WHY WE HAVE TO 'OPEN THE LOST VALLEY': CRITERIA AND SIMULATIONS FOR SUSTAINABLE FISHERIES

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ABSTRACT

This paper examines why and how sustainable fisheries might be opened in a restored marine ecosystem in the 'Back to the Future' (BTF) approach, termed 'Opening the Lost Valley' (LV). A sequential list of nine criteria for designing LV fisheries includes historical gear types, conservation, community and cultural values. Sustainability is estimated by maximizing ecological, social and economic objective functions, moderated by a set of rules ensuring both sustainability and social acceptance. Pyramids of trophic flows, a surrogate diversity index and biomass profile diagrams provide comparison with present day ecosystems.

An example LV analysis is presented for the North Sea restored to its 1880 condition. Optimizing an equal balance of economic, social and ecosystem objectives results in larger fisheries than adopting ecosystem objectives alone, and larger catches entail trade-offs of conservation with depletions of some ecosystem components. Model uncertainty resides principally in 'top-down' or 'bottom-up' trophic control parameters that govern predator-prey interactions. Process uncertainty mainly lies in responses to climate change.

Imagine a restored ecosystem. All the grief and pain of fisheries being closed to get there. Then the goal is achieved and the fisheries are opened again. In the fishing ports, laid-up fishing vessels are de-rusted, repaired, gear refurbished and the fleets sets off for the first open season in many years. Naturally, huge catches are made. But this situation does not last long, and the depletions of the past are soon repeated because of the huge overcapacity of the fishing fleet (Figure 1). In an ecosystem restored to some state resembling the past under the BTF process, it is clear that we cannot use today's fleet. This paper examines a way to design sustainable fisheries to use in a restored future.

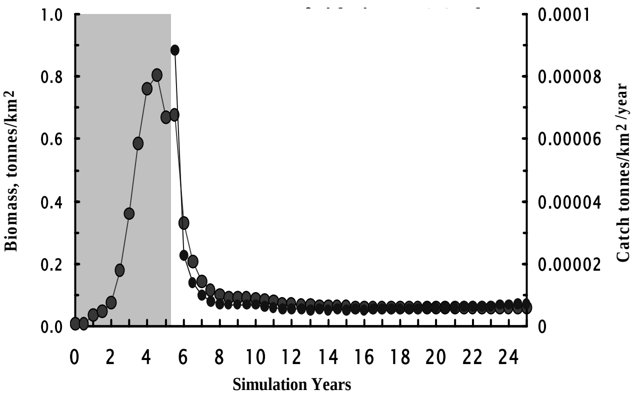

Figure 1. Biomass of one group (large reef fish; left axis) from an ecosystem simulation model of Hong Kong. Biomass recovers during a 5-year no-take period (shaded), only to be rapidly depleted when fisheries are re-opened (catch: right axis) with the former fishing fleets.

A marine ecosystem restored to some semblance of its past state might be thought of as a 'Lost Valley'[1], an ecosystem, like Arthur Conan Doyle's Lost World (Figure 2, Doyle 1912), discovered complete with all of its former diversity and abundance of creatures. This paper describes how we might achieve sustainable fishing in a restored

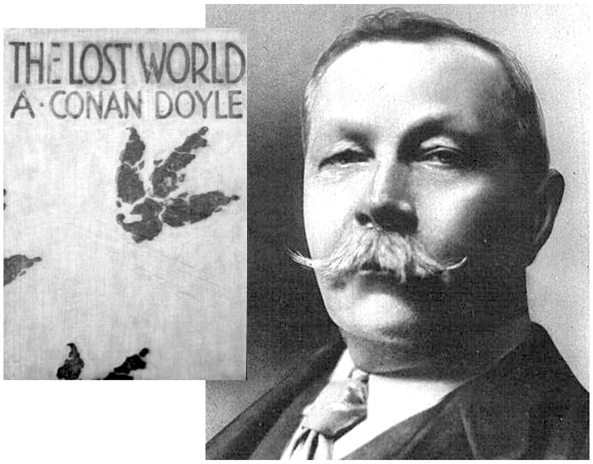

Figure 2. Cover (left) of the first 1912 edition of 'The Lost World' by Sir Arthur Conan Doyle (1859-1930, right), creator of the detective Sherlock Holmes. This book, in which explorers discover an intact ecosystem of dinosaurs from the Jurassic, was one of a series of stories about Professors Summerlee and Challenger, whose characters were based on real life Professors William Rutherford and Sir Robert Christison from Edinburgh University. Another character in the stories, Lord John Roxton, was based on Roger Casement, a British diplomat executed for treason in 1916 because he persuaded the Germans in the First World War to allow Irish nationalists to fight on their side. The 'Lost Valley' term used in BTF combines the 'Lost World' term with the title of an earlier Conan Doyle novel 'The Valley of Fear' (1911).

Pitcher, T.J. (2004) Why we have to open the lost valley: criteria and simulations for sustainable fisheries. Pages 78–86 in Pitcher, T.J. (ed.) Back to the Future: Advances in Methodology for Modelling and Evaluating Past Ecosystems as Future Policy Goals. Fisheries Centre Research Reports 12(1): 158 pp.

