
1903	0.040	0.072	0.150	0.150	0.2	0.149	0.000	0.071	0	0.000
1904	0.040	0.072	0.150	0.150	0.2	0.127	0.000	0.090	0	0.000
1905	0.040	0.072	0.150	0.150	0.2	0.129	0.000	0.061	0	0.000
1906	0.040	0.072	0.200	0.200	0.2	0.132	0.000	0.080	0	0.000
1907	0.040	0.072	0.200	0.200	0.2	0.123	0.001	0.103	0	0.000
1908	0.040	0.072	0.200	0.200	0.2	0.144	0.004	0.129	0	0.000
1909	0.040	0.072	0.250	0.250	0.3	0.173	0.001	0.163	0	0.000
1910	0.041	0.072	0.250	0.250	0.3	0.071	0.003	0.168	0	0.000
1911	0.247	0.072	0.250	0.250	0.3	0.059	0.007	0.155	0	0.000
1912	0.187	0.142	0.300	0.300	0.3	0.066	0.044	0.204	0	0.001
1913	0.081	0.044	0.300	0.300	0.3	0.043	0.016	0.185	0.029	0.001
1914	0.084	0.053	0.300	0.300	0.3	0.000	0.018	0.182	0.047	0.001
1915	0.039	0.289	0.350	0.350	0.4	0.010	0.008	0.170	0.036	0.001
1916	0.100	0.067	0.350	0.350	0.4	0.035	0.008	0.110	0.063	0.001
1917	0.024	0.067	0.350	0.350	0.4	0.023	0.007	0.104	0.088	0.001
1918	0.078	0.067	0.400	0.400	0.4	0.092	0.006	0.176	0.030	0.001
1919	0.071	0.067	0.400	0.400	0.4	0.079	0.006	0.205	0.011	0.003
1920	0.083	0.067	0.400	0.400	0.4	0.032	0.007	0.240	0.026	0.003
1921	0.060	0.067	0.450	0.450	0.5	0.082	0.011	0.337	0.020	0.002
1922	0.037	0.015	0.450	0.450	0.5	0.062	0.011	0.313	0.019	0.002
1923	0.071	0.132	0.450	0.450	0.5	0.101	0.015	0.370	0.017	0.002
1924	0.053	0.190	0.500	0.500	0.5	0.159	0.012	0.379	0.018	0.002
1925	0.052	0.199	0.500	0.547	0.5	0.202	0.011	0.377	0.015	0.003
1926	0.071	0.137	0.500	0.547	0.5	0.155	0.011	0.387	0.010	0.002
1927	0.077	0.117	0.600	0.547	0.6	0.223	0.023	0.346	0.016	0.002
1928	0.072	0.080	0.600	0.547	0.6	0.417	0.036	0.402	0.013	0.004
1929	0.089	0.095	0.600	0.547	0.6	0.499	0.033	0.420	0.015	0.005
1930	0.035	0.075	0.700	0.547	0.7	0.184	0.041	0.368	0.017	0.002
1931	0.021	0.095	0.700	0.547	0.7	0.246	0.045	0.275	0.006	0.000
1932	0.021	0.072	0.700	0.547	0.7	0.055	0.016	0.267	0.006	0.000
1933	0.006	0.059	0.700	0.547	0.7	0.172	0.014	0.284	0.006	0.001
1934	0.040	0.050	0.700	0.547	0.7	0.282	0.028	0.308	0.006	0.000
1935	0.016	0.040	0.700	0.547	0.7	0.174	0.024	0.301	0.010	0.001
1936	0.029	0.248	0.700	0.547	0.7	0.244	0.158	0.303	0.007	0.001
1937	0.025	0.166	0.700	0.547	0.7	0.215	0.223	0.347	0.013	0.001
1938	0.027	0.208	0.700	0.547	0.7	0.343	0.344	0.362	0.008	0.001
1939	0.037	0.080	0.700	0.547	0.7	0.292	0.452	0.421	0.009	0.002
1940	0.048	0.009	0.700	0.547	0.7	0.301	0.118	0.426	0.014	0.002
1941	0.046	0.007	0.700	0.547	0.7	0.478	0.179	0.215	0.017	0.006
1942	0.013	0.013	0.700	0.547	0.7	0.583	0.026	0.167	0.012	0.010
1943	0.010	0.003	0.700	0.547	0.7	0.704	0.200	0.171	0.021	0.022
1944	0.013	0.006	0.700	0.547	0.7	1.067	0.213	0.154	0.022	0.037
1945	0.016	0.019	0.700	0.547	0.7	0.800	0.076	0.152	0.021	0.060
1946	0.018	0.019	0.700	0.547	0.7	0.391	0.068	0.165	0.024	0.099
1947	0.021	0.023	0.700	0.547	0.7	0.516	0.353	0.211	0.013	0.088
1948	0.028	0.009	0.700	0.547	0.7	0.417	0.421	0.162	0.022	0.077
1949	0.043	0.030	0.700	0.547	0.7	0.548	0.427	0.156	0.028	0.066
1950	0.053	0.173	0.927	0.596	0.7	0.076	0.921	0.164	0.018	0.055

References

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- All light grey entries are estimated based on catch and biomass, except lingcod.
- Unshaded entries (guesses) are based on trend or average of adjacent values.
- Groups not displayed use baseline fishing mortality (assumed constant).

Table A5.3.6 Fishing mortality time series data (yr⁻¹): 1950-2000

References listed in Appendix 9.5.3.

Group #	Mysticetacae 2	Seals & SL 4	Trans.salm. 6	Coho 7	Chinook 8	Dogfish 12	Herring 16	Ad. POP 20	Turbot 27	Halibut 31	Pac. cod 33	Sablefish 35	Lingcod 37	Skates 39	Crabs 40	Shrimp 42
1950	0.053	0.173	0.927	0.596	0.7	0.076	0.921	0.050	0.002	0.164	0.159	0.018	0.055	0.008	0.005	0.1
1951	0.082	0.019	1.053	1.495	0.7	0.137	1.336	0.050	0.007	0.149	0.159	0.018	0.044	0.008	0.005	0.1
1952	0.088	0.021	1.070	0.953	0.7	0.105	1.377	0.050	0.007	0.169	0.159	0.018	0.029	0.008	0.005	0.1
1953	0.095	0.025	1.253	1.092	0.7	0.107	0.720	0.050	0.007	0.169	0.159	0.018	0.007	0.004	0.005	0.1
1954	0.097	0.015	1.436	0.898	0.7	0.086	1.429	0.050	0.004	0.163	0.159	0.018	0.046	0.008	0.005	0.1
1955	0.099	0.014	1.080	0.919	0.7	0.090	1.013	0.050	0.009	0.125	0.159	0.018	0.069	0.008	0.005	0.1
1956	0.069	0.024	0.618	1.090	0.7	0.039	2.231	0.050	0.014	0.150	0.159	0.018	0.100	0.008	0.005	0.1
1957	0.112	0.043	1.058	0.922	0.7	0.085	3.806	0.050	0.004	0.148	0.124	0.018	0.104	0.008	0.005	0.1
1958	0.152	0.090	1.109	1.200	0.7	0.055	0.896	0.050	0.003	0.194	0.330	0.018	0.090	0.008	0.005	0.1
1959	0.153	0.572	0.971	1.076	0.7	0.219	0.965	0.050	0.005	0.208	0.198	0.018	0.105	0.008	0.005	0.1
1960	0.154	0.347	0.695	0.637	0.8	0.150	0.513	0.050	0.008	0.229	0.222	0.018	0.115	0.015	0.005	0.1
1961	0.156	0.138	0.735	0.931	0.8	0.203	1.362	0.050	0.015	0.198	0.162	0.018	0.125	0.014	0.005	0.1
1962	0.157	0.234	0.470	1.010	0.8	0.014	1.051	0.050	0.010	0.234	0.163	0.018	0.167	0.014	0.005	0.1
1963	0.100	0.175	0.631	0.817	0.8	0.008	2.081	0.053	0.006	0.261	0.239	0.018	0.131	0.014	0.005	0.1
1964	0.107	0.165	0.574	0.815	0.8	0.034	2.344	0.049	0.007	0.250	0.269	0.018	0.142	0.014	0.005	0.1
1965	0.100	0.081	0.648	1.002	0.8	0.009	4.672	0.108	0.006	0.274	0.379	0.018	0.184	0.015	0.005	0.1
1966	0.096	0.041	0.706	1.573	0.8	0.035	4.765	0.315	0.006	0.301	0.456	0.026	0.287	0.014	0.005	0.1
1967	0.079	0.013	1.117	0.707	0.8	0.053	2.514	0.323	0.012	0.280	0.305	0.038	0.321	0.014	0.005	0.1
1968	0	0.003	0.559	1.621	0.8	0.041	0.191	0.283	0.008	0.350	0.363	0.062	0.436	0.014	0.005	0.1
1969	0	0	0.927	0.637	0.8	0.026	0.000	0.253	0.014	0.434	0.287	0.067	0.246	0.017	0.005	0.1
1970	0	0	0.961	0.950	0.9	0.016	0.042	0.215	0.003	0.395	0.169	0.046	0.224	0.014	0.005	0.1
1971	0	0	1.445	1.292	0.9	0.015	0.158	0.127	0.002	0.348	0.198	0.074	0.193	0.012	0.005	0.1
1972	0	0	1.020	0.937	0.9	0.014	0.535	0.194	0.002	0.313	0.288	0.054	0.168	0.013	0.005	0.1
1973	0	0	1.648	1.049	0.9	0.013	0.350	0.188	0.005	0.213	0.289	0.065	0.140	0.013	0.005	0.1
1974	0	0	1.056	1.251	0.915	0.012	0.267	0.294	0.004	0.110	0.278	0.107	0.192	0.012	0.005	0.1
1975	0	0	1.374	0.879	0.932	0.011	0.228	0.201	0.007	0.157	0.286	0.115	0.142	0.023	0.005	0.1
1976	0	0	1.029	0.944	0.901	0.010	0.443	0.119	0.011	0.165	0.325	0.089	0.138	0.023	0.005	0.1
1977	0	0	1.129	0.897	0.897	0.008	0.578	0.078	0.011	0.122	0.277	0.080	0.101	0.029	0.005	0.1
1978	0	0	1.062	0.886	0.920	0.037	0.833	0.049	0.017	0.109	0.183	0.099	0.068	0.026	0.005	0.1
1979	0	0	0.880	1.129	0.828	0.056	0.433	0.110	0.017	0.077	0.383	0.103	0.093	0.027	0.005	0.1
1980	0	0	0.975	1.334	0.986	0.257	0.105	0.192	0.015	0.078	0.440	0.109	0.116	0.047	0.005	0.1
1981	0	0	1.223	0.763	1.313	0.079	0.131	0.177	0.010	0.056	0.324	0.114	0.174	0.039	0.005	0.1
1982	0	0	1.335	0.888	0.808	0.004	0.135	0.218	0.007	0.057	0.216	0.119	0.221	0.031	0.005	0.222
1983	0	0	1.005	0.901	0.873	0.002	0.184	0.187	0.006	0.056	0.287	0.103	0.262	0.034	0.005	0.431
1984	0	0	0.825	1.172	1.183	0.003	0.245	0.222	0.006	0.084	0.172	0.107	0.175	0.032	0.005	0.442
1985	0	0	1.228	0.733	1.262	0.006	0.276	0.191	0.009	0.148	0.117	0.107	0.212	0.029	0.007	0.314
1986	0	0	0.901	1.373	0.781	0.009	0.280	0.150	0.007	0.143	0.219	0.110	0.343	0.033	0.009	0.273
1987	0	0	0.814	1.141	1.026	0.021	0.264	0.162	0.014	0.145	0.514	0.128	0.409	0.040	0.010	0.327
1988	0	0	0.694	0.853	0.687	0.032	0.203	0.178	0.005	0.131	0.326	0.131	0.398	0.038	0.015	0.250
1989	0	0	1.471	0.871	0.950	0.006	0.251	0.132	0.007	0.124	0.303	0.132	0.469	0.030	0.014	0.274
1990	0	0	1.135	1.051	0.804	0.043	0.368	0.120	0.019	0.117	0.274	0.166	0.637	0.044	0.031	0.261
1991	0	0	0.928	1.189	0.748	0.038	0.424	0.089	0.017	0.095	0.538	0.174	0.703	0.053	0.018	0.285
1992	0	0	0.782	1.486	0.589	0.037	0.280	0.073	0.019	0.041	0.480	0.148	0.588	0.050	0.064	0.254
1993	0	0	1.652	0.673	0.798	0.005	0.361	0.069	0.020	0.054	0.513	0.103	0.660	0.048	0.192	0.246
1994	0	0	2.033	1.773	0.889	0.006	0.417	0.094	0.021	0.047	0.365	0.113	0.697	0.058	0.171	0.297
1995	0	0	0.522	1.187	0.324	0.011	0.410	0.124	0.013	0.043	0.469	0.110	0.774	0.053	0.109	0.528
1996	0	0	0.472	2.515	0.033	0.096	0.231	0.102	0.018	0.052	0.199	0.156	0.404	0.072	0.133	0.605
1997	0	0	1.152	2.831	0.296	0.054	0.239	0.095	0.018	0.064	0.410	0.138	0.317	0.093	0.063	0.457
1998	0	0	0.668	0.717	0.531	0.064	0.283	0.102	0.018	0.076	0.376	0.191	0.446	0.050	0.044	0.308
1999	0	0	0.727	1.805	0.195	0.082	0.286	0.099	0.018	0.078	0.243	0.115	0.434	0.084	0.054	0.239
2000	0	0	0.700	1.805	0.095	0.172	0.367	0.107	0.018	0.074	0.207	0.152	0.658	0.096	0.053	0.271
2001	0	0	0.700	1.805	0.123	0.170	0.256	0.100	0.018	0.067	0.110	0.093	0.685	0.094	0.102	0.122
2002	0	0	0.700	1.805	0.248	0.110	0.284	0.100	0.018	0.103	0.269	0.138	0.685	0.092	0.074	0.122
2003	0	0	0.700	1.805	0.248	0.129	0.146	0.100	0.018	0.093	0.269	0.138	0.685	0.125	0.125	0.122
2004	0	0	0.700	1.805	0.248	0.184	0.146	0.100	0.018	0.138	0.269	0.138	0.685	0.059	0.039	0.122

Reference

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33

42

- All light grey entries are estimated based on catch and biomass, except lingcod.
- Unshaded entries (guesses) are based on trend or average of adjacent values.
- Groups not displayed use baseline fishing mortality (assumed constant).

Appendix 5.4 Dynamic Fit to Data: 1950-2000

Figure A5.4.1 Biomass fit to data ($t \cdot km^{-2}$).

Predicted from model (line) versus observed trend (circles). Large circles at beginning and end of series show 1950 and 2000 model values. Error bars in year 2000 show confidence interval of data point based on data pedigree in 2000 model. Observed trend is from A) stock assessment; B.) local environmental knowledge; C.) model start/end points. The anomaly trends for phytoplankton production and herring recruitment that best reduce the sum of squares between predicted and observed time series are provided in section C (lower right); the forcing patterns are used in all simulations.

A.) Stock assessment

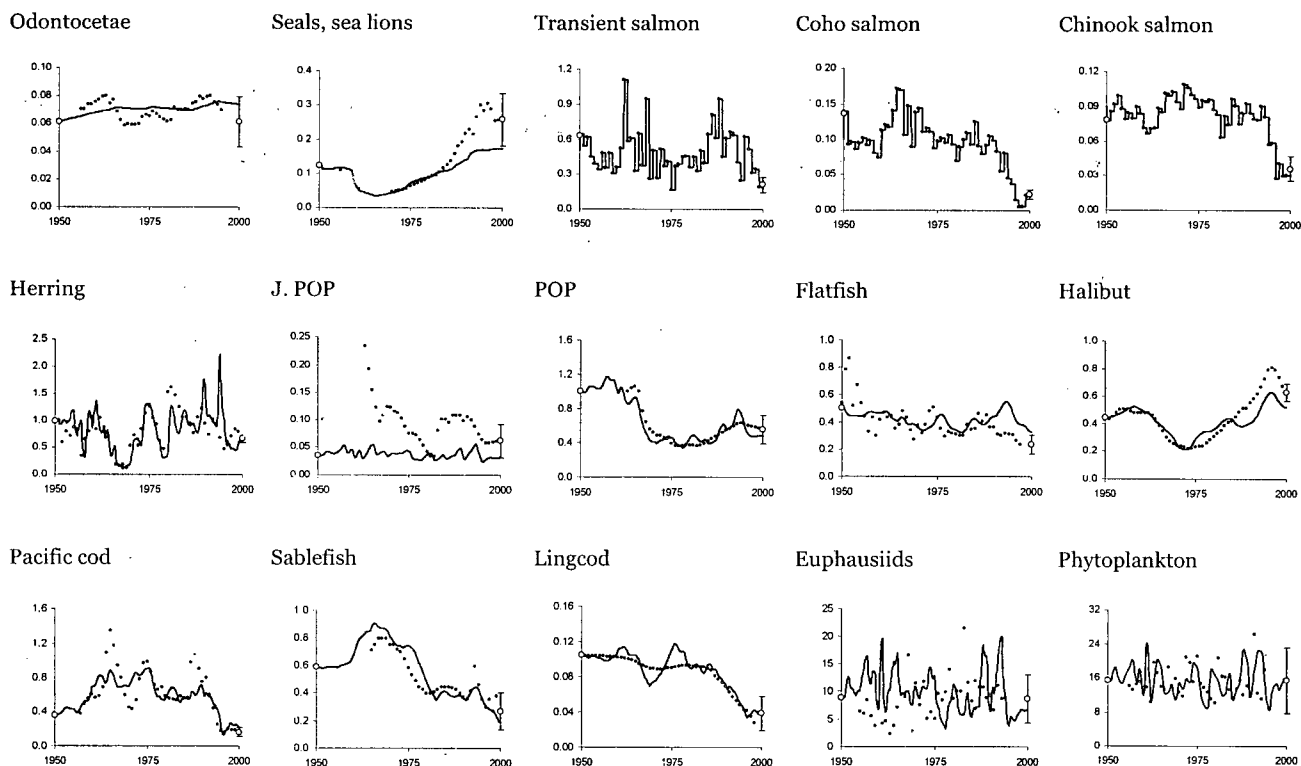


Figure A5.4.1 Biomass fit to data (cont.)