[1] We are grateful to Dr Daniel Pauly for suggesting this term in 2001. Although I think the Conan Doyle reference is the most appropriate, 'Lost Valley' is also the title of a Max Brand cowboy novel from the 1950s, and is now the name of several remote ski and dude ranch resorts in USA.

'Lost Valley' by applying a set of objective criteria to design an 'ideal fishery' for a particular location and then using ecosystem simulations to find the relative fishing mortalities that should be used by each fishery to achieve sustainable catches over a long time period, usually 100 years. This is termed 'Opening the Lost Valley' and forms stage 2 of the BTF procedure (see Pitcher 2004, this volume). The BTF process aims to describe a series of such fished 'Lost Valleys'. In addition, we may seek to challenge these results with climate changes that might realistically be expected for the locality in question, and in the face of uncertainty in the simulation modelling as described in Pitcher and Forrest (2004, this volume). Basic whole-ecosystem modelling techniques employed in the BTF process are not described further here. A complete account of the 'Opening the Lost Valley' procedure appears in Pitcher *et al.* (2004) and an example applied to models used in the CUS project in Pitcher (2002a) in Ainsworth *et al.* (2004).

Choosing a portfolio of responsible and sustainable fisheries is a three-stage process. Fisheries are chosen according to a rational list of criteria. Secondly, the species (and hence model groups) caught by each fishing gear are chosen. Finally, once fisheries and their target species and likely by-catch are chosen, their relative intensity can be determined using the policy search optimization interface in *Ecosim* (Christensen and Walters 2004, Walters *et al.* 2002).

CHOOSING SUSTAINABLE FISHERIES

It is not realistic to expect the fishing gear and methods of former times, including those of ancient aboriginal fisheries, to be re-employed. Of course, some former fisheries might have attractively low by-catch, operating costs or ease of construction and use, so it is evident that some rational criteria for the selection and operation of sustainable fisheries need to be devised.

Criteria devised in the BTF project for designing sustainable fisheries in a restored 'Lost Valley' are listed in Table 1. Many of the items are similar to those set out in the FAO Code of Conduct (FAO 1995), but the overall list is much shorter than that document as a result of combining many issues and avoiding repetition. These criteria are meant to be applied sequentially and with the participation of stakeholders. Ideally, the new fisheries are intended to be newly-designed and the gear and vessels equipped with the latest selectively and efficiency devices. Since this ideal may be costly or unacceptable to the fishers, in practice, older vessels and gear may be re-commissioned or brought in from elsewhere. Hence, the list of criteria will have to have to be interpreted and adapted in a particular case provided that the overall aim of creating new fisheries that are genuinely sustainable is not lost sight of.

1. Minimal by-catch discards. Over the past ten years, trawl, trap and purse seine fisheries have demonstrated large improvements through the use of separators and gates (Kennelly and Broadhurst 2002) or through altering fishing practices (e.g. dolphins released in tuna purse seine fisheries, Hall 1988). It is therefore reasonable to assume that technological advances may be successful in greatly reducing unintended catches of non-target species. Moreover, in some jurisdictions such as Norway and Iceland, discards have become illegal.

2. No damage to habitat. Bottom trawls and dredges have long been suspected of doing great harm to sessile benthic invertebrates (e.g., sponges, cold water corals, gorgonids) that act as refuges for the juveniles of many commercial fish species (Hall 1999). We assume here that, in Lost Valley fisheries, technological improvements will minimise damage by trawls – for example by only permitting trawls that fish above the bottom. Where some collateral damage to benthos is inevitable, such as in prawn trawls, we have assumed 10-fold reductions in damage are

Table 1. List of nine criteria for sustainable and responsible fisheries to be opened in a restored ecosystem. For a full discussion see text (From Pitcher 2004, and modified from Pitcher *et al.* 2004).

#	**Criteria for sustainable fisheries**	**Notes**
1	Minimal by-catch discards	Technological modifications to gear
2	No damage to habitat by gear	Technological modifications to gear
3	Include Aboriginal fisheries	Customary rights recognized
4	Include traditional target species	Except where #1 and #2 would bar
5	Minimise risk to charismatic species	Except as under #3 and #7
6	Exclude 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)

possible.

3. *Include aboriginal fisheries*. Some fisheries by indigenous or aboriginal peoples were sustainable over thousands of years (e.g., salmon and halibut in the Pacific Northwest). In terms of equity we believe they should be included in the Lost Valley fisheries portfolio, provided the take is sustainable, and where such customary rights are recognised.

4. Include traditional target species. Provided criteria 1 and 2 above are satisfied, this category is included because there will be an understandable demand for traditional desirable fish species in local fishing communities. For example, even if the historic Atlantic halibut fishery has not proven sustainable, the species would be in demand as a target in a restored ecosystem.