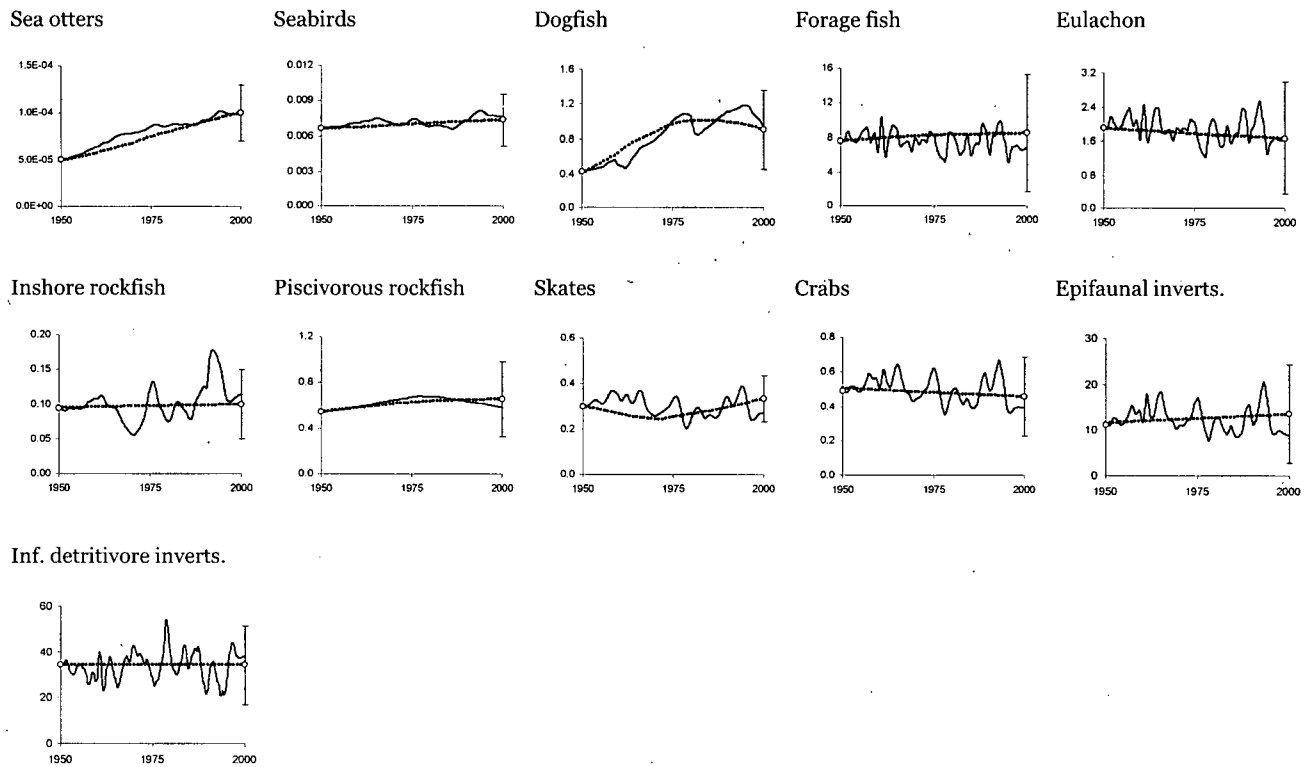
B.) Local Environmental Knowledge

Figure A5.4.1 Biomass fit to data (cont.)

C.) Model start/end points

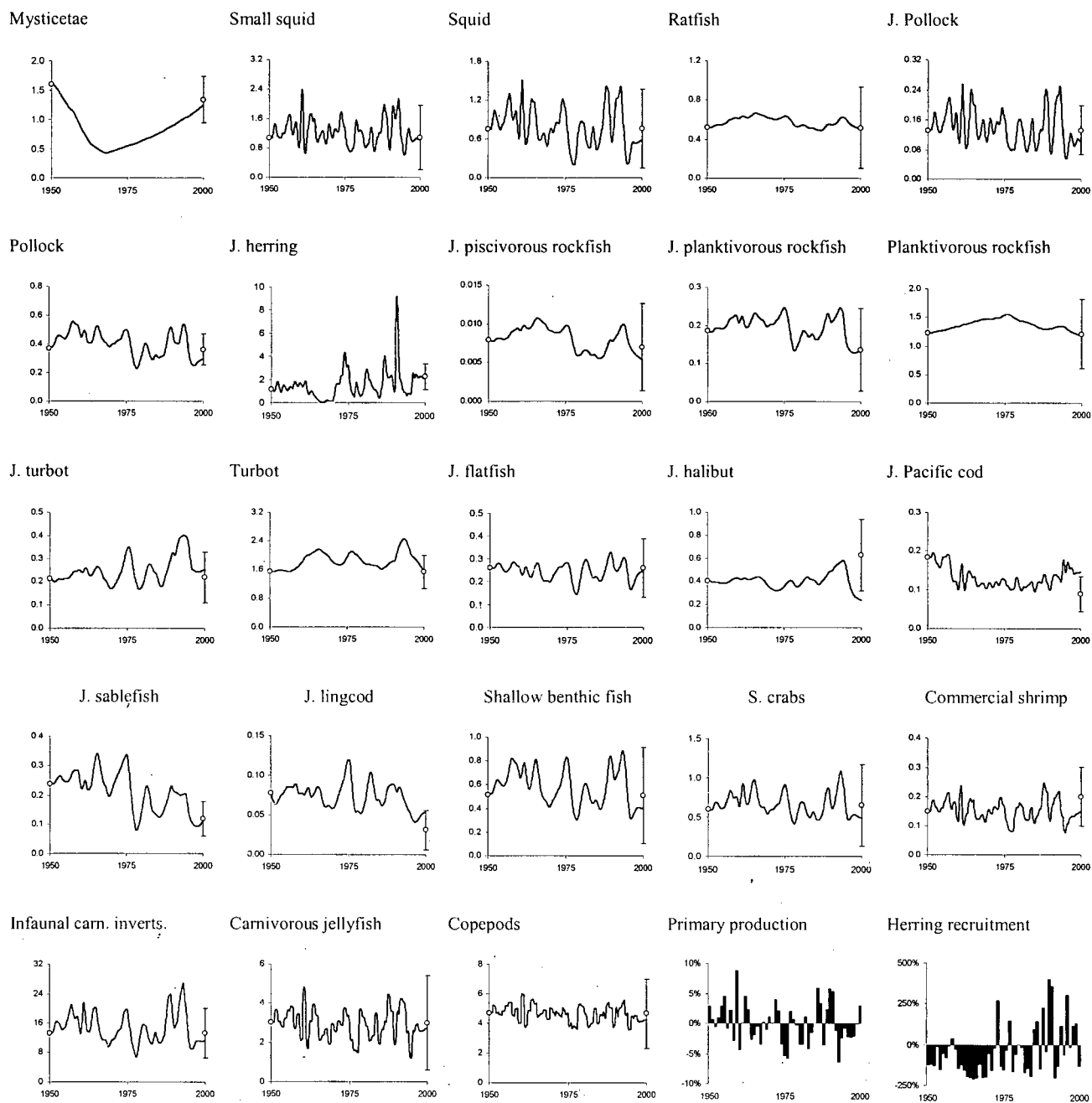
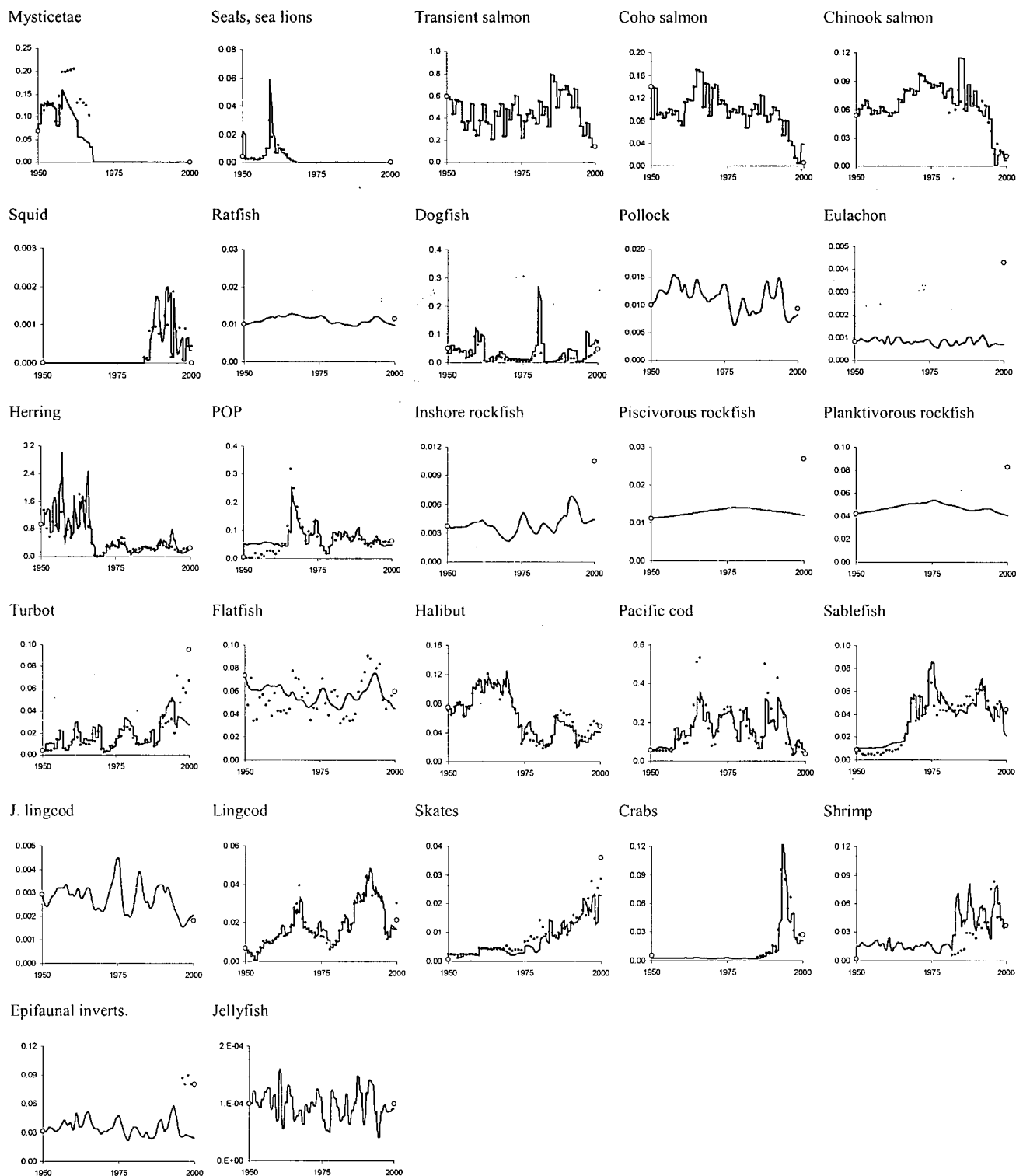


Figure A5.4.2 Catch fit to data ($t \cdot km^{-2}$)

Predicted from model (line) versus observed trend (circles). Large circles at beginning and end of series show 1950 and 2000 model values.

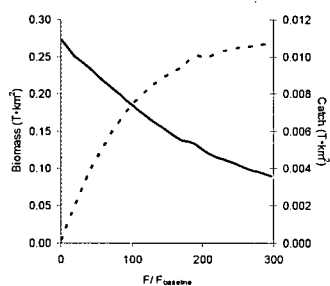


Appendix 5.5 Equilibrium Analysis of 2000 Model

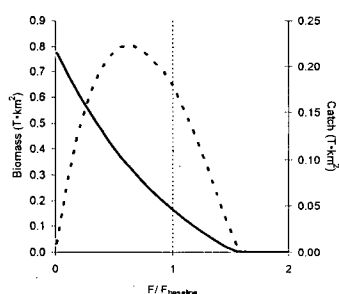
Figure A5.5.1 Equilibrium analysis of 2000 model

Solid curve shows biomass; broken curve shows catch. Vertical broken line shows baseline fishing mortality.

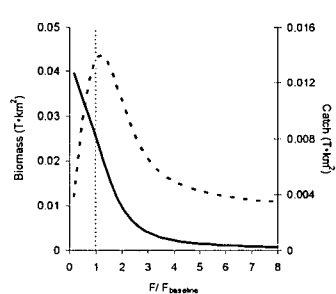
Seals, sea lions



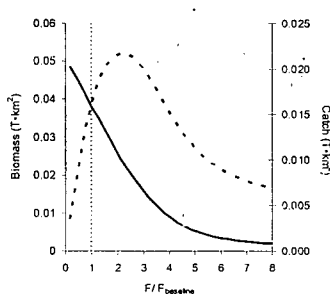
Transient salmon



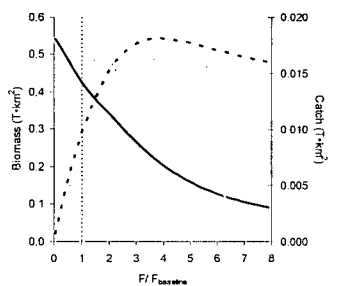
Coho salmon



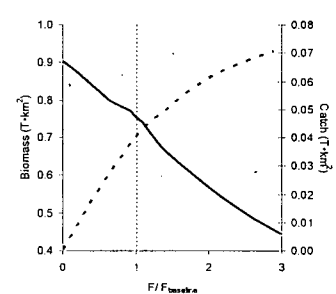
Chinook salmon



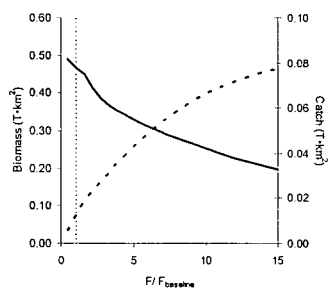
Ratfish



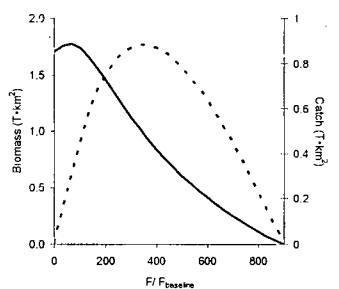
Dogfish



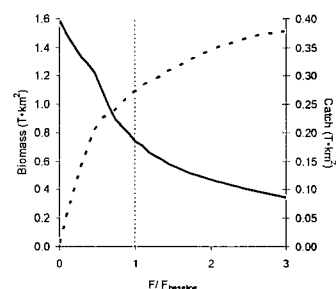
Pollock



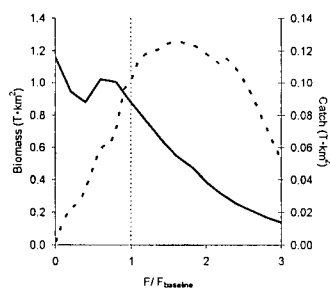
Eulachon



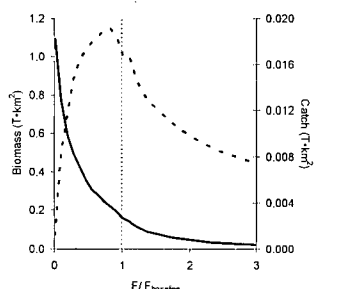
Herring



POP



Inshore rockfish



Piscivorous rockfish

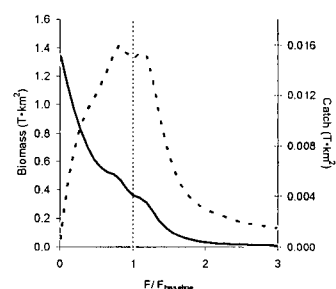
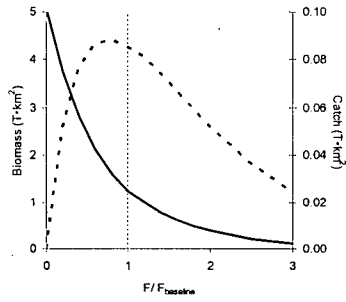
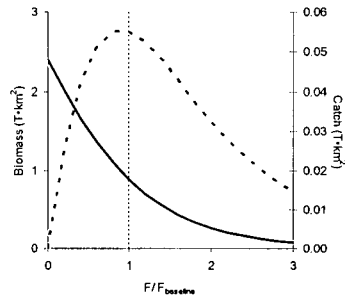


Figure A5.5.1 Equilibrium analysis of 2000 model (cont.)

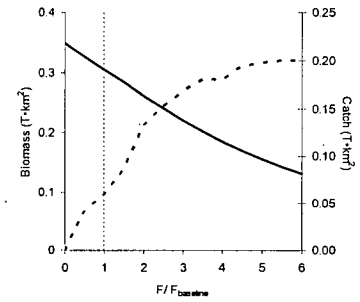
Planktivorous rockfish



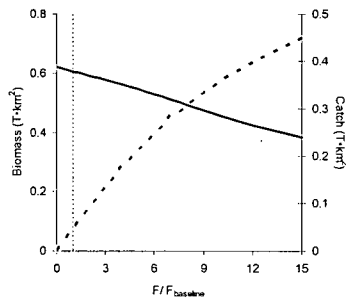
Turbot



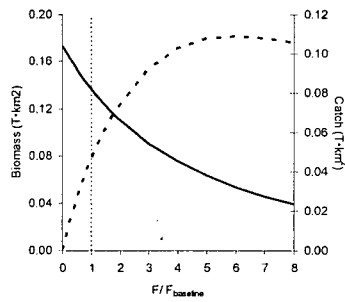
Flatfish



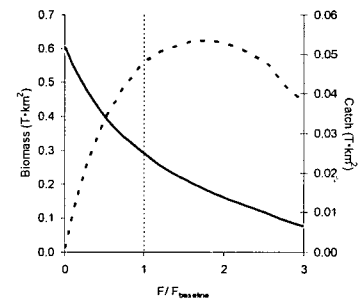
Halibut



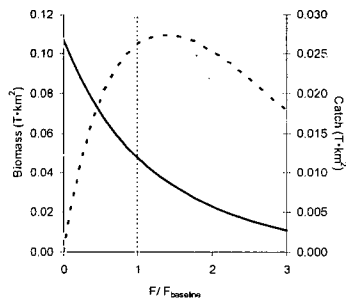
Pacific cod



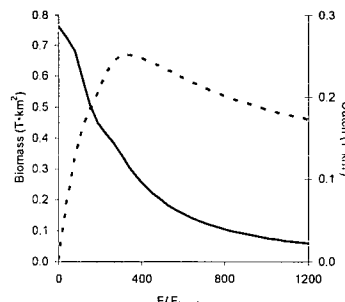
Sablefish



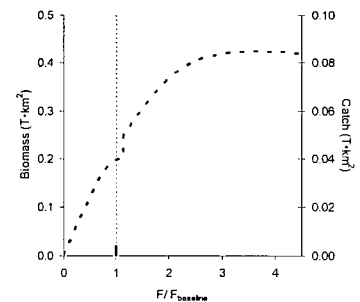
Lingcod



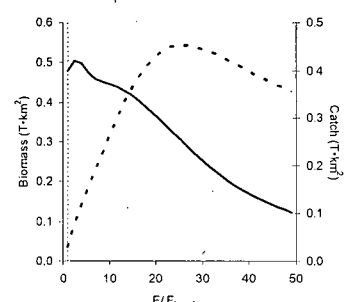
Shallow water benthic fish



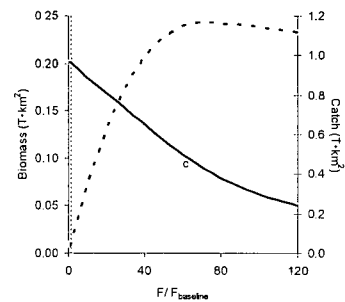
Skates



Crabs



Shrimp



Appendix 5.6 Comparison of Derived 2000 Model with Proper 2000 Model

Table A5.6.1 Comparison of derived 2000 model with proper 2000 model

Proper 2000 model is determined using current scientific data; derived 2000 model represents the ecosystem condition predicted by the 1950 model driven forward 50 years under historic fishing drivers and climate forcing. Biomass in $t\cdot km^{-2}$; Ecopath pedigree describes data quality of proper 2000 model; confidence intervals (CI) shown in Fig. A5.4.1. Predictions of the 1950 model agree with the 2000 model for most functional groups.

	Group	1950 biomass	(proper) 2000 biomass	(derived) 2000 biomass	derived /proper	2000 pedigree ranking	2000 pedigree CI	2000 biomass error (-/+)	Within CI?
1	Sea otters	5E-5	1E-4	1E-4	98%	5	0.3	3E-5	Y
2	Mysticetae	1.593	1.339	1.260	94%	5	0.3	0.402	Y
3	Odontocetae	0.062	0.061	0.074	120%	5	0.3	0.018	Y
4	Seals sea lion	0.122	0.258	0.173	67%	5	0.3	0.078	low
5	Sea birds	0.007	0.007	0.008	103%	4	0.5	0.004	Y
6	Trans. Salm.	0.633	0.208	0.206	99%	5	0.3	0.062	Y
7	Coho	0.136	0.024	0.022	90%	5	0.3	0.007	Y
8	Chinook	0.078	0.036	0.035	98%	5	0.3	0.011	Y
9	S. Squid	1.078	1.090	1.134	104%	1	0.8	0.872	Y
10	Squid	0.757	0.765	0.659	86%	1	0.8	0.612	Y
11	Ratfish	0.518	0.517	0.501	97%	1	0.8	0.414	Y
12	Dogfish	0.428	0.909	0.881	97%	1	0.8	0.727	Y
13	J. Pollock	0.132	0.132	0.111	84%	4	0.5	0.066	Y
14	Pollock	0.368	0.359	0.304	85%	5	0.3	0.108	Y
15	Forage fish	7.618	8.478	7.248	85%	1	0.8	6.782	Y
16	Eulachon	1.899	1.660	1.755	106%	1	0.8	1.328	Y
17	J. Herring	1.151	2.265	1.905	84%	4	0.5	1.133	Y
18	Herring	0.991	0.658	0.757	115%	6	0.1	0.066	high
19	J. POP	0.035	0.062	0.032	52%	4	0.5	0.031	Y
20	POP	1.006	0.561	0.494	88%	5	0.3	0.168	Y
21	Insh. Rock.	0.094	0.100	0.114	114%	4	0.5	0.050	Y
22	J. Pisc. Rock	0.008	0.007	0.005	76%	1	0.8	0.006	Y
23	Pisc. Rock.	0.543	0.654	0.579	89%	4	0.5	0.327	Y
24	J. Plank. Rock.	0.185	0.136	0.139	102%	1	0.8	0.109	Y
25	Plank. Rock.	1.220	1.207	1.157	96%	4	0.5	0.604	Y
26	J. Turbot	0.212	0.218	0.245	112%	4	0.5	0.109	Y
27	Turbot	1.532	1.530	1.480	97%	5	0.3	0.459	Y
28	J. Flatfish	0.259	0.259	0.252	97%	4	0.5	0.130	Y
29	Flatfish	0.504	0.236	0.342	145%	5	0.3	0.071	high
30	J. Halibut	0.404	0.628	0.233	37%	4	0.5	0.314	low
31	Halibut	0.442	0.628	0.529	84%	6	0.1	0.063	low
32	J. Pac. Cod	0.183	0.089	0.151	170%	4	0.5	0.045	high

Table A5.6 Comparison of derived 2000 model with proper 2000 model (cont.)

	Group	1950 biomass	(proper) 2000 biomass	(derived) 2000 biomass	derived /proper	2000 pedigree ranking	2000 pedigree CI	2000 biomass error (-/+)	Within CI?
33	Pac. Cod	0.354	0.163	0.230	141%	5	0.3	0.049	high
34	J. Sablefish	0.238	0.119	0.108	91%	4	0.5	0.060	Y
35	Sablefish	0.589	0.269	0.181	67%	4	0.5	0.135	Y
36	J. Lingcod	0.072	0.031	0.054	174%	1	0.8	0.025	Y
37	Lingcod	0.104	0.039	0.035	90%	4	0.5	0.020	Y
38	SWB fish	0.518	0.509	0.412	81%	1	0.8	0.407	Y
39	Skates	0.301	0.335	0.270	80%	5	0.3	0.101	Y
40	L. Crabs	0.491	0.456	0.401	88%	4	0.5	0.228	Y
41	S. Crabs	0.596	0.650	0.496	76%	1	0.8	0.520	Y
42	Shrimp	0.150	0.200	0.164	82%	4	0.5	0.100	Y
43	Epi. Inv.	11.094	13.448	9.155	68%	1	0.8	10.758	Y
44	Inf. Carn. Inv.	13.256	13.245	12.059	91%	4	0.5	6.623	Y
45	Inf. Det. Inv.	34.324	34.305	38.848	113%	4	0.5	17.153	Y
46	Jellyfish	3.006	3.000	3.109	104%	1	0.8	2.400	Y
47	Euphasiids	8.815	8.700	7.930	91%	4	0.5	4.350	Y
48	Copepods	4.683	4.667	4.551	98%	4	0.5	2.334	Y
49	Coral, sponge	1.929	1.929	1.914	99%	4	0.5	0.964	Y
50	Macrophytes	5.280	5.280	5.283	100%	1	0.8	4.224	Y
51	Phytoplankton	15.416	15.406	15.803	103%	4	0.5	7.703	Y

Appendix 6.1 Policy Search Parameters

Table A6.1.1 Market prices (\$·kg⁻¹) for *lost valley* fleet.

Groundfish trawl	
Ratfish	2.09
Dogfish	0.35
Pollock	0.31
A. POP	0.81
Inshore rockfish	0.81
A. pisc. rockfish	0.81
A. plank. rockfish	0.81
A. turbot	0.20
A. flatfish	0.73
A. Pacific cod	0.67
A. sablefish	0.63
A. lingcod	1.06
Skates	0.14
Large crabs	4.54
Shrimp trawl	
Ratfish	2.09
Dogfish	0.35
Eulachon	1.26
A. turbot	0.20
A. flatfish	0.73
S. benthic fish	0.52
Skates	0.14
Comm. shrimp	3.07
Shrimp trap	
S. benthic fish	0.52
Comm. shrimp	3.07
Herring seine	
A. herring	0.29
S. benthic fish	0.52
Halibut longline	
Inshore rockfish	0.81
J. turbot	0.20
A. turbot	0.20
J. flatfish	0.73
A. flatfish	0.73
J. halibut	2.56
A. halibut	2.56
A. Pacific cod	0.67
A. sablefish	0.63
A. lingcod	1.06
Skates	0.14
Salmon freezer troll	
Transient salmon	2.48
Coho salmon	1.44
Chinook salmon	3.70
Dogfish	0.35
Inshore rockfish	0.81
A. pisc. rockfish	0.81
A. plank. rockfish	0.81
Salmon wheel	
Transient salmon	2.48
Rockfish live	
Inshore rockfish	8.06
A. lingcod	1.06
Crab trap	
Large crabs	4.54
Small crabs	3.64
Clam dredge	
Epifaunal inv.	1.42
Aboriginal	
Transient salmon	1.65
Coho salmon	0.96
Chinook salmon	2.47
Eulachon	1.26
A. halibut	2.56
Recreational	
Coho salmon	19.15
Chinook salmon	49.39
Inshore rockfish	16.13
A. pisc. rockfish	16.13
J. halibut	51.16
A. halibut	51.16
A. lingcod	21.29

Table A6.1.2 Biomass/production (B/P) ratios by functional group.

B/P is an index for species longevity used to assign relative weightings of groups in ecology (B/P) objective function.

	1750	1900	1950	2000
Sea otters	7.6	7.6	7.6	7.6
Mysticetae	50.0	50.0	50.0	50.0
Odontocetae	25.0	25.0	25.0	25.0
Seals, sea lions	16.6	16.6	16.6	16.6
Seabirds	10.0	10.0	10.0	10.0
Transient salmon	2.0	2.0	2.0	0.4
Coho salmon	0.8	0.8	0.8	0.4
Chinook salmon	1.4	1.4	1.4	0.4
Small squid	0.2	0.2	0.2	0.2
Squid	0.2	0.2	0.2	0.2
Ratfish	5.0	5.0	5.0	10.2
Dogfish	9.0	9.0	9.0	10.2
J. pollock	4.4	4.4	4.4	1.0
Pollock	6.6	6.6	6.6	3.8
Forage fish	1.6	1.6	1.6	0.6
Eulachon	1.6	1.6	1.6	0.6
J. herring	0.8	0.8	0.8	0.4
A. herring	1.2	1.2	1.2	1.4
J. POP	3.0	3.0	3.0	1.4
A. POP	4.4	4.4	4.4	7.0
Inshore rockfish	5.4	5.4	5.4	5.2
J. pisc. Rockfish	3.8	3.8	3.8	3.8
A. pisc. Rockfish	27.0	27.0	27.0	27.0
J. plank. rockfish	3.8	3.8	3.8	3.8
A. plank. rockfish	14.8	14.8	14.8	14.8
J. turbot	3.0	3.0	3.0	3.0
A. turbot	4.6	4.6	4.6	4.6
J. flatfish	2.6	2.6	2.6	0.6
A. flatfish	3.8	3.8	3.8	1.0
J. halibut	10.2	10.2	10.2	1.6
A. halibut	15.0	15.0	15.0	2.4
J. Pacific cod	3.8	3.8	3.8	0.6
A. Pacific cod	5.8	5.8	5.8	0.8
J. sablefish	3.6	3.6	3.6	1.6
A. sablefish	5.4	5.4	5.4	3.6
J. lingcod	2.6	2.6	2.6	0.8
A. lingcod	3.8	3.8	3.8	1.2
S. benthic fish	3.8	3.8	3.8	0.6
Skates	6.6	6.6	6.6	3.2
Large crabs	0.6	0.6	0.6	0.6
Small crabs	0.2	0.2	0.2	0.2
Comm. shrimp	0.2	0.2	0.2	0.0
Epifaunal inv.	0.6	0.6	0.6	0.6
Inf. carn. inv.	0.4	0.4	0.4	0.4
Inf. det. inv.	0.8	0.8	0.7	0.7
Cam. jellyfish	0.1	0.1	0.1	0.1
Euphausiids	0.2	0.2	0.2	0.2
Copepods	0.04	0.04	0.04	0.04
Corals and sponges	100	100	100	100
Macrophytes	0.2	0.2	0.2	0.2
Phytoplankton	0.01	0.01	0.01	0.01

Appendix 6.2 Evaluation of ORB Ecosystems

Figure A6.2.1 Equilibrium harvest benefits from ORB ecosystems derived from 1750, 1900, 1950 and 2000.

Box and whisker plot shows variation in ORB solutions from $n = 25$ random-F initializations. Whiskers show total range, boxes show mean ± 1 SD. Harvest objectives: economic objective (profit), social utility (jobs), mixed objective (profit and B/P), ecological objectives ecosystem maturity (B/P) and biodiversity (Q-90).

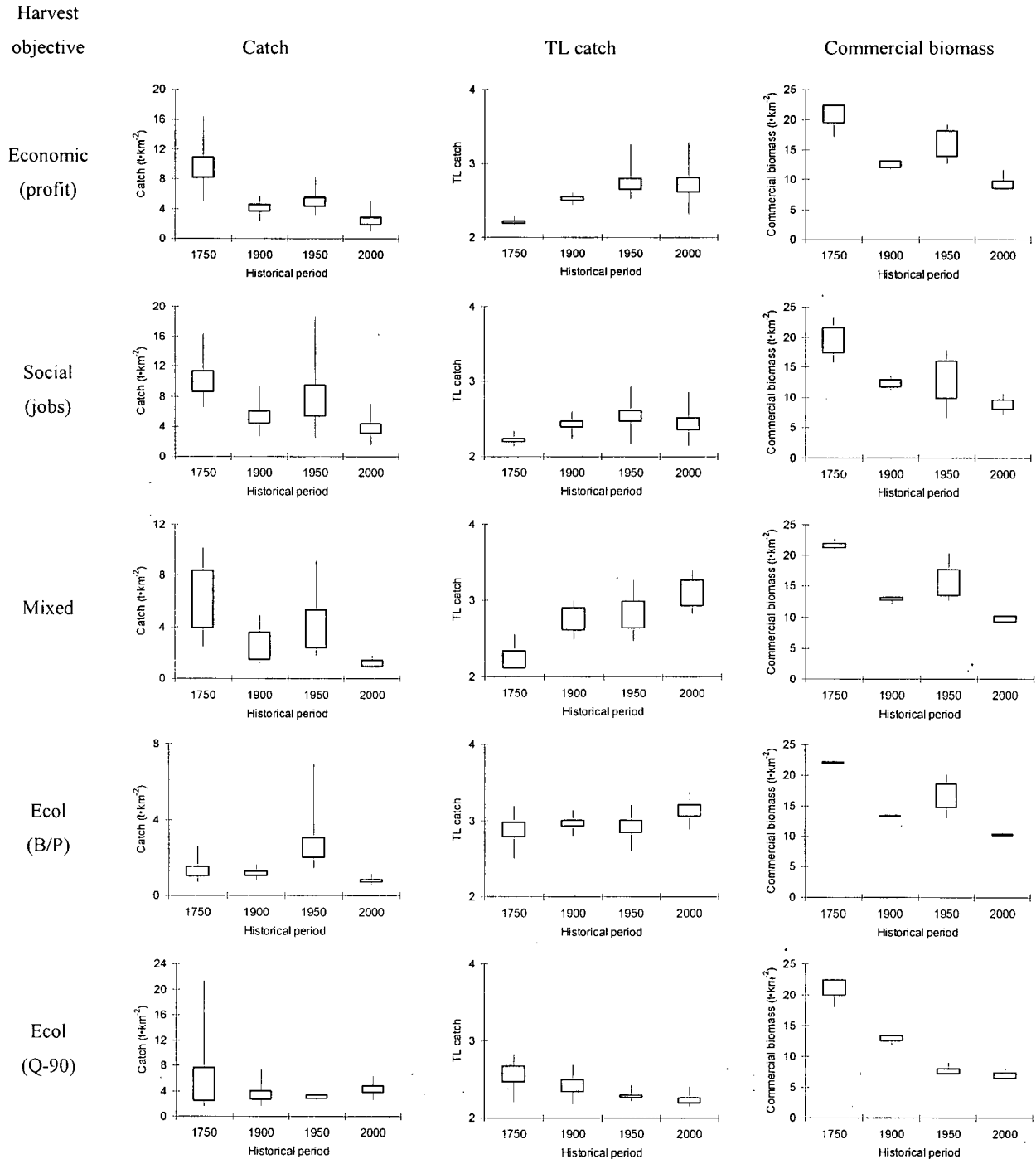


Figure A6.2.1 Equilibrium harvest benefits of ORB ecosystems (cont.)

Harvest
objective

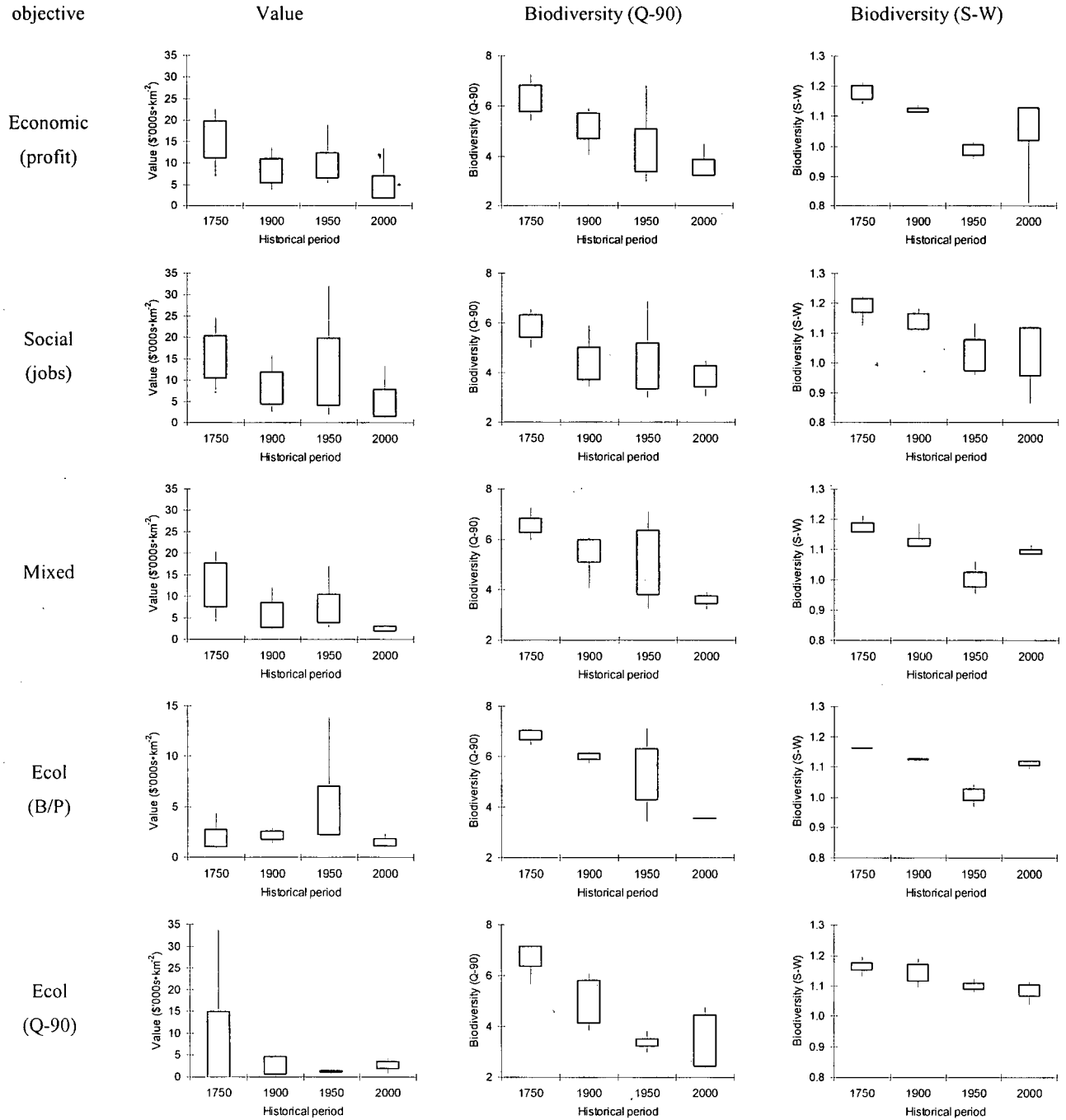


Table A6.2.1 Functional group biomass (t·km²) for selected ORB ecosystems.

One example ORB configuration is shown for each period and harvest objective; the example is selected from a series of random-F initializations, and represents the response surface peak most commonly located by the optimization routine. Mean c.v. is based on Monte Carlo runs ($n = 100$) varying biomass and production rate of the historic model (c.v. 10% and 5% respectively).