5. Minimise risk to charismatic species. Whilst it is evident from the recorded history of seabirds, whales, seals and sirenians that many 'charismatic' species are sensitive to exploitation by humans (e.g., Roman and Palumbi 2003), this criterion may well be in conflict with #3 and #4 above, since coastal peoples traditionally exploited seals, sea lions, whales, dugongs, turtles, ducks, gulls, petrels, auks and other seabirds. (e.g., Australia: Williams and Baines 1993, British Columbia: Brown *et al.* 1997). Where customary rights are recognised, an aboriginal take of these species would be allowed under criterion 3, with appropriate consent under criterion 7 below. On the other hand, many marine mammal, bird and shark species have recently become 'charismatic' to the conservation movement, and legal bans on killing them reflect public revulsion at their use for human food. But these views are volatile and local, so in the last resort, the choice of whether to exploit these types of animals will be locally or nationally determined. The only rational criterion is avoidance of excessive depletion and minimal risk of extirpation.

6. Exclude fishing on juvenile groups. Generally, heavy fishing on juveniles leads to recruitment failure, so such fisheries would not normally be allowed in opening a 'Lost Valley'. In some cases traditional fisheries (criterion 4) include eggs, fry and juveniles of highly fecund species such as herring, anchovy, sardines, milkfish or hake, so such fisheries would be permissible where impacts can be proven to be minimal.

7. Participatory vetting of fisheries. To retain support, the local fishing community has to vet and approve the list of fisheries. In addition, the management agency must be convinced that management and monitoring (criterion 9) are feasible for the chosen fisheries, and that the scientific basis of the 'Lost Valley' forecasting (criterion 8) represents best practice.

8. Simulations show fisheries are sustainable. Assessments must show that, given constant environmental conditions, the biomass of the main ecosystem groups, biodiversity, and the fishery catches themselves are sustainable and do not fluctuate more than a predetermined and agreed amount over a 100-year period. A tougher criterion would be that they are robust against climate fluctuations and uncertainty on that time scale to a specified level of risk (see Pitcher and Forrest 2004, this volume).

9. Adaptive monitoring plan is in place. Because environmental changes (climate, pollution) and our ignorance of fundamental ecology always lead to the unexpected in natural ecosystems, it would be prudent for the restored 'Lost Valley' and its fisheries to be subject to regular monitoring of the indices from criterion 8. This would allow passive adaptive shifts in fishing according to circumstances, much as the way catch quotas and fishing locations are regulated today.

The complete portfolio of fisheries designed for a specific LV ecosystem will depend to a large extent on markets and local tradition. Before a final choice is made, modelling could consider a range of target species of fish and shellfish. And the scope of the new sustainable fisheries would certainly the subject of much debate in the local fishing community. As yet there have been no rigorous comparisons of the effect on ecosystems and sustainability among different fishery portfolio strategies. At one extreme, typical perhaps of aboriginal communities, a broad spectrum of harvested seafood is consumed locally, while at the other extreme, a small number of targeted fish species produce large catches suitable for processing and export.

Compliance of candidate LV fisheries with the listed criteria can be evaluated using a rapid appraisal technique such as *Rapfish*, which has already been applied to compliance with the FAO Code of Conduct (Pitcher 1999).

Even after a fishery design based on the listed criteria is adopted, management mistakes in the form of unacceptable depletions and species losses may well still occur. Two items in the list can help recovery after this unfortunate situation. Criterion 8, simulation modeling, may pick up many potential problems. Criterion 9, passive adaptive management, should eventually identify problems not captured by the modelling. In practice, neither of these fall-backs are perfect, and #9 may not operate fast enough to deal with pollution events or rapid climate shifts for example. Nevertheless, they are included as intended 'fail-safe' mechanisms in the BTF management procedure.

Species caught in opened 'Lost Valley' fisheries

For each fishery in the portfolio, designed according to the criteria above, the species targeted by the gear are determined and related to the ecosystem model groups. In addition, probable by-catch that cannot be avoided by improvements in gear technology (#1) is identified by species and likely percentage amount in relation to catches of the target species.

Initial catches, transformed to tonnes per km2, are entered into the fishery parameter input tables in *Ecopath*. Starting values of 1% and 2.5% of unfished biomass have been used for the optimality simulations in *Ecosim*, but both of these values is a little low for the way that the software is presently written. No systematic analysis of the effect of varying this starting value has yet been performed. Any discarded by-catches, along with ex-vessel prices by species and gear, and operating costs by gear, are also entered in the tables in proportion to the target species.

At this point, the basic parameters of the underlying *Ecopath* model have to be readjusted slightly to achieve mass-balance. For replicability, this was performed with an automated search procedure (adjustments to mortality rates and diet, Kavanagh *et al.* 2004).

SEARCHES FOR OPTIMAL 'LOST VALLEY' FISHERIES

After an 'ideal' set of fisheries and its catches have been selected according to the procedure discussed above, simulations are used to forecast fishing and its effect over a long time period, typically 50 or 100 years (criterion 8). Relative fishing mortalities over the set of fisheries are adjusted from small starting values (see above) until catches are sustainable and impacts on the ecosystem meet specified criteria. In *Ecosim*, the adjustments are carried out automatically using an automated search routine that seeks to maximize a specified objective function using a multi-dimensional Davidon-Fletcher-Powell search algorithm (Christensen and Walters 2004, Walters *et al.* 2002). The search iteratively varies the fishing mortality per gear type to maximize an objective function over the simulated time horizon, usually 50 or 100 years.

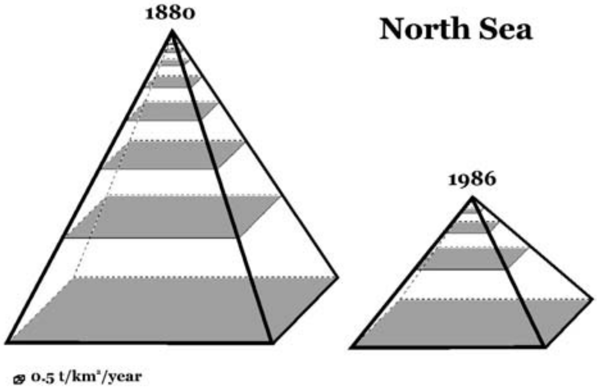

Figure 3. Flow pyramid for the North Sea in Mackinson's 1880 *Ecopath* model, and for a model representing 1981 (Christensen 1995). Horizontal 'floors' represent adjacent trophic levels and distance between floors the relative flow between them. Height of pyramid represents relative length of food chains. Pyramids approximately to same scale, redrawn from *Ecopath* outputs. Note the much smaller flow pyramid and considerably fewer trophic levels (horizontal slices) in the recent model.

Alternative fishery objectives may be selected, including economic value, numbers of jobs, the biomass of long lived species, a log portfolio utility function (Cochrane 2002). Combinations of these policy goals may also be attempted A range of policy options can be used: maximising ecological objectives alone; maximising ecological objectives roughly balanced with employment; and maximising ecology, employment and economics roughly equally balanced. In practice, many searches have to be performed to reduce the chances of finding a local optimum.

The results of the search provide forecast fishery catches, biomass, economic values, numbers of jobs, and biomass changes in all other groups in the fished 'Lost Valley' ecosystem. Results are examined and any scenarios that cause extirpation, or severe depletion of species, are eliminated. In fact, the biomass of designated species may be protected from large changes in biomass as part of the policy search objective function (Cochrane 2002). Adjustments to the weightings in the objective function enable (after some iteration) policies that attempt to balance economic with ecological or social values. This search procedure is repeated for a wide range of policy objectives and for each candidate restored ecosystem, producing a number of forecast scenarios that may be compared.

Table 2. Fisheries selected for North Sea '1880 fished Lost Valley' marine ecosystem simulations. Fisheries were assumed 'clean' of discards due to improved technology. Initial values for the policy search modelling were set at 2.5% of the 'Lost Valley' biomass. Jobs per unit of effort, modified from Mackinson (2002), are required for job optimizations. P = species groups protected from extirpation using the 'mandated rebuilding' option.

Fishery	**Landed Species**	**Relative jobs per unit of catch**
Herring	herring	7
Small mixed fish	hake, angler, conger, tusk, ling, redfish, gurnards P, John Dory, blue whiting	7
Salmon	Atlantic salmon, sea trout	5.75
Crabs & lobsters	edible crab P, lobster P	1.5
Tuna	bluefin tuna P	7
Gadoids	cod, haddock, whiting	4.5
Small flatfish	plaice, sole, brill	4.5
Large flatfish	halibut P, turbot P	4.5
Saithe	saithe	4.5
Mackerel	North Sea & western mackerel stocks	5.75
Sprat	sprat	5.75
Not caught	Other prey fish P, other small predatory fish P, rays and skates P	

When running he *Ecosim* policy optimisation method, the underlying n-year dynamic ecosystem model is continuously in operation in the background. This means that ecological parameters inimical to heavy fishing such as long life, low fecundity, slow growth, or reliance on volatile or high trophic level prey, will automatically reduce catches of charismatic species or traditional slow-growing target species to very low values. The different policy objectives available in *Ecosim* mean that a number of different optimisations can be compared. In practice, we have found that runs aiming to optimise only jobs or economics often resulted in unacceptably large (>90%) depletion of some biomasses. Hence, it may normally be best to use three policy options: ecological objectives alone; ecology equally balanced with employment; and ecology, employment and economics equally balanced.

EXAMPLE 'LOST VALLEY' ECOSYSTEM: THE NORTH SEA AS IT WAS IN 1880

The example LV analysis here is based on a published 46-group *Ecopath* model describing the North Sea as it was in 1880 prior to the expansion of steam trawlers (Table 2: Mackinson 2001, see also Pitcher *et al.* 2004). Mackinson describes how historical archives, catch and survey data, and interviews with experts were used to construct this model.

Table 2 shows a portfolio of 11 fisheries set up on the basis of the criteria in Table 1. Relative employment values per fishery were modified from Mackinson (2002). 'Lost Valley' fisheries were assumed to be clean of discards as a result of improved technology. Table 2 also shows seven species groups 'protected' from extirpation in the simulations using the 'mandated rebuilding' option. Weightings applied to the objective functions to achieve equalize three policy goals (ecological goals, an equal balance of ecology with employment, and an equal three-way-mix goal of ecology, employment and economics) are shown