#	Groups	Social				Economic				Mix				Ecology (B/P)				Ecology (Q-90)			
		1750	1900	1950	2000	1750	1900	1950	2000	1750	1900	1950	2000	1750	1900	1950	2000	1750	1900	1950	2000
1	Sea otters	0.001	0.000	0.001	0.000	0.001	0.000	0.001	0.000	0.002	0.000	0.001	0.000	0.002	0.000	0.002	0.000	0.002	0.000	0.000	0.000
2	Mysticetae	2.905	1.666	2.099	1.460	2.867	1.723	2.067	1.474	2.878	1.710	2.064	1.310	2.804	1.637	2.074	1.425	2.972	1.740	1.669	1.585
3	Odontocetae	0.089	0.140	0.077	0.065	0.118	0.142	0.076	0.065	0.132	0.141	0.078	0.066	0.090	0.143	0.078	0.064	0.100	0.142	0.068	0.068
4	Seals, sea lions	0.082	0.074	0.217	0.276	0.094	0.070	0.210	0.257	0.092	0.072	0.231	0.299	0.094	0.070	0.258	0.261	0.082	0.070	0.106	0.266
5	Seabirds	0.008	0.021	0.014	0.009	0.009	0.019	0.014	0.009	0.009	0.018	0.014	0.009	0.011	0.018	0.017	0.008	0.008	0.019	0.006	0.010
6	Transient salmon	0.989	0.883	0.493	0.137	1.081	0.880	0.494	0.144	0.905	0.864	0.481	0.104	0.945	0.876	0.463	0.169	0.648	0.877	0.501	0.184
7	Coho salmon	0.122	0.193	0.092	0.004	0.289	0.203	0.093	0.024	0.315	0.205	0.105	0.018	0.261	0.221	0.097	0.025	0.244	0.194	0.103	0.032
8	Chinook salmon	0.036	0.033	0.087	0.011	0.085	0.153	0.087	0.039	0.090	0.147	0.105	0.034	0.288	0.169	0.089	0.040	0.185	0.156	0.100	0.043
9	Small squid	1.351	0.371	1.088	0.983	0.362	0.470	1.174	0.976	0.338	0.452	1.121	1.036	0.425	0.445	1.499	0.989	0.424	0.494	1.092	0.966
10	Squid	0.499	0.851	0.879	0.833	1.850	0.620	0.761	0.810	1.866	0.648	0.813	0.781	1.118	0.666	0.331	0.791	1.332	0.574	0.821	0.931
11	Ratfish	0.160	0.180	1.357	0.527	0.123	0.190	1.294	0.496	0.145	0.213	1.459	0.572	0.228	0.196	1.842	0.508	0.222	0.192	0.474	0.430
12	Dogfish	1.067	0.710	0.649	0.820	1.370	0.745	0.617	0.835	1.627	0.717	0.650	0.817	1.745	0.759	0.666	0.833	1.405	0.748	0.387	0.817
13	J. pollock	1.283	1.048	0.213	0.158	1.348	1.012	0.194	0.161	1.380	1.025	0.213	0.158	1.345	1.031	0.160	0.159	1.448	1.013	0.138	0.157
14	Pollock	0.501	0.710	0.594	0.506	0.582	0.652	0.493	0.601	0.738	0.678	0.592	0.478	0.726	0.713	0.330	0.587	0.733	0.648	0.411	0.555
15	Forage fish	31.753	23.676	11.068	9.883	29.731	24.594	10.905	9.630	30.610	24.251	11.405	10.212	32.021	24.067	12.503	9.466	31.927	24.701	7.002	9.154
16	Eulachon	7.290	4.407	3.260	2.061	6.275	3.868	3.161	2.088	6.699	4.669	3.168	1.895	6.836	4.982	2.886	2.076	6.701	3.996	1.754	1.872
17	J. herring	5.694	4.468	1.453	2.297	5.841	4.367	1.404	2.059	5.660	4.344	1.380	2.221	6.147	4.286	0.615	2.031	5.915	4.317	1.418	2.498
18	A. herring	6.943	3.014	1.039	0.773	7.391	2.853	0.928	0.804	6.653	2.833	0.949	0.669	8.179	2.760	0.319	0.753	7.197	2.796	1.076	0.839
19	J. POP	0.231	0.170	0.040	0.057	0.234	0.167	0.040	0.059	0.221	0.166	0.037	0.060	0.223	0.165	0.036	0.057	0.227	0.167	0.042	0.057
20	A. POP	1.301	1.132	1.038	0.764	1.509	1.152	0.952	0.781	1.485	1.189	1.087	0.727	1.414	1.188	0.747	0.785	1.544	1.160	0.855	0.778
21	Inshore rockfish	0.019	0.057	0.072	0.041	0.034	0.067	0.079	0.064	0.033	0.074	0.182	0.180	0.067	0.082	0.153	0.222	0.057	0.082	0.080	0.070
22	J. pisc. Rockfish	0.003	0.001	0.005	0.001	0.008	0.017	0.005	0.014	0.054	0.016	0.034	0.012	0.053	0.021	0.027	0.014	0.032	0.020	0.004	0.003
23	A. pisc. Rockfish	0.018	0.007	0.138	0.062	0.046	0.108	0.172	1.256	0.308	0.102	1.975	1.110	0.236	0.142	2.453	1.503	0.200	0.136	0.221	0.213
24	J. plank. rockfish	0.075	0.117	0.179	0.238	0.148	0.147	0.160	0.284	0.346	0.154	0.215	0.217	0.273	0.182	0.160	0.291	0.273	0.165	0.124	0.121
25	A. plank. rockfish	0.652	0.970	1.325	1.923	1.285	1.337	1.215	2.391	3.396	1.390	1.916	1.843	2.847	1.721	1.664	2.501	2.636	1.558	0.693	0.729

Table A6.2.1 Functional group biomass (t·km²) for selected ORB ecosystems (cont.)

#	Groups	Social				Economic				Mix				Ecology (B/P)				Ecology (Q-90)			
		1750	1900	1950	2000	1750	1900	1950	2000	1750	1900	1950	2000	1750	1900	1950	2000	1750	1900	1950	2000
26	J. turbot	0.320	0.013	0.397	0.140	0.249	0.059	0.582	0.013	0.076	0.039	0.410	0.186	0.080	0.010	1.226	0.013	0.148	0.058	0.136	0.039
27	A. turbot	3.016	0.095	2.125	0.823	2.037	0.492	3.313	0.071	0.541	0.313	2.181	1.120	0.524	0.078	7.200	0.071	1.184	0.492	0.977	0.217
28	J. flatfish	2.462	1.655	0.357	0.128	2.357	1.673	0.331	0.142	2.141	1.723	0.362	0.102	2.294	1.698	0.292	0.149	2.140	1.691	0.223	0.168
29	A. flatfish	1.488	1.140	0.643	0.115	1.435	1.155	0.561	0.127	1.567	1.214	0.653	0.092	1.553	1.205	0.422	0.133	1.618	1.176	0.386	0.147
30	J. halibut	0.435	0.240	0.407	0.698	0.391	0.323	0.595	0.589	0.168	0.329	0.747	0.673	0.386	0.304	0.947	0.567	0.318	0.321	0.242	0.490
31	A. halibut	0.812	0.260	0.143	0.756	0.495	0.532	0.281	0.689	0.117	0.542	0.425	0.813	0.445	0.433	0.725	0.665	0.262	0.526	0.426	0.705
32	J. Pacific cod	0.373	0.301	0.180	0.120	0.372	0.299	0.167	0.112	0.393	0.301	0.181	0.109	0.465	0.297	0.174	0.107	0.420	0.299	0.180	0.094
33	A. Pacific cod	1.592	1.082	0.885	0.219	1.689	1.098	0.814	0.200	1.789	1.111	0.847	0.190	1.775	1.110	0.729	0.188	1.961	1.122	0.276	0.162
34	J. sablefish	0.152	0.127	0.285	0.139	0.189	0.127	0.276	0.125	0.276	0.125	0.269	0.133	0.227	0.121	0.243	0.120	0.250	0.125	0.209	0.138
35	A. sablefish	0.170	0.671	0.575	0.311	0.204	0.692	0.563	0.279	0.325	0.682	0.546	0.297	0.355	0.644	0.497	0.266	0.287	0.681	0.502	0.301
36	J. lingcod	0.002	0.003	0.036	0.017	0.003	0.004	0.033	0.038	0.004	0.004	0.129	0.041	0.006	0.004	0.133	0.040	0.005	0.004	0.073	0.038
37	A. lingcod	0.050	0.032	0.020	0.013	0.068	0.106	0.020	0.044	0.096	0.107	0.123	0.050	0.121	0.111	0.132	0.051	0.128	0.113	0.100	0.046
38	S. benthic fish	6.461	4.826	1.596	0.682	8.312	4.452	1.282	0.778	8.114	4.546	1.546	0.634	7.157	4.647	1.063	0.793	7.410	4.436	0.488	0.739
39	Skates	0.240	0.135	0.864	0.424	0.055	0.170	0.819	0.309	0.010	0.163	0.737	0.420	0.024	0.124	0.833	0.300	0.032	0.166	0.275	0.359
40	Large crabs	0.550	0.444	1.852	0.818	0.358	0.507	1.669	0.747	0.796	0.514	1.889	0.882	0.949	0.494	1.868	0.714	0.824	0.506	0.357	0.624
41	Small crabs	2.245	1.831	3.193	1.047	2.043	1.909	2.981	0.966	2.593	1.910	3.035	1.157	2.977	1.837	3.413	0.942	2.916	1.920	0.362	0.820
42	Comm. shrimp	0.058	0.083	0.146	0.205	0.123	0.066	0.135	0.207	0.366	0.065	0.137	0.193	0.209	0.064	0.101	0.197	0.190	0.060	0.167	0.219
43	Epifaunal inv.	37.155	40.185	72.356	33.131	34.791	41.312	67.875	30.200	47.029	41.883	75.860	37.704	50.964	40.032	90.920	27.942	52.508	41.532	5.962	23.092
44	Inf. carn. inv.	13.347	1.942	1.071	0.921	10.207	2.171	2.009	1.878	1.908	1.745	0.931	0.938	0.011	2.646	0.827	5.079	0.459	2.062	14.286	4.054
45	Inf. det. inv.	39.779	37.878	40.467	32.475	41.581	37.812	40.007	35.025	45.609	37.935	40.065	26.798	42.518	37.889	35.508	32.514	43.344	37.618	33.427	38.065
46	Carn. jellyfish	5.185	3.326	2.996	2.987	5.510	3.470	3.000	3.024	5.529	3.391	3.025	3.034	4.871	3.422	3.106	3.033	5.024	3.508	3.051	2.963
47	Euphausiids	21.859	15.230	9.729	9.074	22.307	14.849	9.521	8.577	21.548	14.837	9.268	8.955	25.724	14.634	7.928	8.508	22.655	14.646	9.714	9.654
48	Copepods	13.839	8.625	4.885	4.924	13.185	8.785	4.968	5.027	13.510	8.736	4.981	5.017	14.363	8.761	5.439	5.087	13.305	8.833	4.899	4.829
49	Corals and sponges	1.144	2.234	1.959	2.874	0.852	2.315	1.958	2.867	1.646	2.750	1.956	3.071	2.776	2.735	1.951	3.203	2.335	2.396	1.939	1.762
50	Macrophytes	12.489	4.808	5.191	5.060	13.217	4.719	5.200	5.088	10.756	4.703	5.170	5.062	8.536	4.777	5.179	5.102	9.565	4.702	5.288	5.117
51	Phytoplankton	16.641	14.822	15.164	16.044	16.311	14.945	15.247	16.204	16.672	15.090	15.291	16.050	15.600	15.099	15.484	16.177	16.274	14.939	15.191	15.903
Mean biomass c.v.		11.0%	5.7%	3.6%	1.4%	23.8%	6.9%	4.0%	1.6%	6.8%	4.7%	3.4%	2.4%	0.0%	4.8%	3.9%	2.1%	1.5%	6.1%	1.9%	1.5%
Biodiversity (Q-90)		6.05	4.75	4.46	3.29	6.07	5.93	3.36	3.50	6.20	6.07	6.92	3.83	6.57	5.98	5.30	3.56	6.75	5.96	3.16	4.62

Table A6.2.2 Fisheries landings by gear type (t·km⁻²) for selected ORB ecosystems.

Annual landings for some example ORB ecosystems; one example is provided for each period and harvest objective; the example is selected from a series of random-F initializations, and represents the response surface peak most commonly located by the optimization routine. Uncertainty surrounding the equilibrium catch rate is determined through Monte Carlo analysis ($n = 100$), where basic Ecopath parameters for biomass and production are varied (c.v. 10% and 5%, respectively).

Landings	Social				Economic				Mix				Ecology (B/P)				Ecology (Q-90)			
	1750	1900	1950	2000	1750	1900	1950	2000	1750	1900	1950	2000	1750	1900	1950	2000	1750	1900	1950	2000
Groundfish trawl	0.224	0.189	0.169	0.045	0.257	0.078	0.050	0.224	0.073	0.046	0.103	0.027	0.049	0.052	0.044	0.023	0.076	0.042	0.167	0.167
Shrimp trawl	0.258	0.233	0.259	0.140	0.414	0.303	0.074	0.258	0.118	0.167	0.341	0.268	0.066	0.081	0.495	0.040	0.381	0.286	0.054	0.057
Shrimp trap	0.017	0.006	0.005	0.007	0.041	0.015	0.006	0.017	0.014	0.007	0.007	0.008	0.011	0.007	0.004	0.005	0.026	0.012	0.005	0.003
Herring Seine	0.171	0.110	0.037	0.049	0.058	0.248	0.051	0.171	0.176	0.133	0.070	0.079	0.221	0.158	0.022	0.045	0.411	0.168	0.051	0.053
Halibut longline	0.090	0.164	0.352	0.263	0.221	0.116	0.200	0.090	0.173	0.122	0.356	0.166	0.140	0.136	0.446	0.180	0.167	0.106	0.036	0.142
Salmon freezer troll	0.029	0.109	0.139	0.021	0.033	0.073	0.014	0.029	0.028	0.087	0.060	0.020	0.020	0.031	0.074	0.006	0.043	0.059	0.023	0.015
Salmon wheel	0.014	0.037	0.034	0.009	0.009	0.051	0.007	0.014	0.014	0.056	0.038	0.011	0.011	0.023	0.053	0.003	0.023	0.040	0.012	0.004
Rockfish live	0.005	0.015	0.021	0.011	0.005	0.007	0.009	0.005	0.005	0.006	0.012	0.004	0.003	0.004	0.004	0.004	0.003	0.003	0.006	0.007
Crab trap	0.132	0.207	0.115	0.018	0.558	0.043	0.080	0.132	0.057	0.015	0.132	0.040	0.027	0.029	0.044	0.025	0.089	0.040	0.018	0.061
Clam dredge	5.663	1.041	2.012	0.447	6.368	0.897	0.299	5.663	5.816	0.134	1.388	0.348	0.318	0.200	0.434	0.091	1.284	0.757	0.211	0.996
Aboriginal	0.260	0.357	0.381	0.245	0.381	0.396	0.137	0.260	0.127	0.274	0.416	0.389	0.089	0.134	0.656	0.089	0.397	0.366	0.063	0.093
Recreational	0.017	0.088	0.126	0.037	0.023	0.031	0.024	0.017	0.021	0.038	0.037	0.023	0.015	0.017	0.039	0.016	0.024	0.025	0.009	0.014
Total value (\$10 ⁶)	1074	565	835	326	1079	574	657	307	881	393	501	169	130	150	323	100	511	184	83	191
ORB value vs. today	549%	289%	427%	167%	552%	293%	336%	157%	450%	201%	256%	87%	67%	77%	165%	51%	261%	94%	42%	98%

Landings coefficient of variation (%)	Social				Economic				Mix				Ecology (B/P)				Ecology (Q-90)			
	1750	1900	1950	2000	1750	1900	1950	2000	1750	1900	1950	2000	1750	1900	1950	2000	1750	1900	1950	2000
Groundfish trawl	0.090	0.056	0.034	0.015	0.083	0.025	0.007	0.090	0.063	0.012	0.046	0.003	0.000	0.019	0.044	0.003	0.030	0.016	0.013	0.036
Shrimp trawl	0.059	0.147	0.235	0.073	0.090	0.216	0.032	0.059	0.041	0.117	0.233	0.206	0.000	0.046	0.260	0.015	0.060	0.207	0.014	0.022
Shrimp trap	0.006	0.003	0.002	0.004	0.216	0.009	0.003	0.006	0.004	0.004	0.003	0.004	0.000	0.004	0.002	0.003	0.001	0.007	0.001	0.002
Herring Seine	0.274	0.044	0.009	0.019	0.109	0.103	0.022	0.274	0.013	0.051	0.021	0.035	0.000	0.056	0.290	0.019	0.043	0.057	0.011	0.016
Halibut longline	0.035	0.103	1.039	0.079	0.046	0.079	0.053	0.035	1.532	0.073	0.847	0.052	0.001	0.077	0.662	0.044	0.359	0.067	0.011	0.038
Salmon freezer troll	0.031	0.059	0.038	0.005	0.011	0.028	0.004	0.031	0.009	0.032	0.018	0.009	0.000	0.013	0.031	0.001	0.007	0.025	0.006	0.005
Salmon wheel	0.003	0.028	0.025	0.009	0.001	0.037	0.007	0.003	0.004	0.044	0.029	0.015	0.000	0.017	0.042	0.003	0.008	0.032	0.009	0.003
Rockfish live	0.005	0.008	0.042	0.011	0.005	0.002	0.003	0.005	0.001	0.002	0.006	0.001	0.000	0.001	0.002	0.003	0.000	0.001	0.001	0.002
Crab trap	0.099	0.187	0.096	0.009	0.523	0.054	0.043	0.099	0.018	0.012	0.100	0.029	0.000	0.024	0.028	0.015	0.004	0.038	0.048	0.033
Clam dredge	15.619	0.652	0.987	0.124	49.601	0.650	0.143	15.619	3.329	0.070	0.578	0.099	0.000	0.112	0.158	0.054	0.199	0.500	1.355	0.347
Aboriginal	0.070	0.121	0.124	0.057	0.152	0.188	0.029	0.070	0.019	0.084	0.172	0.155	0.002	0.039	0.280	0.022	0.057	0.181	0.018	0.070
Recreational	0.040	0.065	0.062	0.014	0.012	0.009	0.006	0.040	0.009	0.012	0.016	0.006	0.000	0.005	0.026	0.005	0.003	0.008	0.001	0.004

Appendix 7.1 Input for Restoration Scenarios

Table A7.1.1 Catch profile for *maxdex* fleet.

Catch in t·km⁻². Fleet consists of 27 gear types, each catching one or two functional groups.

Gear#	Gear name	Catch	Functional group#
1	Seals	0.013	4
2	Seabirds	0.001	5
3	Trans. salm.	0.223	6
4	Coho salm.	0.013	7
5	Chinook salm.	0.015	8
6	Ratfish	0.012	11
7	Dogfish	0.049	12
8	Pollock	0.009	14
9	Forage fish	0.848	15
10	Eulachon	0.004	16
11	Adult herring	0.242	18
12	Adult POP	0.062	20
13	Inshore rock.	0.011	21
14	Pisc. Rock.	0.027	23
15	Plank. Rock	0.083	25
16	Turbot	0.095	27
17	Flatfish	0.060	29
18	Halibut	0.081	30,31
19	Pac. cod	0.055	33
20	Sablefish	0.044	35
21	Lingcod	0.023	36,37
22	SWB fish	0.001	38
23	Skates	0.036	39
24	Large crabs	0.027	40
25	Shrimp	0.037	42
26	Epi. Inverts.	0.080	43
27	Carn. jellyfish	<0.001	46
Sum		2.151	

Appendix 7.2 Candidate Restoration Trajectories

Figure A7.2.1 Restoration scenarios using the BC fishing fleet.

Unit of improvement is biomass; marginal improvement valuation model is linear. Optimization criterion varies economic benefit versus restoration success.