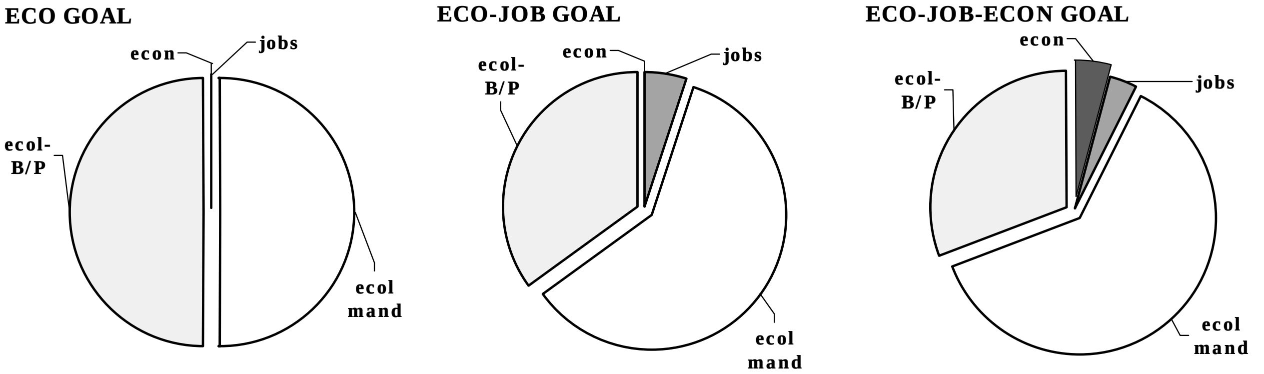

Figure 4. Pie diagrams illustrating relative weightings of conservation (ecol, B/P ratio), mandated rebuilding (ecol mand), employment (jobs) and economic (econ) goals in the optimal fishery searches for the North Sea 1880 ecosystem. Initial figures output by the software for each goal are arbitrary and depend on the units and values chosen as input: hence weightings used in the optimisation are adjusted iteratively so that each of the chosen goals enter equally into the overall objective function. (Weightings are further discussed in Aisworth 2004, this volume.)

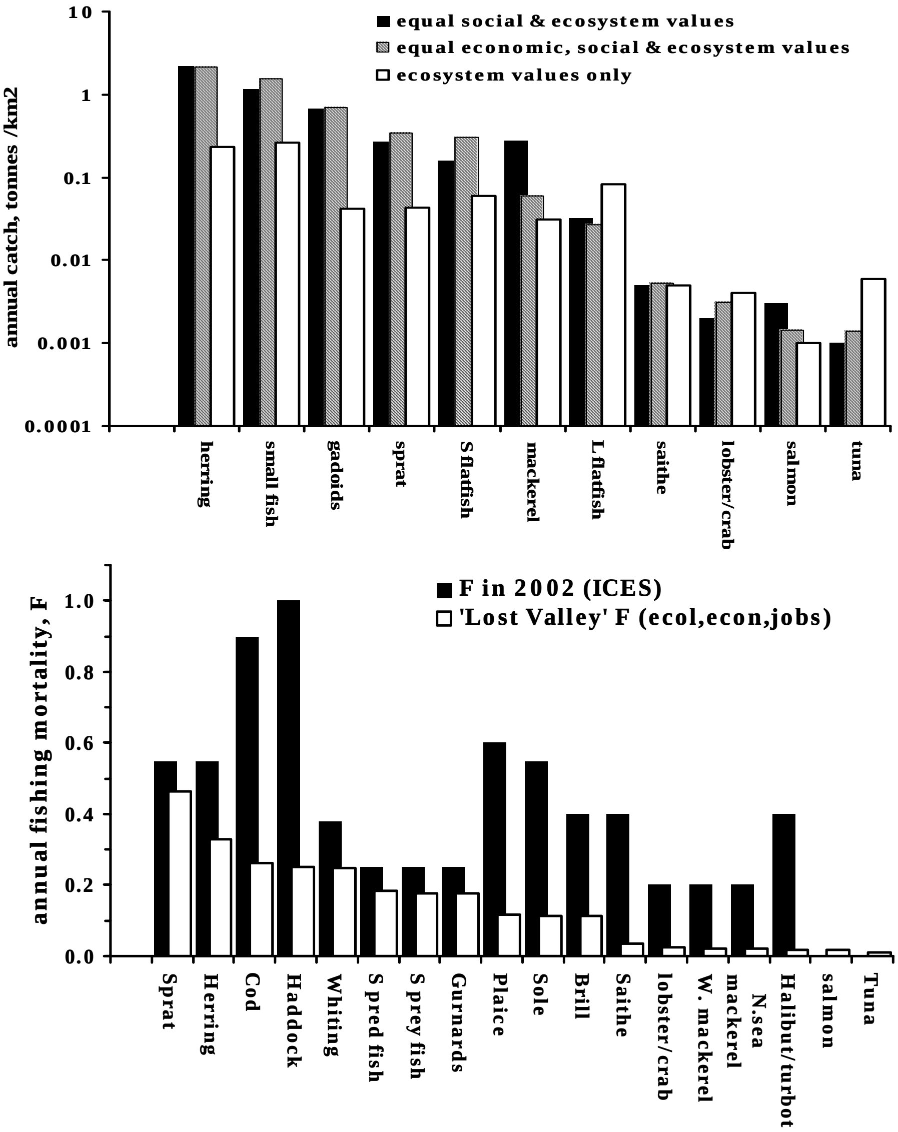

Figure 5. *Upper panel*: Sustainable North Sea 'Lost Valley' fisheries operating in North Sea ecosystem restored to the state of the 1880s. Annual catch rates are shown on a log scale. Dark bars show catches when ecosystem and social objectives are equal, striped bars when a thee-way objective is optimised, light bars show ecosystem objective for function optimisations. *Lower panel:* Light bars show fishing mortalities of modelled groups for equal ecosystem/social/economic policy objectives. Dark bars show approximate fishing mortalities for these groups in 2002 (ICES).

in Figure 4. In all, over 150 simulations were performed, each starting from random values of F. Alternative solutions found by the software were accepted or rejected using the constraints discussed above.

Figure 4 (top) shows the sustainable catch for ecosystem objectives (total catch; around 0.8 tonnes per km^2 per year), for equal ecosystem and employment objectives (catch; 4.8 tonnes per km^2 per year) and the three-way-mix objective (catch; 5.1 tonnes per km^2 per year). For all objectives, the largest fisheries, producing around 70% of the total catch, are for herring, small fish and gadoids, although the large flatfish fishery is third instead of seventh largest (11%) for the pure ecological goal. Fisheries under the two- and three-way mix goals are quite similar. The largest difference is for the mackerel fishery, which is almost ten times larger under the ecology/social goal. Catches in the 'ecosystem alone' fishery are considerably lower, as in the Newfoundland example. This objective reduces the top six fisheries by about 15% compared to the 2- and 3-way mix, while flatfish, lobster and tuna fisheries are about twice as large. The saithe fishery remains about the same for all objectives.

Figure 5 (bottom) plots sustainable fishing mortalities of the main fished groups for the 3-way-mix objective, compared to 2002 estimates of fishing mortality from ICES. While sprat is similar, herring, whiting, and small fish 'Lost Valley' fisheries have fishing mortalities only 30% less than today's value. We note that cod, haddock, plaice, sole, saithe, and both mackerel fishing mortalities are on average 6-fold greater today than our LV simulations suggest is sustainable.

Currently, North Sea cod (Cook *et al.* 1997), plaice, saithe and haddock (ACFM 2002) are heavily depleted and the biomass of several other stocks is not healthy. The LV restored system could clearly support a modest North Sea fishing industry sustainable over long periods, while maintaining reasonable biodiversity and balance. But there are trade-offs in fishing the 'Lost Valley'. Compared to the basic 1880 LV ecosystem, our Lost Valley fisheries reduce 7 biomasses (herring, sprat, horse mackerel, cod, brill, gurnards, seabirds) by more than 25%, but only one (tuna) by more than 50%. Compared to 1880 LV, 19 groups have been reduced in biomass by more than 75%.

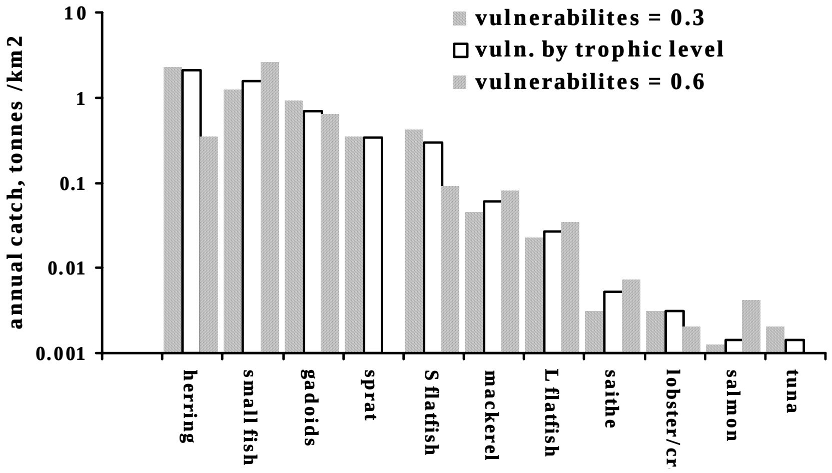

Figure 6. Effect of *Ecosim* predator/prey vulnerabilities on the catch results for the North Sea 'Lost Valley' ecosystem under the 3-way-mix objective. Open bars: vulnerabilities proportional to trophic level. Left shaded bars: vulnerabilities all set to 0.6, a 'top-down' ecosystem. Right shaded bars: vulnerabilities all set to 0.3, a 'donor-control' system.

Figure 6 shows the effect of altering the *Ecosim* predator/prey vulnerabilities. The 3-way-mix objective was used for this comparison. The simulations with vulnerability proportional to trophic level, as above, were compared with a 'top down' system, where v = 0.6, and a 'bottom up' system, where v = 0.3. For the three largest fisheries, changes are relatively minor except for reducing the herring fishery under the 'top down' option by 84%. With one exception (sprat), the LV fisheries remain in the same order of magnitude under all v assumptions. Both sprat and tuna have almost no LV fisheries under the 'top down' option. The direction in which the vulnerability assumption changes the fisheries appears does not appear to be obvious: small fish, mackerel, large flatfish, saithe, and salmon have higher LV fisheries under the top down assumption, while herring, gadoids, small flatfish, and lobster have smaller ones.