Groups

mandated for

Goal

rebuilding

ecosystem

Commercial biomass

Biodiversity

Profit

Sum of squares versus goal

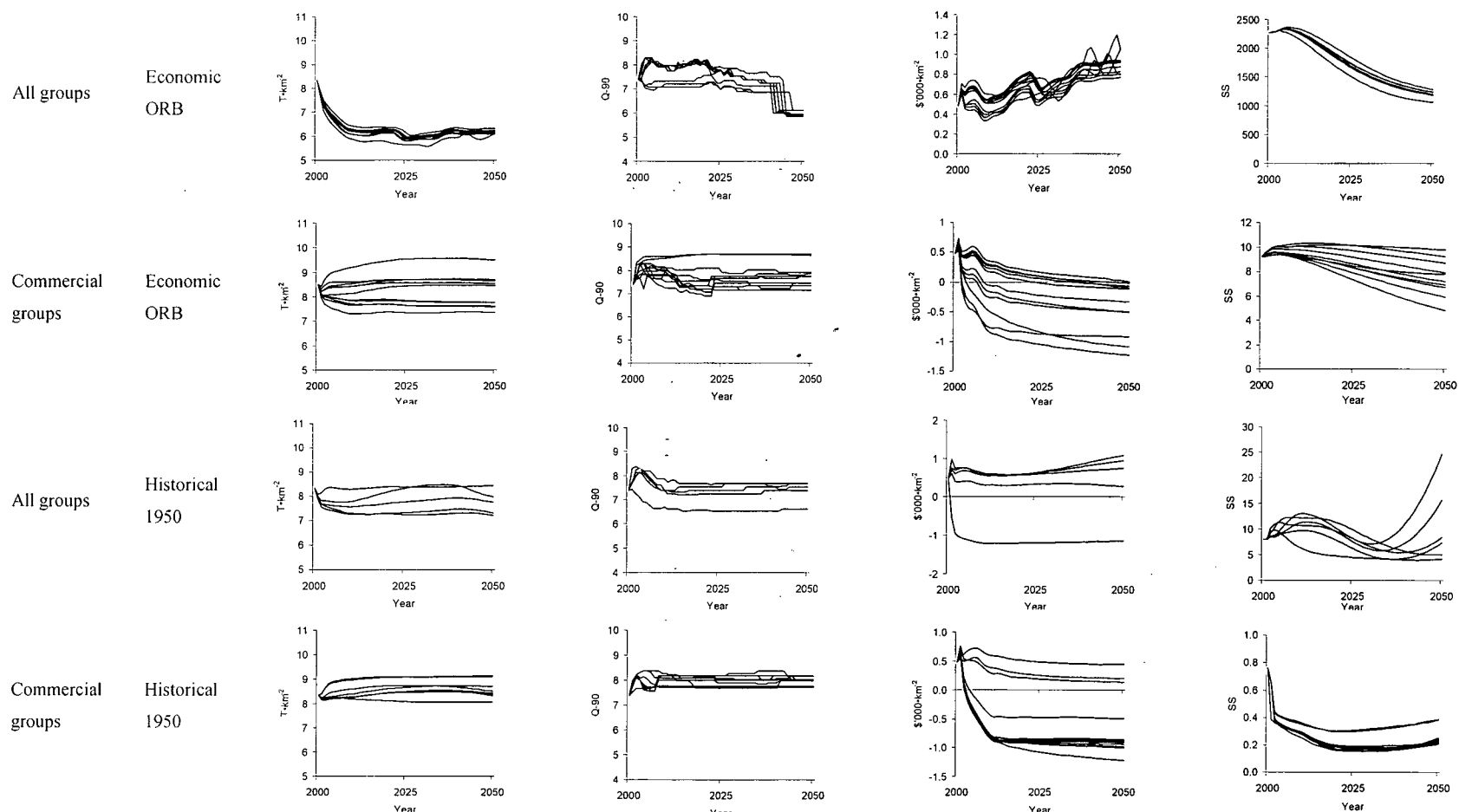


Figure A7.2.2 Restoration scenarios using the *lost valley* fishing fleet.

Unit of improvement is biomass; marginal improvement valuation model is linear. Optimization criterion varies economic benefit versus restoration success.

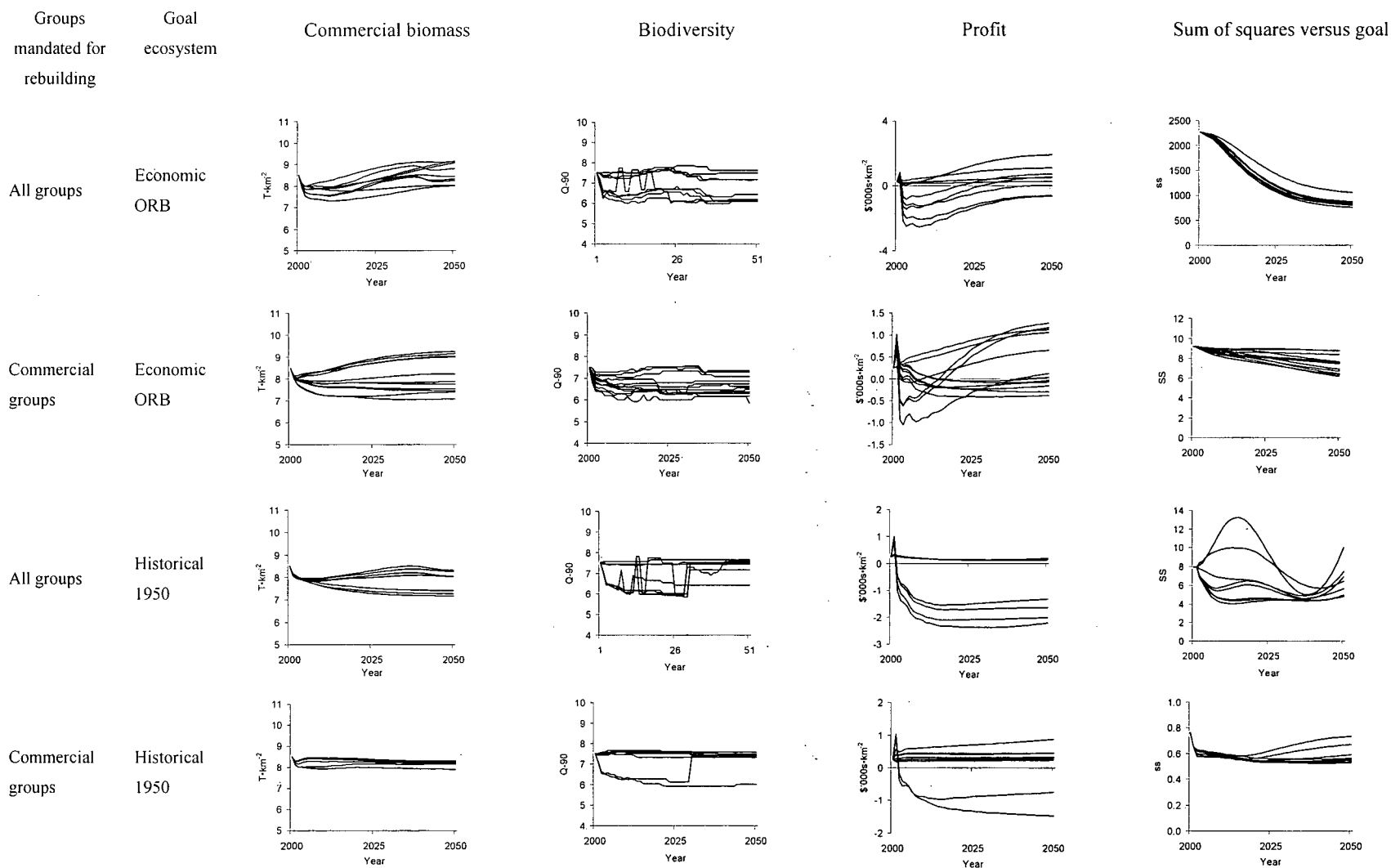
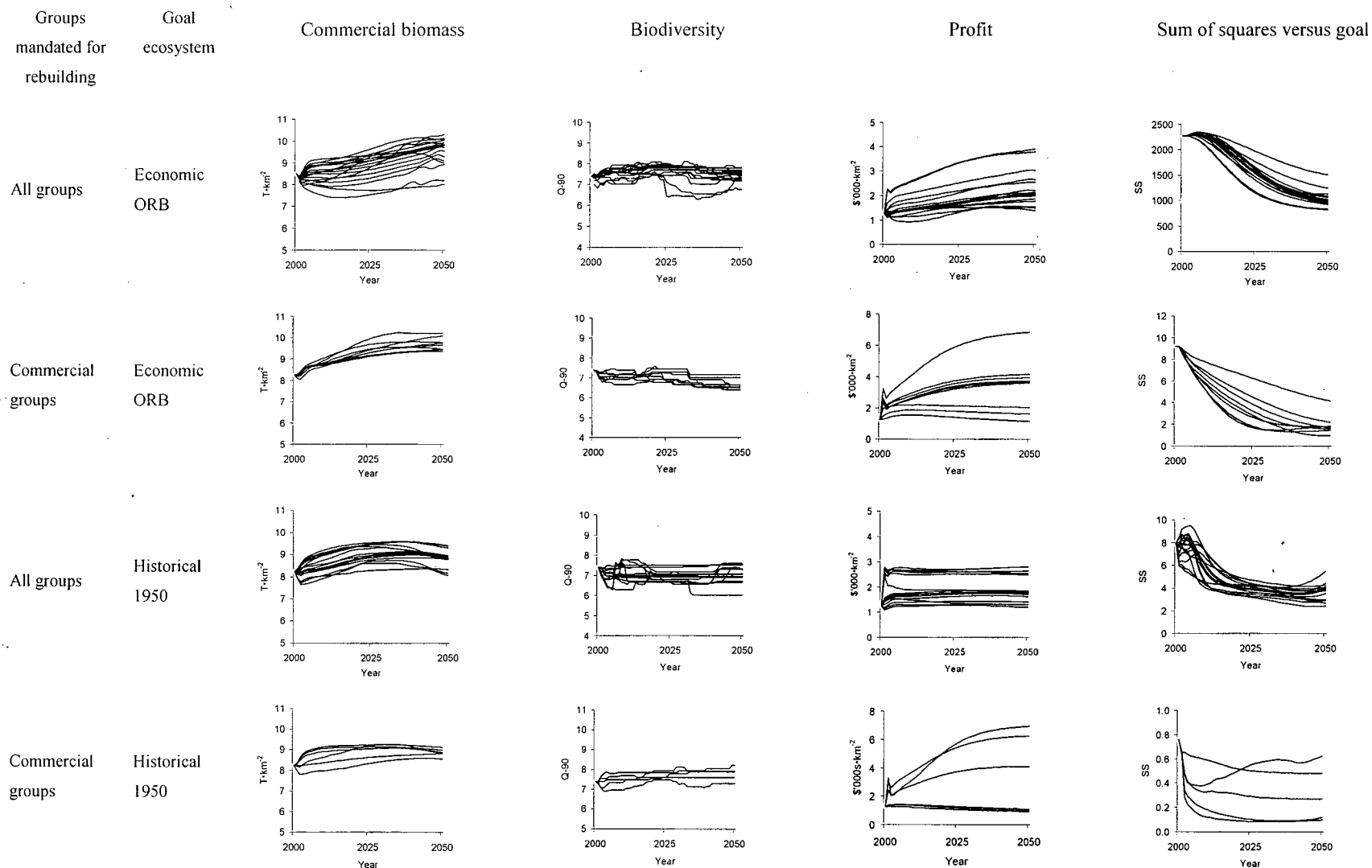


Figure A7.2.3 Restoration scenarios using the *maxdex* fishing fleet.

Unit of improvement is biomass; marginal improvement valuation model is linear. Optimization criterion varies economic benefit versus restoration success.



Appendix 8.1 Gwaii Haanas Spatial Investigations

Abstract

A preliminary spatial food web model is constructed for central and northern British Columbia using Ecospace. Various fishery closure schemes are simulated for the proposed Gwaii Haanas National Marine Conservation Area, on Moresby Island. Time closures (0, 2, 4, 6, 8, 10 and 12 months per year) test the effect of strict effort reduction in the MPA, and specific gear restrictions (no groundfish, no commercial) tests benefits of effort redistribution. Local and regional MPA effects are examined in terms of fisheries benefits and ecological impact. Coast-wide catch is improved over baseline under any MPA configuration, with greatest benefits occurring under a total closure to fishing. The model suggests that the recreational fleet and the commercial salmon fleet stand the most to gain from a partial or complete closure because of spillover effects. Although many groundfish and rockfish species benefit from MPA protection, cross-boundary groundfish catch does not improve because of lower fish dispersal rates. The herring fleet suffers reduced harvests under MPA protection schemes because of inflated piscivore populations. Although protection results in an overall loss of biomass within the MPA, the species assemblage changes to include proportionately more long-lived species and longer trophic chains compared to baseline (no area protection).

Introduction

This appendix summarizes the findings of a preliminary exploration into the economic, social and ecological benefits that could be provided by the proposed Gwaii Haanas National Marine Conservation Area (NMCA) on the southern end of Gwaii Haanas. Using EwE and Ecospace, I represent the marine ecosystem of northern BC and examine the effects of various area closure schemes for the proposed Gwaii Haanas National Marine Conservation Area (NMCA) in Haida Gwaii.

Benefit of marine reserves

As fishing technology improved, particularly over the last few centuries, humans became able to exploit areas of the world's oceans that were previously out of reach. When deep seas, offshore areas and areas with obstructive terrain were once safe from exploitation, they functioned as massive natural marine reserves that buffered fisheries impacts (Pauly 2003; Pauly *et al.*, 2005). However, today there are few areas left untouched by exploitation – and virtually none on the continental shelves. Combined with high exploitation rates, this has permitted depletion and stock collapse in many of the worlds most valuable fisheries (Christensen *et al.*, 2003; Myers and Worm, 2003).

In the face of declining marine resources, MPAs may offer an important tool to reduce fishing mortality, mediate habitat damage, increase stock biomass, and preserve ecosystem biodiversity in the face of environmental variability (Pitcher, 1997; Gell and Roberts, 2003). The spatial nature of the protection scheme may offer unique advantages over other strategies of conservation in terms of effectiveness and implementation. MPAs can be used to preserve sedentary benthic fauna and habitat from damaging gear, but mobile and migratory species can benefit as well if critical spawning and nursing habitats are protected. 'Source' populations, in a metapopulation context, can also be protected to ensure sustenance of sink populations.

The benefits of marine reserves as conservation tools extend beyond ecology. Compared to other conservation measures, this form of fishing control can be inexpensive to implement, easy to enforce and may encourage communities to take an active role in management. The wider public may support the establishment of marine reserves too for aesthetic purposes, and an area restricted to fishing may encourage and facilitate development of non-consumptive industries. Reserves that prohibit fishing activity also provide scientists with a baseline to evaluate the true impacts of human disturbance on marine populations elsewhere.

However, the 'spillover' effect may engage fishers' support where preservation of reproductive stocks leads to a net outflow of living resources to areas outside the marine reserve through random diffusion and/or density dependant emigration. Although the ecological benefits of MPAs are well established, direct experimental evidence for spillover remains scarce and

scientists are divided in the debate whether these fisheries benefits occur (e.g., Sladek-Nowlis and Yoklavich, 1999; Crowder *et al.*, 2000; Roberts *et al.*, 2001), and especially whether MPAs can be used to protect migratory stocks (Willis *et al.*, 2003). This report therefore considers the question; the simulations presented here give us an idea of the quantity and composition of spillover, and estimates how far from the reserve those benefits extend.

Importance of scale

In developing an ecosystem model, it is necessary to consider what questions will be asked of it. Its ability to predict dynamics, test hypotheses and deliver pertinent outputs will be wholly dependent on the model's structure and scale. Like the current effort, the two previous attempts to model the BC coast using Ecospace (Beattie, 2001; Salomon *et al.*, 2002) investigated benefits of marine protected areas (MPAs), but used very different scales in time, space and model complexity to answer their research questions.

Beattie's (2001) main objective was to determine the optimal size and placement of an MPA in order to maximize market and non-market benefits derived from living resources. For this, he developed a recursive algorithm, Ecospace, which increases the size of the MPA based on an initial 'seed' location – a single Ecospace cell that is at least partly closed to fishing. As the calculation progresses, fisheries benefits from the entire system are maximized through incremental increases in the area protected. The MPA grows in a regular fashion until optimality is reached. At each step, the routine calculates which adjacent grid cell will contribute most to the objective function. Optimal placement of the MPA can be determined by varying seed location. As in the present work, he used a large spatial scale (70,000 km²) to explore regional effects, and a long time simulation (50 years) to allow the ecosystem to reach equilibrium under simulations.

Salomon *et al.* (2002) used a smaller Ecospace map (1600 km²) and a short 10-year time horizon to resolve questions of fine scale edge effects and multi-zoning issues in an MPA surrounding Burnaby Narrows (SE Moresby Is.). Their analysis tested the ability of an MPA buffer zone (an area of restricted human use surrounding a no-take core) to protect system biomass, varying the relative size of the buffer and no-take zones, and testing different gear restriction schemes. Also

investigated, was the predator-prey biomass gradient established under various MPA designs, and indirect trophic effects caused by shifts in ecosystem structure under protection (i.e., cascade effects).

A small map size is particularly useful for (and practically restricted to) an examination of organisms with low dispersal rates and heterogeneous distributions, such as demersal fish and benthic invertebrates. This is no hindrance if small MPAs are best suited to protect the same. However, the problem that EwE and Ecospace have with migration (Martel, 2004) becomes acute at small spatial scales since many functional groups roam beyond the boundaries of the study area (see 'disadvantages' below). Because of this limitation, Salomon *et al.* (2002) made the simplifying assumption to exclude salmon from the model, though some other migratory groups were included.

Benefits of a large scale spatial model

At the large spatial scale adopted for this report, fishery activity represents an average of what is occurring in the reserve. Considering the whole NMCA as a uniform protected area type in Ecospace does not imply that the only testable management strategy is one of a single, sweeping area protection scheme that treats the entire reserve as a homogeneous unit. Actually, any number of zoning strategies can be modeled so long as the broad scale model adequately represents net effects inside the MPA. For instance, the effort reduction schemes tested in this paper may result, in practice, from any combination of time or area closures that affect gear types evenly. Effort redistribution schemes could be achieved through a number of strategies that differentially favour fishing sectors. The reason to use a broad scale model then, is to estimate regional consequences of the MPA zoning policy; fisheries and ecological effects that extend beyond the boundaries of the reserve, in this case affecting the ecosystem of northern BC. Use of a broad scale model for policy analysis would allow us to select an overarching management objective for the NMCA that is relevant at the regional scale (e.g., improved resilience of weak stocks, increased catch). Subsequently, we can choose an appropriate multi-zone configuration to achieve this effect while minimizing user conflict.

Benefits of a small scale spatial model

The main advantage of modeling a small geographic area at high resolution is that one can incorporate a detailed description of the study area. We can make use of available site-specific information as may be collected on a local scale. Small spatial models are especially well suited to represent animals of low dispersal rate, like sedentary and sessile invertebrates, and demersal fish. As such, the benthic environment becomes a critical and controlling factor in the simulation. The composition and distribution of the assemblage is highly influenced by sediment type, bottom structural complexity, relief and other physical determinants; these data can be used to develop a habitat map and determine species presence or optimality of habitat. Considering this information will lend reality to the predicted dynamics, and make a small spatial scale model more pertinent to the area studied.

Where predator and prey occupy overlapping habitats, it is only in the area of co-occupation where direct trophic interactions may occur. This constraint will considerably influence the Ecospace simulation, although spatial information is actually implicit in Ecopath and Ecosim as well. It is established through the diet matrix and flow control parameters (Walters *et al.*, 1997), which manage feeding interactions between functional groups. For instance, if in Ecopath we include juvenile herring as part of a piscivore's diet, we imply that the piscivore must spend at least part of its time feeding inshore. Accordingly, it is up to the modeler to make that piscivore available to any appropriate inshore predators. However, Ecospace makes our assumptions on group habitation explicit.

Disadvantages of a small scale model

As we reduce the geographic area represented by an Ecospace model, we increase the number of functional groups that tend to stray beyond the borders of the study area. Whether due to seasonal, periodic or ontogenic migrations, groups that move outside of the study area cause problems within the trophic simulation for a number of reasons.