CONCLUSIONS

The results presented here are preliminary. Ecosystem simulations like these tend to reveal the superficiality of our understanding of natural aquatic ecosystems and relatively simple ecological processes. The underlying ecosystem models can always be corrected and refined. Actual use of such models has to be tempered with feedback from adaptive management policies. Pitcher (2002b) warns of undue reliance on modelling without such feedback from the real world. Note that it is not suggested that the results reported here provide a realistic goal for current North Sea fisheries. Not only are the *Ecopath* models of past states preliminary, but also there has been no participatory vetting of the LV fisheries and a number of uncertainties have not yet been addressed. However, the example serves to illustrate what may be done with the 'Lost Valley' process.

Changing the vulnerability parameters in the North Sea 1880 LV model had a smaller effect on the overall fishery results than might have been anticipated, although two out of eleven LV fisheries showed large, and three fisheries exhibited moderate differences when vulnerabilities were set to extreme values. To reduce this uncertainly, much more research is needed to obtain parameter values characteristic of each predator-prey interaction.

Using the ecological objective alone in the search routine produces the most sustainable set of LV fisheries, but with smaller annual yields compared to the present day. Using the social or economic objectives alone tends to produce a small number of large fisheries, or instability in the model, because the search engine tends to create jobs or profit by expanding gear sectors with little consideration for distributing catches among the fleets, so long as there is some catch remaining at the end of the 50-year simulation run. Hence, an attempt to emulate real policy choices using equally balanced social, ecosystem and economic objectives is presented. Even then, it is not possible to rely on the software alone to produce biomass trajectories and fisheries that might satisfy the sustainability and social acceptance criteria of a real policy maker. Hence the use of a set of rules to accept or reject solutions offered by the optimisation routine. Fortunately, there were a number of peaks of similar height in the likelihood surface among which one could choose. The overall finding, which is not surprising, is that truely sustainable fisheries in restored ecosystems will very likely produce much smaller yields than those seen during the recent age of fishery expansions (Pauly *et al.* 2002).

The LV fishery solutions confirm that there will always be a trade-off between sustainable fisheries and biodiversity. However, the full LV process presented in this paper ensures that fisheries are sustainable, accepted by local fishing communities, and monitored against unexpected events or incorrect science.

It may be argued that 'Opening the Lost Valley' is unrealistic, because, as yet, it has not been worked out exactly how restoration might be achieved. Focussing on a long-term policy goal, and the benefits that will accrue from its attainment is essential, because it deflects attention from the present-day allocation wars that continually prejudice any attempts at restoration. In parallel with such work in terrestrial environments (e.g., Sinclair *et al.* 1995), restoration of past abundance may require habitat zoning with a mix of reduced fisheries, no-take zones and, perhaps, more proactive management, such as reintroductions of locally extinct species. In addition, the 'Lost Valley' simulations need to be made robust against climate change (see Pitcher and Forrest 2o04, this volume).

This paper does not describe how one might choose amongst alternative "Lost Valley" restoration goals. That choice requires ecological, social and economic criteria. A preliminary approach is discussed in Pitcher (2004), in Ainsworth et al., and Sumaila (2004, this volume), in Sumaila *et al.* (2001) and Pitcher *et al.* (1999). Some case studies for the *Coasts Under Stress* BTF project are currently in progress.

In the face of the disaster witnessed in fisheries over the past 50 years (Pitcher 2001; Pauly *et al.* 2002), only a radical solution stands a chance of succeeding. Many have begun to adopt rebuilding goals. The concepts of 'Back to the Future' and 'Opening the Lost Valley' have a resonance that may serve to guide recovery, and recapture both the biodiversity and wealth that may be provided by healthy marine ecosystems. The 'Lost Valley' reconstruction of whole marine ecosystems to the point where a suite of sustainable fisheries may be chosen provides a set of clear policy goals against which progress can be measured quantitatively. Rebuilding to the state of a 'Lost Valley' is a process that benefits both conservation and fisheries (Pitcher 2002b). Moreover, the 'Back to the Future' approach is in accord with Aldo Leopold's Land Ethic (Leopold 1933, 1949) which states that:

> "A thing is right when it tends to preserve the integrity, stability and beauty of the biotic community."

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EVALUATING THE ECOLOGICAL EFFECTS ON EXPLOITED ECOSYSTEMS USING INFORMATION THEORY

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ABSTRACT

The ecological effects of exploitation on the eastern Newfoundland and southeastern Labrador ecosystem (NAFO Div. 2J3KLNO) were evaluated using information theory. The 1900 model of this ecosystem was subjected to two different scenarios: 1) an increase in fishing mortality of 1% per year for 100 years, or 2) removing fishing from the system for 100 years. The effect of different vulnerability settings on the outcome of these two scenarios was also tested by assuming that the vulnerability of each prey was related to its trophic level, or alternatively the vulnerabilities were kept at the baseline of 0.3. The results show that removing the fishing mortality increase the resilience of the system to an asymptote, while an increase in fishing mortality cause the system to become less resilient over time, until the system becomes unstable after which the resilience increase again. The different vulnerability settings have an effect on the crash of the system in the fishing scenario and on the reduction of some species to very low biomasses in the no-fishing scenario, but does not effect the overall outcome of the resilience.