Where feeding occurs outside the modeled system, a diet import term must be used in Ecopath to maintain the mass-balance assumption. The import term, however, is not subject to the same

systemic fluctuations in productivity that affect local diet components. In the case of the model presented here, Pacific salmon are doubly troubled by this. Juvenile phases consume in the adjacent freshwater systems, and adult phases consume throughout their oceanic migration. During periods of low productivity in the simulation, migratory Pacific salmon have an advantage over resident functional groups. Their diet import is steady and continuous, and it becomes a greater fraction of their total consumption when prey becomes scarce in the modeled region. The opposite is also true. During periods of increased productivity, salmon are not able to capitalize on available resources, and their biomass may fall in relation to other groups.

When translated into Ecospace, the fraction of total diet occupied by an external import is not subject to the same adjustment in feeding rate that distinguishes optimal from suboptimal habitat. In other words, where the division between optimal and suboptimal habitat is usually defined by adjustments in availability of prey and vulnerability to predators, Ecospace can use only the latter variable to define the distinction for a migratory species group that has diet import, since their feeding is largely unaffected by trophic dynamics in the model.

In the case of functional groups that undergo range expansion into the modeled area as may be due to temperature induced regime shifts, or range collapse out of the modeled area as may be due to over-harvesting, other variables than diet composition will fail. In this case, range inhabited is a function of stock size. Where only a fraction of the contiguous population is represented in the system, the recruitment ability of the adult stock may be underestimated, particularly at low stock size. If a supporting biomass of adult animals in the adjacent system is, in actuality, available to sustain the fraction of the stock considered by the model, then depletion of the functional group within the modeled system will not be accompanied by the expected reduction in recruitment, and the safe level of harvest may be underestimated. This is also true in the case of metapopulations, where an external adult pool may fortify the modeled stock.

Methods

The EwE models used for this analysis are based on Ainsworth *et al.* (2002). Costs of fishing and market prices for species are based on this volume (Appendix Table A5.1.6).

Habitat type

The Ecospace base map divides northern BC into 26 x 38 grid cells (Fig. A8.1.1). Nine Ecospace habitat types are identified by depth and tidal speed (Table A8.1.1). Most of the habitat types describe areas less than 500 m depth on the continental shelf. 'Shallow' refers to areas less than 20 m in depth, and is further divided into areas of high and low tidal speed (greater than or less than $30 \text{ cm}\cdot\text{s}^{-1}$). 'Continental slope' (501 m – 1000 m) describes the transition to the bathyal zone, or 'offshore' area, which constitutes the remainder of the map and extends from 1001 m to 4032 m (maximum depth). NB: I include a special habitat for sponge biohermes in central Hecate Strait and Queen Charlotte Sound (Conway, 1999; Sloan *et al.* 2001). Hexactinellid sponges are not explicitly modeled,

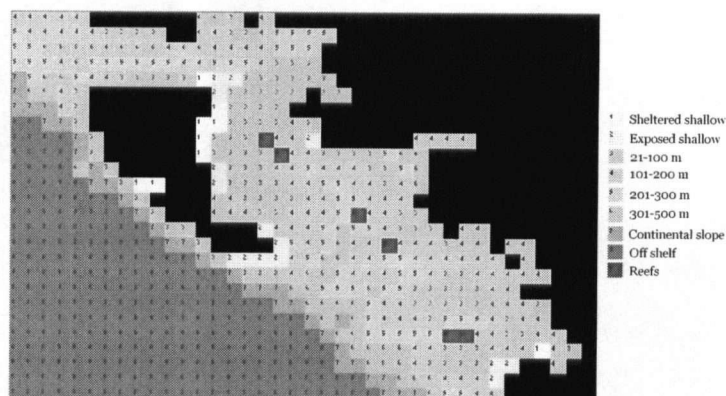


Figure A8.1.1 Ecospace habitats. Nine habitat types are described, demarked by depth and tidal speed.

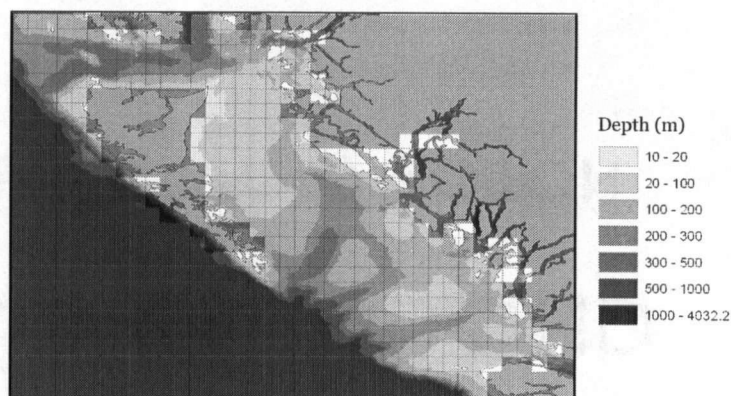


Figure A8.1.2 Bathymetry. Contours show depth classification used to assign habitats. Overlaid is Ecospace grid showing land cells (shaded).

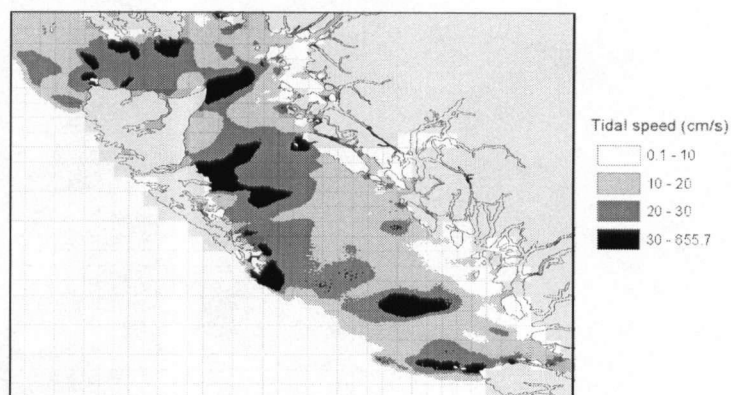


Figure A8.1.3 Tidal speed. Areas of high tidal speed ($> 30 \text{ cm}\cdot\text{s}^{-1}$ dark areas) were used to classify 'exposed shallow' habitat (in 'shallow' habitat only $< 20 \text{ m}$). Overlaid is Ecospace grid showing land cells (shaded).

the aggregate Ecopath group 'corals and sponges' currently represents them. Figs. A8.1.2 and A8.1.3 show depth and tidal speed in the study area, supporting information used to classify habitat types.

Table A8.1.1 Ecospace habitat definitions.

Habitat	Name of habitat	Depth range	% Area
0	All habitats	10-4032.2 m	100%
1	Sheltered shallow	10-20 m	1%
2	Exposed shallow ¹	10-20 m	2%
3	21-100 m	21-100 m	17%
4	101-200 m	101-200 m	19%
5	201-300 m	201-300 m	9%
6	301-500 m	301-500 m	8%
7	Continental slope	501-1000 m	5%
8	Off shelf	1001-4032.2 m	37%
9	Reefs	~160-230 m	< 1%

¹ Includes tidal speeds $>30 \text{ cm}\cdot\text{s}^{-1}$

Occurrence in habitat

In order to represent the spatial distribution of all functional groups in an ecosystem, it would be necessary to model a large number of habitat types in Ecospace, enough to cover all observed combinations of species occurrence. To simplify the spatial representation, it is necessary to describe habitats in broad terms. Depth has been used (Beattie 2001), or used in combination with a physical description of the substrate (e.g., Okey and Pauly 1998; Pitcher and Buchary 2002a/b) to classify areas. Since Ecospace offers a two-dimensional representation of the environment, each habitat type must implicitly describe both benthic and pelagic zones. For this reason, pelagic near-shore species as herring are also present in deep habitats despite being confined to the upper water column.

Occurrence by habitat, as entered into Ecospace, only describes the preferred habitat of the functional group. By altering dispersal parameters, the distinction between optimal and

suboptimal habitat types can be blurred or removed. Appendix Table A8.2.1 summarizes habitat occupancy. Fishery presence by habitat types is shown in Appendix Table A8.2.2.

Base dispersal rate

Dispersal rate, measured in kilometers per year, represents the average distance traveled in a single year by individuals within the study area (e.g., between mark and recapture locations). The value will affect how quickly a functional group spreads to occupy adjacent regions. Direct quantitative estimates from tagging studies were available for only two groups, herring and sablefish. The parameter is not related to migration or motility but reflects the tendency to roam as a result of random movements (V. Christensen, UBC Fisheries Centre, *pers. comm.*).

Failure to home was taken as an index of dispersal rate for migratory salmon functional groups, for which tagging data were available. The relative base dispersal rate of salmon groups was set to agree with the rank order of straying rates suggested from tagging studies. The group 'transient salmon' was assumed to stray the most, containing pink salmon, which are notorious strayers (Mortensen *et al.* 2000), and chum salmon, which stray the most of all species (Tallman and Healy 1994). Following transient salmon are coho and chinook (Appendix Table A8.2.3).

For other functional groups, a number of guidelines were used to parameterize dispersal rates. Territorial groups, rockfish and lingcod, were assigned a low rate. Their juveniles received a higher dispersal rate relative to the adults. A common value was assigned to all groups whose primary means of dispersal is through planktonic drift (i.e., jellyfish, zooplankton and phytoplankton; or through propagules, as in the case of sessile invertebrates). The absolute dispersal rate in this case was set lower ($200 \text{ km} \cdot \text{yr}^{-1}$) than most actively swimming groups.

The precise value used to describe the base dispersal rate becomes less important as the spatial-temporal model approaches equilibrium (D. Pauly, UBC Fisheries Centre, *pers. comm.*). Therefore, under long time horizons as the one applied in this report, the resulting end-state ecosystem configuration should be invariant to the initial dispersal rates modeled (supported by testing). Only the speed of the corrective reaction after a system perturbation should be affected by this rate in most cases. However, in specific cases, the end-state ecosystem configuration

may be affected by the dispersal rates (C. Walters, UBC Fisheries Centre *pers. comm.*). For instance, if a juvenile group recruits in an inshore spawning area, and must then travel to join the offshore adult pool, their dispersal rate must be fast enough to transverse the sub-optimal habitat (where they are vulnerable) before they are completely consumed by predators. In cases like this, dispersal rates should be set high for juvenile groups, or their vulnerability to predation in sub-optimal habitat kept low.

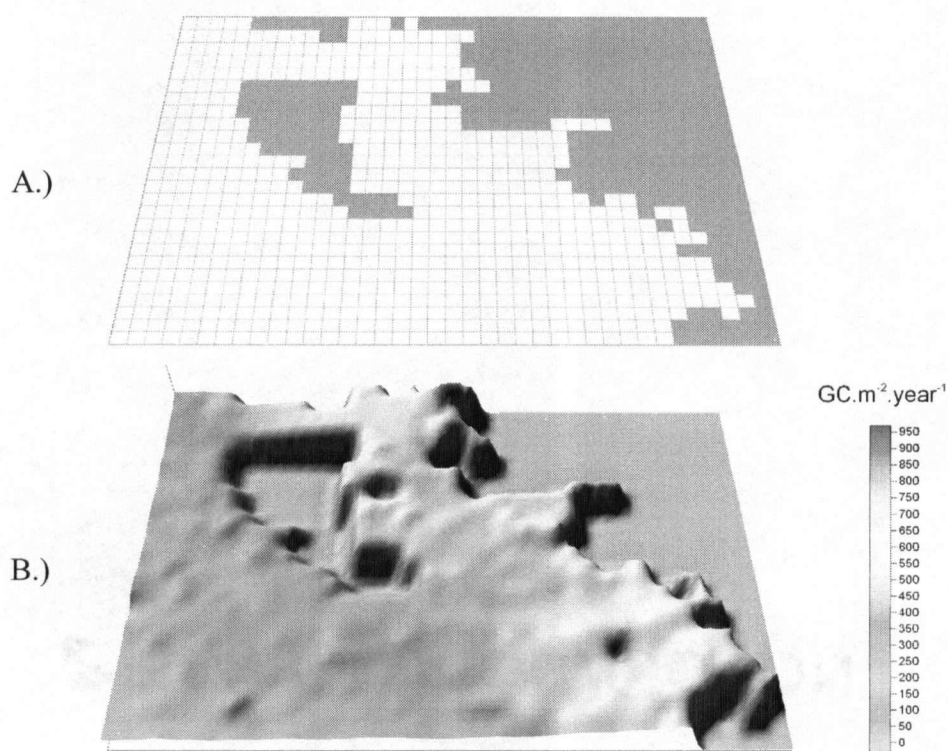


Figure A8.1.4 Primary production forcing pattern used in Ecospace. A.) Ecospace base map; B.) Primary production. *Source:* Global Monitoring of Oceanic Resources. SAI, Italy.

Primary production

A value representing primary production is associated with each marine grid cell in the basemap (Fig. A8.1.5). Ecospace scales the input values (entered here in $\text{gC}\cdot\text{m}^{-2}\cdot\text{y}^{-1}$) into a relative index of primary production, and distributes phytoplankton biomass proportionately. As available

production trickles up the food web, secondary and tertiary production is also affected, leading to spatial concentration of predator and prey species.

Primary production data used to parameterize this model were taken from the Global Monitoring of Oceanic Resources (GoMOR) database provided by the Joint Research Centre of Space Applications Institute, Italy. Data represent depth integrated aerial primary production (integration at depth is to the -0.1% light level) at 18 km^2 cell resolution, and is averaged for all of 1998 based on monthly information. Each grid cell in the GoMOR dataset contains a primary production value measured in $\text{gC}\cdot\text{m}^{-2}\cdot\text{y}^{-1}$. If the grid cell contains land, or if there were no available production data for that area, the dataset included a value of zero. When the available dataset was applied to this model, several marine cells were discovered to lack associated values for primary production. In this case production was taken as the average value of adjacent cells.

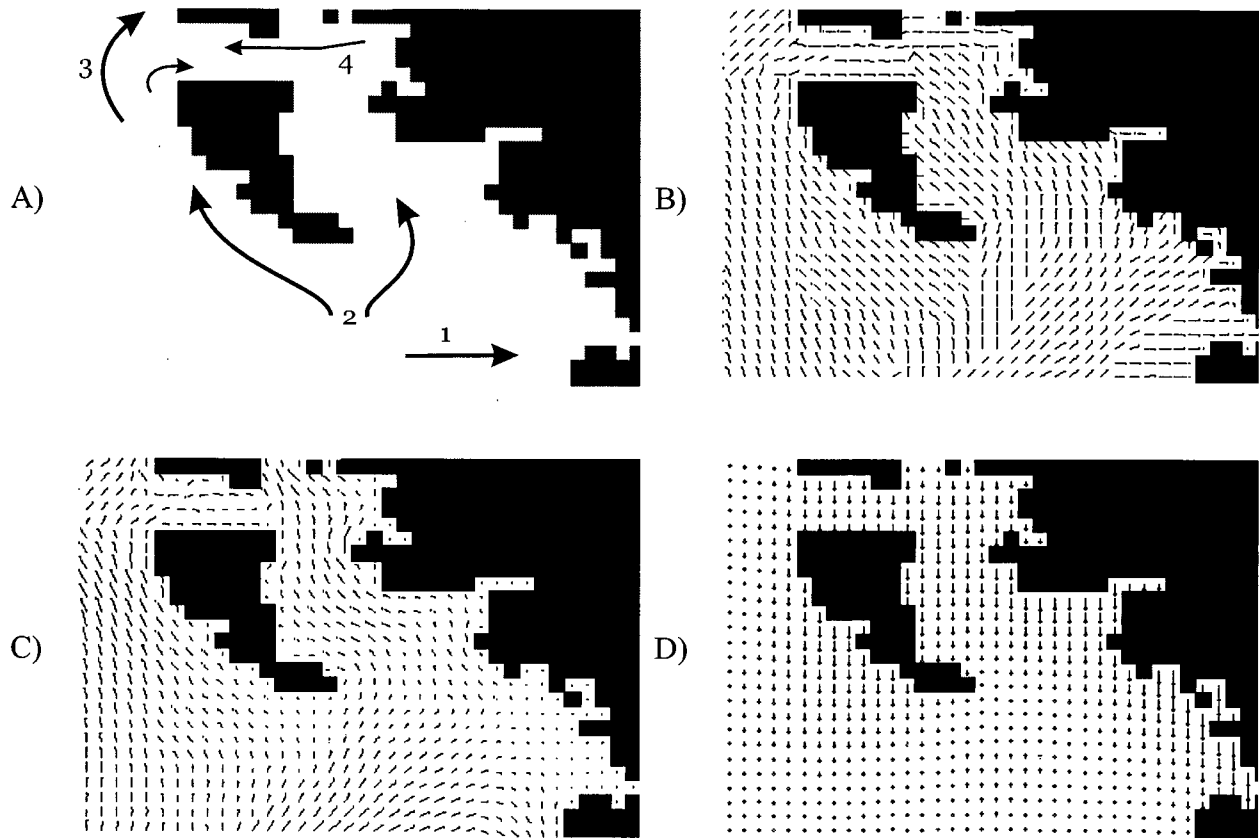


Figure A8.1.5 Modeled current circulation. A.) Conceptual diagram showing modeled residual currents. Numbers correspond to oceanographic features mentioned in text; B.) Advection field sketched as Ecospace input. Arrows show direction of residual current; C.) Resultant transport velocities calculated by Ecospace; D.) Resultant upwelling and downwelling zones calculated by Ecospace. Down arrows show downwelling zones (mainly W. coast of Haida Gwaii and Mainland Coast of Queen Charlotte Sound and Hecate Strait); up arrows show upwelling zones.

Advection

Ecospace allows the user to sketch in circulation or wind as a forcing pattern that will affect surface currents. The advection routine solves linearized pressure field and velocity equations for each grid cell to estimate horizontal and vertical flow rates at equilibrium (maintaining mass-balance of water movement). It considers upwelling/downwelling rate, water acceleration due to bottom slope, bottom friction and the Coriolis Effect in order to generate an advection pattern that is sensitive to the base map design. The modeled flow represents net water movement, or residual current, and is irrespective of bi-directional tidal motion (though not asymmetric tidal motion). Any functional group may be associated with the resulting pattern, so that its

movement is influenced by advective drift in addition to swimming and random movement. With this, we can model oceanographic features; for example, a convergence zone with concentrating effects on pelagic components, or an upwelling zone's effect on primary productivity. When applied to primary and secondary producers, the advection field has the effect of smearing the underlying map of productivity in the direction of the net current. The advection model created for this exercise was based on the descriptions of physical oceanography provided by Thompson (1981), Ware and McFarlane (1989) and Hunt (1995).

The advection field modeled in this exercise (Fig. A8.1.5) contains the oceanographic features listed below. Functional groups that were associated with the advection pattern include carnivorous jellyfish, euphausiids, copepods and phytoplankton.

1. Divergence of North Pacific Current into the northeastern flowing Alaska Current and southeastern flowing California Current near the northwest tip of Vancouver Island. Although the centre of the divergence actually occurs at latitudes south of the study area (approximately southern BC or Washington), transition is confused between 45-50° N (Thomson, 1981). It is modeled in this exercise as a region (surrounding the seaward face of Vancouver Is.), where north-south flow is minimal, currents run almost perpendicular to the coast.
2. The North Pacific Current enters Hecate Strait in a northeasterly direction via Queen Charlotte Sound, and on the continental slope, flows northeast along the western shore of Haida Gwaii.
3. There is a downwelling zone along the northwest extent of the study area, where the Alaska Coastal Current (reinforced by northeasterly winds in fall/winter) skirts north along the panhandle, causing coastal convergence.
4. Freshwater outflow (current is modeled, not salinity) from Chatham Sound, along the north shore of Dixon Entrance, into the open sea and north towards the Gulf of Alaska. Also modeled, is the saline counter flow along the south side of Dixon Entrance. This exchange is most significant in the spring during peak run-off. It occurs year-round and is reinforced by a counter-clockwise vortex in the centre of Dixon Entrance, which is

caused by asymmetric tidal action (Thompson 1981). However, cell size is too coarse to incorporate this effect.

5. Coastal convergence occurs mainly on the west coast of Haida Gwaii and along the mainland shoreline of Queen Charlotte Sound and Hecate Strait.

Tuning the spatial model

The initial Ecospace model was well behaved. No runaway or oscillating dynamics were present, and only slight changes to dispersal patterns were required to cause the dynamic behaviour of most functional groups to conform to guidelines in Table A8.1.2. Note that some early changes in group abundances are expected under baseline simulation, as groups settle to their equilibrium level (biomass accumulations are present). The model takes between 15 and 20 years to stabilize under baseline fishing mortalities. The equilibrium condition arrived at by the dynamic model depends on the base level of exploitation inherited from Ecopath, in addition to other parameter specifics. However, no functional group was allowed to decrease in biomass by more than 50% of its baseline abundance. Problems encountered with group behaviour involved depletions in biomass and unrealistic or patchy distributions.

Table A8.1.2 Group behaviour guidelines used to standardize Ecospace functional groups.

	Effect on functional group biomass	
	Optimal habitat	Sub-optimal habitat
No fishing	Increase	Increase
Baseline F	No change	No change
Increased F	Depletion	Depletion/Extinction
Much increased F	Depletion/extinction	Extinction

Since most functional groups are restricted to shelf habitat, there is a high concentration of predators in those regions. As a result, prey species which also inhabit oceanic waters as prime habitat (i.e., 'offshore' areas), tended to increase in biomass offshore (where they were relatively free from predators), while becoming depleted on the shelf. Groups most prone to this error were large and small squid. In order to distribute their biomass more evenly, offshore areas were removed as prime habitat, and then dispersal parameters were adjusted to allow them to occupy off shore areas. Vulnerability to predation in bad habitat was reduced, and relative feeding time in bad habitat was increased (Appendix Table A8.2.4). This distributed their biomass more evenly over the whole study area, with a slim line of concentration appearing over the continental slope in the case of large squid. A similar procedure was used to correct the distribution of small squid. A less severe distribution problem in forage fish, pollock and skates was managed by increasing their dispersal rate. Their populations became more continuous, and predation effects more evenly distributed.

In preliminary tests of the spatial model, chinook salmon became depleted to less than one third of their initial biomass under a 50-year baseline simulation, and went practically extinct under increased fishing pressure. The negative biomass accumulation originally described in Ainsworth *et al.* (2002) contributed to the effect. Although negative accumulation was retained, dispersal parameters were edited to slow the decline after about 15 years of simulation.

Vulnerability to predators in bad habitat was reduced, and feeding rate in bad habitat was increased. Modifications to dispersal parameters restricted their baseline depletion to about 40% by the time equilibrium was established. A gradual decline was allowed to persist in order to reflect recent real-world evidence.

Results Regions

Ecospace can summarize spatial results according to output regions, which are set by the user. Fig. A8.1.6 shows the 19 output regions used and Appendix Table A8.2.5 provides a description of each. The output regions were designed to correspond roughly to DFO management areas to facilitate easy comparison of model output with DFO data. Regions 1 and 2 represent the east and west coasts of the proposed NMCA. Regions 3 and 8 are adjacent to the protected zone. These were included to examine local ‘spillover’ effects.

MPA Investigations

Ecological effects of five effort reduction schemes are tested for the proposed NMCA (2, 4, 6, 8, 10 and 12 month closures) and compared to the status quo, 0 month closure (i.e., “full exploitation”). Fishing rates during open seasons remain at baseline levels for all sectors and closures affect all sectors evenly. Additionally, two special effort redistribution schemes are tested, which include gear restrictions “no trawl” and “no commercial”.

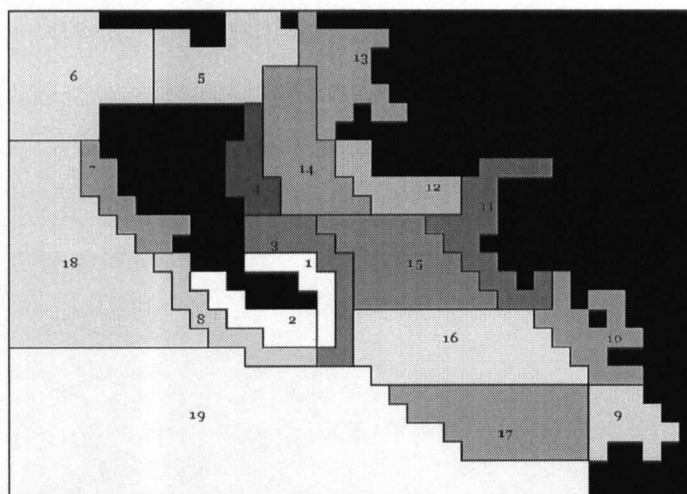


Figure A8.1.6 Ecospace output regions used to summarize results by area.

Results

Fisheries Impacts

Change in total catch

In Fig. A8.1.7, catch is shown as a percentage of baseline catch. Warm regions indicate an improved catch in the presence of gear restrictions and cool regions indicate a sacrifice in catch compared to full exploitation. The chasm around Moresby Island shows reduced catch occurring within the NMCA because of fishery closures. Results are presented by output regions. Regions adjacent to the marine reserve (areas 3 and 8) show an improved catch under all gear restriction models.

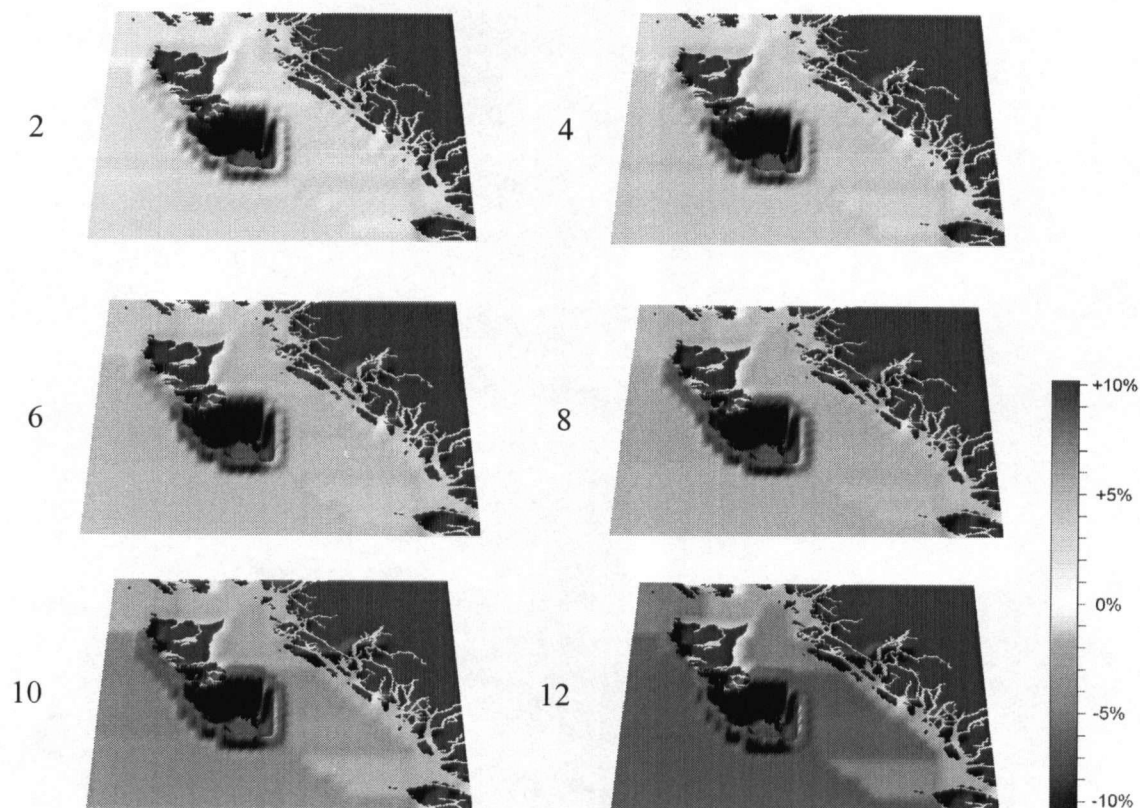
Catch per gear type

Fig. A8.1.8 shows the regional effects of fishery closures in the NMCA as percent change in landings from baseline. Fleet-wide effort reduction programs (i.e., 12 and 6 month fishing closures) are compared with specific gear restrictions (2 scenarios: no groundfish and no commercial fisheries). Protection increases the abundance of piscivorous fish (Fig. A8.1.12); this causes a decline in forage fish species.

Under monthly closures, gear types that pursue high trophic level fish do well (recreational sector, salmon fleet, groundfish fleet), but herring gillnet and herring seine suffer. The recreational fleet reaches a maximum increase in landings under the 12 month MPA closure, 29% over baseline. The rest of the salmon fleet sees an increase between 5-10% (Fig. A8.1.10). Benefits are reduced considerably from this level under the 6 month closure.

Under the 12 month closure, the significant size of the MPA permits total coast-wide landings to increase by as much as 4.4%; under 6 month closure, total landings increase by 1.3%. When groundfish catch is removed from the MPA, biomass increases for several groundfish species (Fig. A8.1.10). However, groundfish have a low dispersal rate and production is not transferred

Monthly closures



Gear restrictions

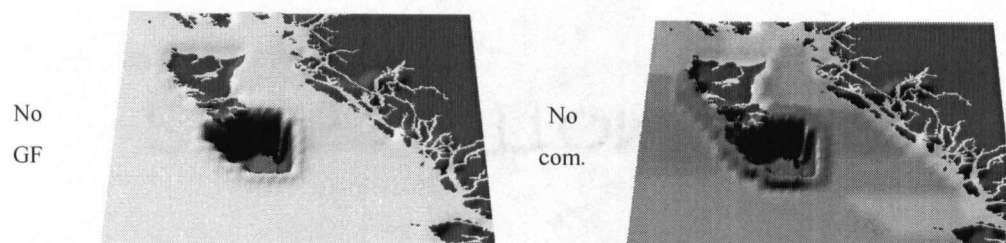


Figure A8.1.7 Catch by output region. Change in catch over baseline (full exploitation) under monthly closures and gear restrictions. Results presented by output region. Shown are 2-12 month closures and gear restrictions (No GF = no groundfish; No com. = no commercial). Red and yellow show an increase in catch; blue and cyan show a decrease. The pit around Moresby Island results from NMCA fishing closures.

outside the protected area. There is little net change for any fishery, and excluding groundfish fisheries from the MPA has a barely noticeable effect on coastal landings. NB: secondary gear effects are not considered in this model, so the ecological benefits of a demersal fishery exclusion policy may not be fully expressed. Closure to all but the recreational fleet results in a substantial increase in salmon availability outside the MPA, except for the prime target species of the recreational fleet, chinook. Appendix Fig. A8.2.1 shows the value of catch per gear type.

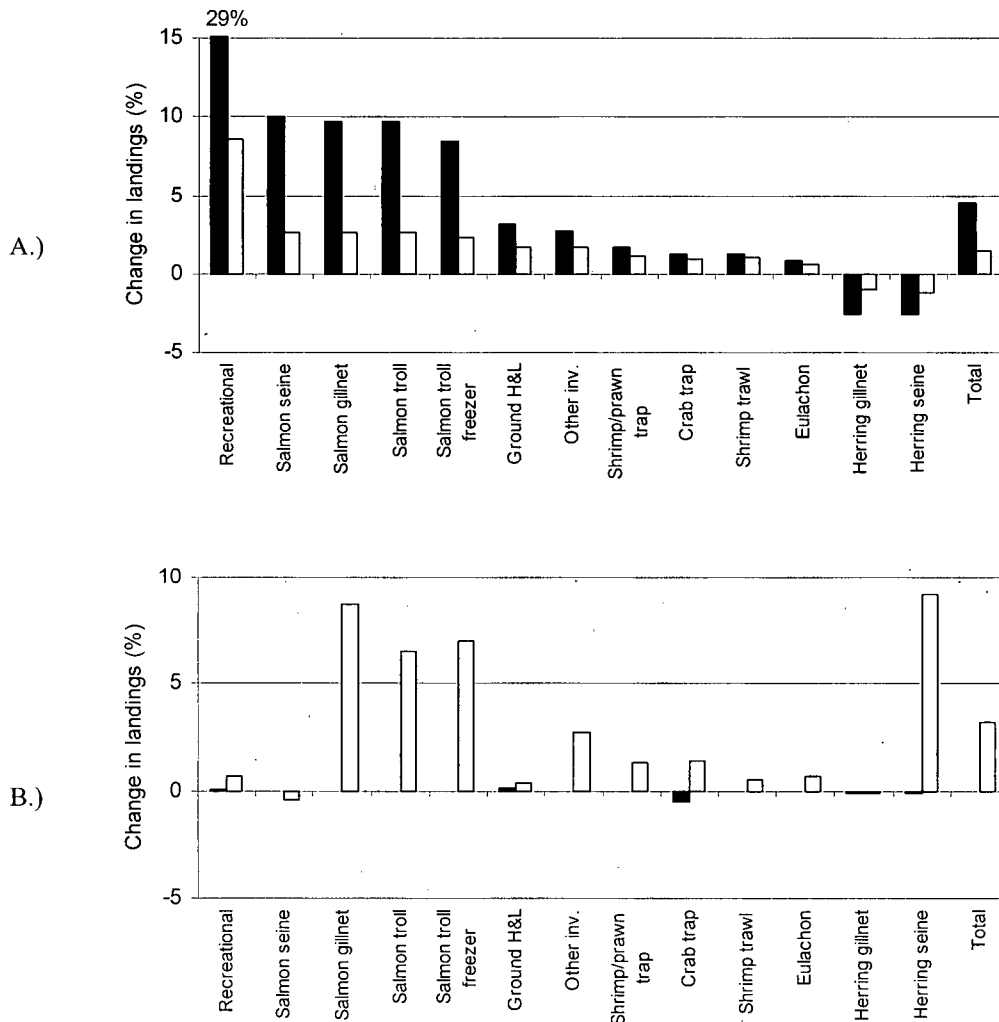


Figure A8.1.8 Regional effects of NMCA area closures on landings. Percent change over baseline (full exploitation). A.) Effect of time closure, 12 months (solid bars) and 6 months (open bars). B. Effect of gear restrictions, no groundfish fisheries (solid bars) and no commercial fisheries (open bars). Not shown: groundfish trawl, halibut hook and line, sablefish and longline. These show less than 1% change from baseline under time closures.

Local effects on trophic level of catch

Fig. A8.1.9 shows the mean trophic level of fish harvested in areas adjacent to the MPA at equilibrium (assumed after 50 years). The simulation predicts that on the oceanic side of the reserve, the mean trophic level of catch will decline under any MPA protection scheme, but the decline is mediated most by a total closure. Towards the shelf, mean TL is predicted to decrease under long-term harvests for all protection plans except a total closure. The 'no groundfish' gear exclusion scheme (not shown) manages to protect the mean trophic level of catch (equilibrium TL 3.319 in area 3; 3.357 in area 8). The 'no commercial' scheme does nothing to halt the decline in average trophic level of landings.

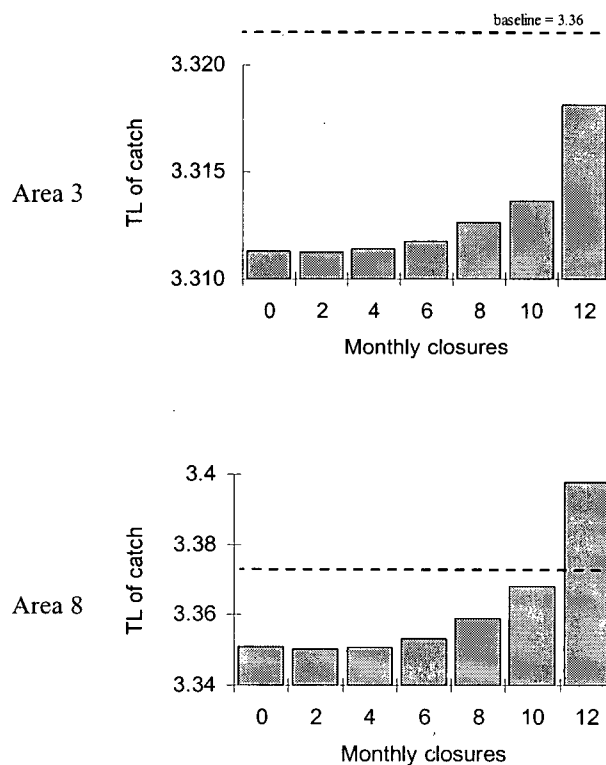


Figure A8.1.9 Equilibrium trophic level of catch in regions adjacent to MPA. Closure is maintained from 0 to 12 months per year. Broken line shows baseline trophic level. Area 3, E. Moresby; area 8, W. Moresby.

Overall, the seaward side of the MPA (area 8) sees the mean trophic level of catch vary by an order of magnitude compared to the side adjacent to Hecate Strait (area 3) under area closure schemes. This suggests that the species composition of the catch in area 8 may be more robust to management decisions regarding the MPA. From a modeling standpoint, one possible explanation is that for many of the predatory fish (e.g., salmon, halibut and rockfish) offshore areas are not described as a prime habitat (Appendix Table A8.2.1). Area 8 contains offshore cells. Since the relative dispersal rate increases in bad habitat, the model may be predicting greater circulation of these species in and out of area 8, and so changes in the species composition of the MPA are revealed in the catch of this adjacent zone.

Ecosystem impacts

Local changes in biomass

Fig. A8.1.10 shows the change in functional group abundance within the MPA under 12 and 6 month closures, and under gear closure schemes 'no groundfish' and 'no commercial'. Qualitatively, 12 and 6 month closures are similar, the greatest increase occurs in salmon groups, especially chinook (see discussion). Species that benefit the most from area closures are those that normally suffer the greatest fishing mortality. When pressure is removed, their populations bounce back and their new equilibrium biomass can thereafter sustain higher catches than are presently realized. Herring are the only species whose equilibrium abundance is expected to drop seriously with MPA protection due to additional mortality from piscivores.

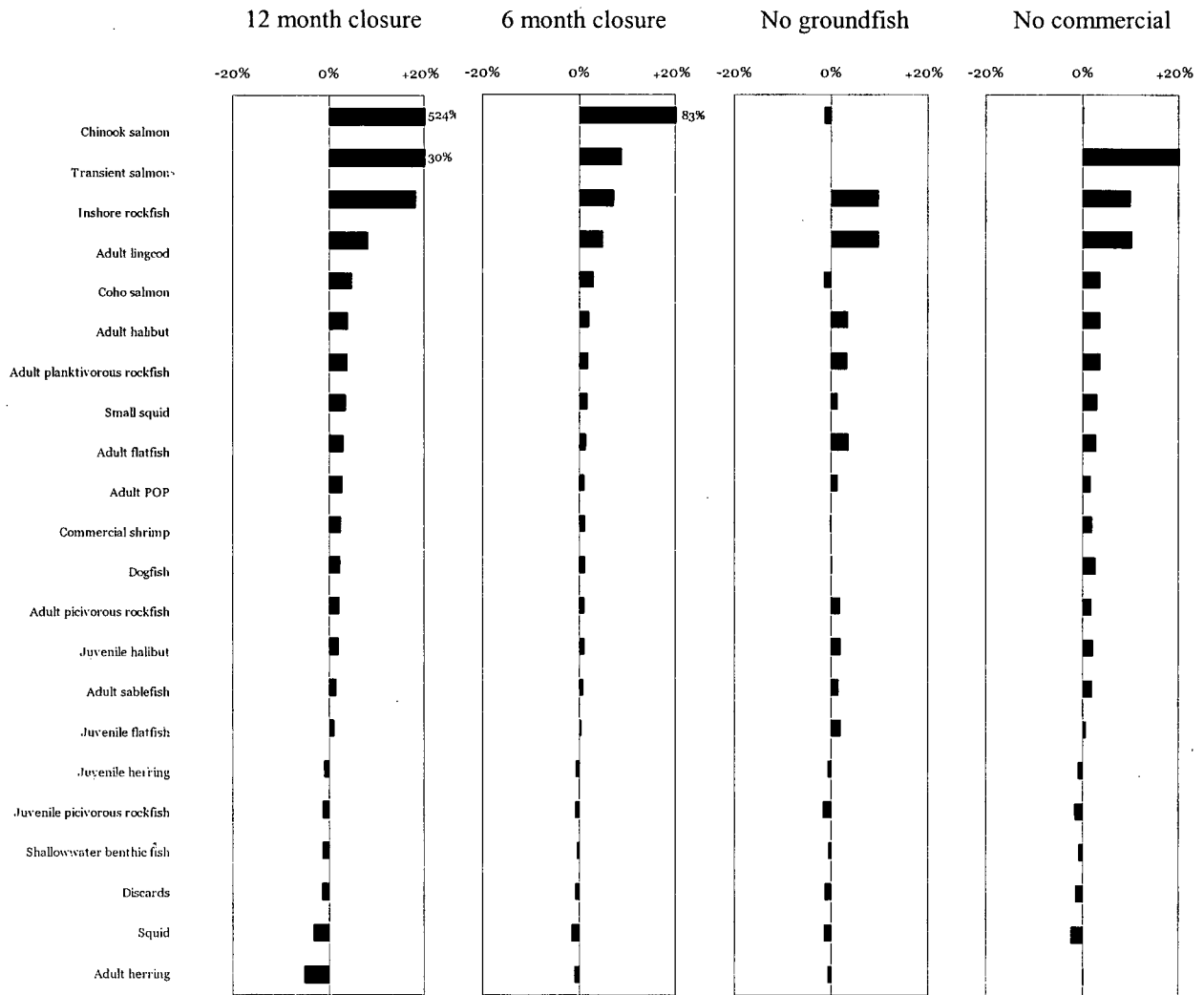


Figure A8.1.10 Group biomass change within MPA resulting from area closures. MPA includes areas 1 and 2; biomass changes >1% are shown.

The 'no groundfish' option causes little change in the species assemblage, except for a marginal increase of groundfish species, although these animals do not tend to emigrate from the reserve. The conservative 'no commercial' option has a low exploitation rate, and it clearly alters the ecosystem within the MPA. Salmon abundance is increased except for chinook, which are removed by the sport fishery. The herring population is not negatively impacted, as it is under the monthly closure schemes because the increase in predator species is not as large.

Local effects on biodiversity

Fig. A8.1.11 shows the equilibrium level of biodiversity under 0 to 12 month time closures for the MPA and adjacent regions. Biodiversity is measured using the Q-90 statistic (Chapter 2; Ainsworth and Pitcher, *in press*). Areas 3 and 8 adjacent to the reserve show a slight increase in biodiversity with area protection. The effect is more pronounced within the MPA. Area 2 sees the expected increase in biodiversity with monthly closures. Area 1 shows a drop in biodiversity under 10 and 12 month fishery closures. Salomon *et al.* (2003)

speculated that an increase in top predators might result in the local depletion of particular prey species, lowering the overall biodiversity within the protected zone.

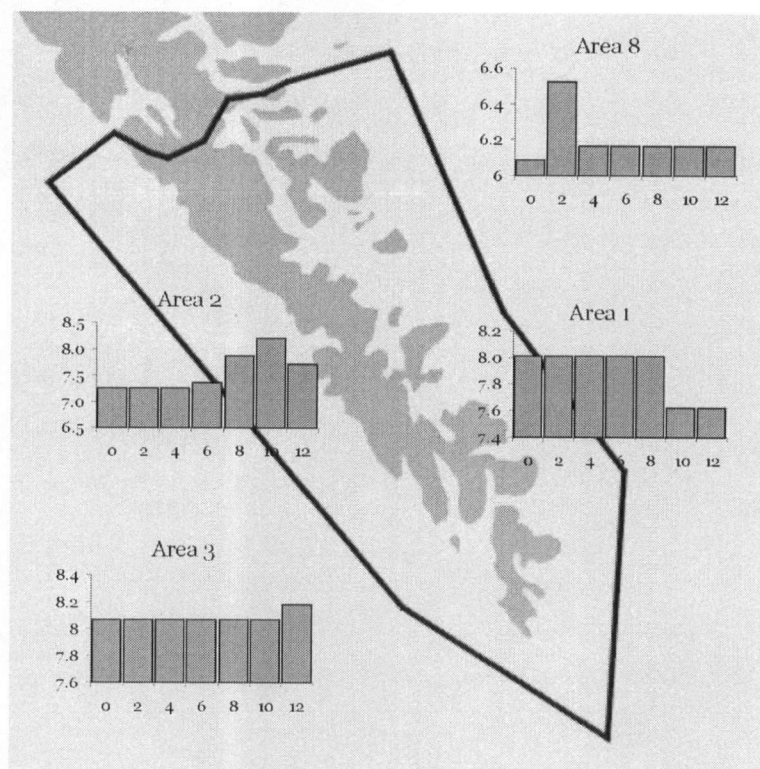


Figure A8.1.11 Equilibrium biodiversity in MPA and adjacent regions following fishery closure. Y-axes show ecosystem biodiversity (Q-90); X-axes present zero to twelve month time closures.

Ecosystem changes in the MPA

Fig. A8.1.12 shows the effects of area protection inside the reserve. Indicated are equilibrium levels of total biomass, trophic level of the assemblage and average biomass over production (B/P) ratio, a proxy for system longevity. As longer and longer time closures are applied, we see less total biomass at equilibrium, but an increase in the average trophic level. Under protection, trophic chains become longer; more production is shunted to high trophic levels and the abundance of low trophic level animals is kept in check by top-down effects.

Discussion

Improving the models

Minor modifications to the base Ecopath model may be required in order to make the results from the present investigation worthwhile as an aid to management. The Ecopath base model used for this report describes the study area as waters of the continental shelf, roughly 70,000 km² (see Chapter 1). This value was used directly in parameter calculations for group biomass densities. Water temperature, which is used for empirical calculation of basic parameters (Chapter 5), also refers to shelf waters. As long as the Northern BC model was restricted to Ecopath and Ecosim, we could describe the shelf waters implicitly, in effect accommodating an irregular shape for the study area (e.g., the western boundary following the shelf break). However, the Ecospace map used here is square, and must cover a larger area than is represented by the base EwE models, approximately 140,000 km² including offshore waters. Although inshore groups are restricted to shelf waters by habitat type, functional groups that also occur in deep water may be under-represented by EwE biomass densities. The simulations presented in this Appendix are based on EwE models done by Ainsworth *et al.* (2002). Following data fitting done in Chapter 5 of this volume, the spatial work should be repeated to verify results.

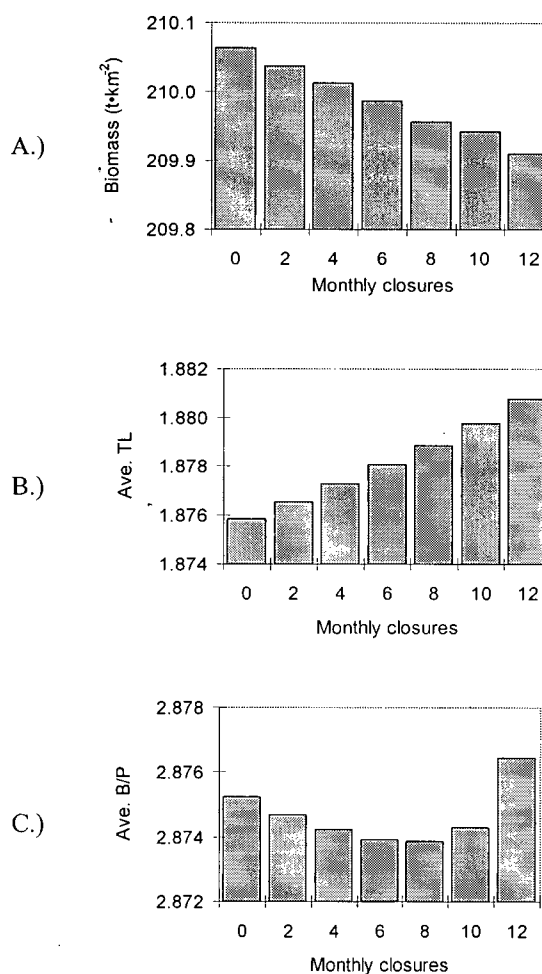


Figure A8.1.12 Equilibrium state changes within the MPA under zero to twelve month area closures. A.) Assemblage biomass. B.) Average trophic level of assemblage. C.) Average biomass/production.

Ecosim has an automated routine to tune model predictions to known time-series. However, there is no analogous procedure in Ecospace, and no previous exercise has attempted to verify spatial output. It would be possible to develop a routine that adjusts Ecospace input (most importantly the dispersal parameters), until maximum agreement is reached between model output and observed spatial data. DFO has georeferenced catch per unit effort data (CPUE) for British Columbia that could be applied as a proxy for biomass (e.g., PacHarv dataset; K. Rutherford, Fisheries and Oceans Canada, Nanaimo, *pers. comm.*). Similarly, predicted fishing effort could be compared with georeferenced effort data. In order to improve the spatial dynamics of non-commercial groups, LEK information compiled from Prince Rupert community interviews (Chapter 3; Ainsworth and Pitcher, 2005) could also be used to verify model dynamics if it were converted into a spatial index (see Fig. 3.6).

Chinook salmon

The predicted increase in chinook abundance and catch under time and gear closures may be a modeling effect. Ecotrophic efficiency tells us how much mortality in a functional group is explained by the model. Beattie (2001) accepted an EE for chinook of 0.26, indicating that 74% of chinook mortality remains unexplained in the model. However, Ainsworth *et al.* (2002) entered a negative biomass accumulation of 10% per year, to reflect real-world evidence of decline. Since Ainsworth *et al.* (2002) did not modify any other input parameters, EE increased to 0.995, which is inappropriate for chinook. To accept such a high EE implies that virtually all of the mortality of chinook salmon occurs within the study area, when in fact chinook suffer losses throughout their Pacific migration. Chinook and any migratory species may be better represented by a low EE to account for losses suffered outside the system, and so that favorable or unfavorable conditions in the study area do not have disproportionate effects on group abundance.

Spillover

Beattie (2001) found little evidence of market benefits offered by marine reserves in northern BC in terms of salmon production outside of the reserve. As the size of the MPA increased in his gaming scenarios, the system became less profitable as more area was closed to the industrial

salmon fleet. This is in contradiction with current findings, which indicate that under any level of protection there are gains to be made by the salmon fleet. Some of the discrepancy may be because Beattie assumed a more homogeneous distribution of primary productivity than was suggested by the GoMOR dataset (used here). Productivity in coastal regions, like the area covered by the MPA in this report, would be underrepresented in his calculations.

The inconsistency is also related to the trophic flow parameters. Beattie (2001) used default Ecosim flow-control vulnerabilities, which impart on the salmon populations greater resistance to fisheries impacts than were assumed in the present model. Under the comparatively top-down settings for salmon used in this report (vulnerabilities set proportional to predator trophic level, as in Ainsworth (2003), the populations are more heavily influenced by the activities of the fleet and so clearly show the benefit of effort reduction. Combined with higher dispersal rates, this makes the present model predict greater fisheries benefits outside the MPA.

Appendix 8.2 Ecospace Parameters

Table A8.2.1 Habitat occupancy

Functional Group	Habitat										Ecospace area occupied
	0	1	2	3	4	5	6	7	8	9	
Sea otters		+	+	+							21%
Mysticetae	+										100%
Odontocetae	+										100%
Seals, sea lions		+	+	+	+						40%
Seabirds	+										100%
Transient salmon		+	+	+	+	+	+	+		+	63%
Coho salmon		+	+	+	+	+	+	+		+	63%
Chinook salmon		+	+	+	+	+	+	+		+	63%
Small squid				+	+	+	+	+		+	96%
Squid				+	+	+	+	+		+	96%
Ratfish		+	+	+	+	+	+	+		+	63%
Dogfish		+	+	+	+	+	+	+		+	63%
J. pollock				+	+	+	+	+	+	+	95%
Pollock				+	+	+	+	+	+	+	96%
Forage fish	+										100%
Eulachon		+	+	+	+	+	+	+		+	63%
J. herring		+	+	+							21%
A. herring		+	+	+	+	+	+	+		+	100%
J. POP		+	+	+	+	+	+	+		+	63%
A. POP		+	+	+	+	+	+	+		+	63%
Inshore rockfish		+	+	+	+					+	41%
J. pisc. Rockfish		+	+	+							21%
A. pisc. Rockfish		+	+	+	+	+	+	+		+	63%
J. plank. rockfish		+	+	+							21%
A. plank. rockfish		+	+	+	+	+	+	+		+	63%
J. turbot		+	+	+	+						40%
A. turbot		+	+	+	+	+	+	+		+	63%
J. flatfish		+	+								4%
A. flatfish		+	+	+	+	+	+			+	58%
J. halibut		+	+	+	+						41%
A. halibut		+	+	+	+	+	+	+		+	63%
J. Pacific cod		+	+	+	+						40%
A. Pacific cod		+	+	+	+	+				+	50%
J. sablefish		+	+	+	+					+	41%
A. sablefish	+										100%
J. lingcod		+	+								4%
A. lingcod		+	+	+	+	+	+			+	58%
S. benthic fish		+	+	+							21%
Skates	+										100%
Large crabs		+	+	+							21%
Small crabs		+	+								4%
Comm. shrimp		+	+	+	+	+	+			+	58%
Epifaunal inv.	+										100%
Inf. carn. inv.	+										100%
Inf. det. inv.	+										100%
Carn. jellyfish	+										100%
Euphausiids	+										100%
Copépods	+										100%
Corals and sponges			+	+	+					+	40%
Macrophytes	+										100%
Phytoplankton	+										100%
Habitat area	100%	1%	2%	17%	19%	9%	8%	5%	37%	1%	

Table A8.2.2 Fishery activity by habitat

Fleet	Habitat									
	0	1	2	3	4	5	6	7	8	9
Groundfish trawl				+	+	+	+	+		+
Sable						+	+	+	+	+
Herring gillnet				+	+	+	+	+		+
Ground H+L				+	+	+	+	+	+	+
Salmon gillnet				+	+	+	+	+		+
Crab trap		+	+	+						
Shrimp / prawn trap		+	+	+	+					
Other Inv.		+	+	+						
Halibut H+L				+	+	+	+	+	+	+
Salmon troll				+	+	+	+	+		+
Salmon seine				+	+	+	+	+		+
Salmon troll freezer				+	+	+	+	+	+	+
Herring seine				+	+	+	+	+	+	+
Shrimp trawl				+	+	+	+			+
Eulachon		+		+	+	+				
Longline	+									
Recreational		+		+	+	+	+	+	+	+

Table A8.2.3 Salmon straying rates

Functional Group	Species	Straying rates	Reference*
Transient salmon	Sockeye	0.6-1.5%	Foerster 1936, 1968
	Pink	0.4-2.2%	Boyd 1964; Blair 1968
	Chum	2.0-46.0%	Sakano 1960, cited in Sano 1966; Harding 1981, cited in Lister <i>et al.</i> 1981
Chinook	Chinook	1.4-13%	Snyder 1931; Sholes and Hallock 1979; Quinn and Fresh 1984
Coho	Coho	0.5-27.0%	Shapovalov and Taft 1954; Donaldson and Allen 1958

*Cited in Tallman and Healey (1994)

Table A8.2.4 Dispersal parameters

Group	Base dispersal rate (km·yr ⁻¹)	Rel. dispersal in bad habitat	Rel. vuln. to predation in bad habitat	Rel. feeding rate in bad habitat	Advected?
Sea otters	300	10	2	0.01	
Mysticetae	300	5	2	0.01	
Odontocetae	300	5	2	0.01	
Seals, sea lions	300	5	2	0.01	
Seabirds	300	5	2	0.01	
Transient salmon	1000	5	4	0.05	
Coho salmon	300	5	1	0.05	
Chinook salmon	250	5	0.5	0.5	
Small squid	300	1	1	0.01	
Squid	300	1	0.5	0.05	
Ratfish	300	5	2	0.01	
Dogfish	300	5	2	0.01	
J. pollock	1000	5	5	0.01	
Pollock	1500	5	2	0.01	
Forage fish	1500	5	2	0.01	
Eulachon	300	5	2	0.01	
J. herring	300	5	5	0.01	
A. herring	300	1	1	0.01	
J. POP	300	5	5	0.01	
A. POP	200	5	2	0.01	
Inshore rockfish	200	5	2	0.01	
J. pisc. Rockfish	300	5	5	0.01	
A. pisc. Rockfish	200	5	2	0.01	
J. plank. rockfish	300	5	5	0.01	
A. plank. rockfish	200	5	2	0.01	
J. turbot	300	5	5	0.01	
A. turbot	300	5	2	0.01	
J. flatfish	300	5	0.5	0.05	
A. flatfish	300	5	2	0.01	
J. halibut	300	5	5	0.01	
A. halibut	300	5	2	0.01	
J. Pacific cod	300	5	5	0.01	
A. Pacific cod	300	5	2	0.01	
J. sablefish	460	5	5	0.01	
A. sablefish	200	5	2	0.01	
J. lingcod	1000	5	0.4	0.5	
A. lingcod	200	5	2	0.1	
S. benthic fish	300	5	2	0.01	
Skates	1500	5	2	0.01	
Large crabs	300	1	1	0.05	
Small crabs	300	5	0.5	0.02	
Comm. shrimp	300	5	2	0.01	
Epifaunal inv.	200	5	2	0.01	
Inf. carn. inv.	200	5	2	0.01	
Inf. det. inv.	200	5	2	0.01	
Carn. jellyfish	200	5	2	0.01	Y
Euphausiids	200	5	2	0.01	Y
Copepods	200	5	2	0.01	Y
Corals and sponges	200	5	50	0.01	
Macrophytes	200	5	2	0.01	
Phytoplankton	200	5	2	0.01	Y

Table A8.2.5 Ecospace output region definitions

Area ¹	Name	DFO management area
Haida Gwaii		
1	E. Gwaii Haanas NMCA	2-6 to 2-19
2	W. Gwaii Haanas NMCA	2-20 to 2-47
3	E. Moresby Island	2-2 to 2-6, 102-2, 102-3
4	E. Graham Island	2-1, 102-1
5	E. Dixon Entrance	101-4 to 101-9
6	W. Dixon Entrance	101-1 to 101-3
7	W. Graham Island	2-48 to 2-99
8	W. Moresby Island	E142*
Mainland coast		
9	N. Vancouver Island	9 to 12
10	Bella Bella	7,8
11	Douglas Channel	6
12	Banks Island	5
13	Chatham Sound	3,4,103
Shelf waters		
14	N. Hecate Strait	104, 105
15	S. Hecate Strait	106
16	N. Queen Charlotte Sound	107 to 109
17	S. Queen Charlotte Sound	110, 111
Off-shelf waters		
18	Offshore Haida Gwaii	W142 ²
19	Offshore Central BC	130+127

¹ Area numbers correspond to Fig.A8.1.6.

² Regions 7 and 17 share statistical area 142 equally

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