INTRODUCTION

Information theory gives us a way to measure the emergent properties of an ecosystem. According to Ulanowicz (1997) it “quantifies changes in probability assignment, in the same way that differential calculus quantifies changes in algebraic quantities and “information” refers to the effects of that which imparts order and pattern to a system”.

From information theory comes the hypothesis that as a system becomes more specialized its ascendancy would increase, but it loses its “strength in reserve” or resilience (Ulanowicz 1986). The ascendancy measures the size and organizational status of the network of exchanges that occur in an ecosystem (Ulanowicz 1999) and the resilience of a system is defined as its probability of recovery after perturbation (Mageau *et al.* 1998). In this paper the assumption is that removing a stressor such as fishing from a ecosystem would increase its resilience, but decrease its specialization, while a constant increase in fishing mortality would reduce its resilience, but increase its specialization. It is therefore hypothesized that the information theory proxy for resilience (the system’s overhead, or the compliment to its ascendancy) would thus increase if fishing was removed, and decrease if fishing presume increased.

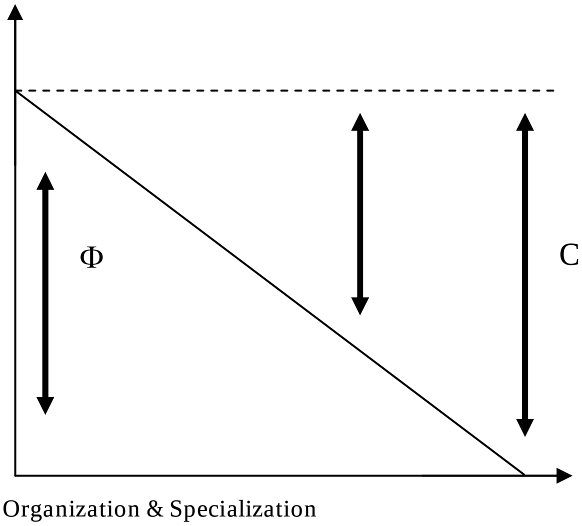

Figure 1. The change in information (on the y axis) as the organization and specialization of the ecosystem increase. The ascendancy (A) increases while the overhead (Φ) decreases. The upper limit to the ascendancy is the development capacity (C).

METHODOLOGY

Information theory

Ulanowicz’s (1986, 1997) theory of ascendancy derives from information theory and is illustrated in Figure 1. As the system becomes more specialized and organized its ascendancy (A) increases, with the upper bound of the ascendancy being the development capacity (C). However, as the system becomes more specialized, it looses overhead (Φ). This is a phenomenon similar to “putting all your eggs in one basket”.

The disorder or freedom of the ecosystem is defined as the overhead. It is complimentary to ascendancy and calculated by (Ulanowicz 2000) as:

$$\Phi = C - A \qquad (1)$$

Heymans, J.J. (2004) Evaluating the Ecological Effects on Exploited Ecosystems using Information Theory, Pages 87–90 in Pitcher, T.J. (ed.) Back to the Future: Advances in Methodology for Modelling and Evaluating Past Ecosystems as Future Policy Goals. Fisheries Centre Research Reports 12(1): 158 pp.

The development capacity (C) is calculated as:

$$C = TST * H \tag{2}$$

where TST is the total systems throughput and H is the systems entropy. The TST is calculated (Mageau *et al.* 1998) as:

$$TST = \sum T_{ij} \tag{3}$$

and systems entropy (H) is calculated as (Mageau *et al.* 1998):

$$H = \sum_{ij} \frac{T_{ij}}{TST} * \log\left(\frac{T_{ij}}{TST}\right) \tag{4}$$

Finally, ascendancy is calculated as (Ulanowicz, *pers. comm.*):

$$A = \sum_{i,j} T_{ij} \log\left(\frac{T_{ij} B_{.}^{2}}{T_{..} B_i B_j}\right) \tag{5}$$

where B_i is the biomass of component i, and a dot as a subscript means that the index has been summed over i.e.,

$$T_{..} = \sum_{i,j} T_{ij} \quad \text{and} \quad B_{.} = \sum_{j} B_j \tag{6}$$

The *Ecopath* software (2003) still uses formulas of H and A that exclude the biomass, thus entropy (H) is calculated as:

$$H = \sum_{i=1}^{n} Q_i \log Q_i \tag{7}$$

where Q_i is the probability that a unit of energy passes through i, or

$$Q_i = \sum_{k=1}^{n} T_{ki} / \sum_{l=1,m=1}^{n} T_{lm} \tag{8}$$

Ascendancy in *Ecopath* is therefore still defined in terms of flow only, or:

$$A = T * \sum_{i,j} \left(\frac{T_{ij}}{T}\right) \log\left(\frac{T_{ij}T}{\sum_{k} T_{kj} \sum_{q} T_{iq}}\right) \tag{9}$$

The application

The ratio of overhead to development capacity has been linked to the resilience of the system (Ulanowicz 1997, Ulanowicz 1980, Ulanowicz and Norden 1990). The resilience of a system is defined as its probability of recovery after perturbation, while biodiversity stabilizes community and ecosystem processes, but not population processes (Tilman *et al.* 1996, referred to in Mageau *et al.* 1998).

To test this hypothesis of the overhead linked to the resilience of the system, an ecosystem model

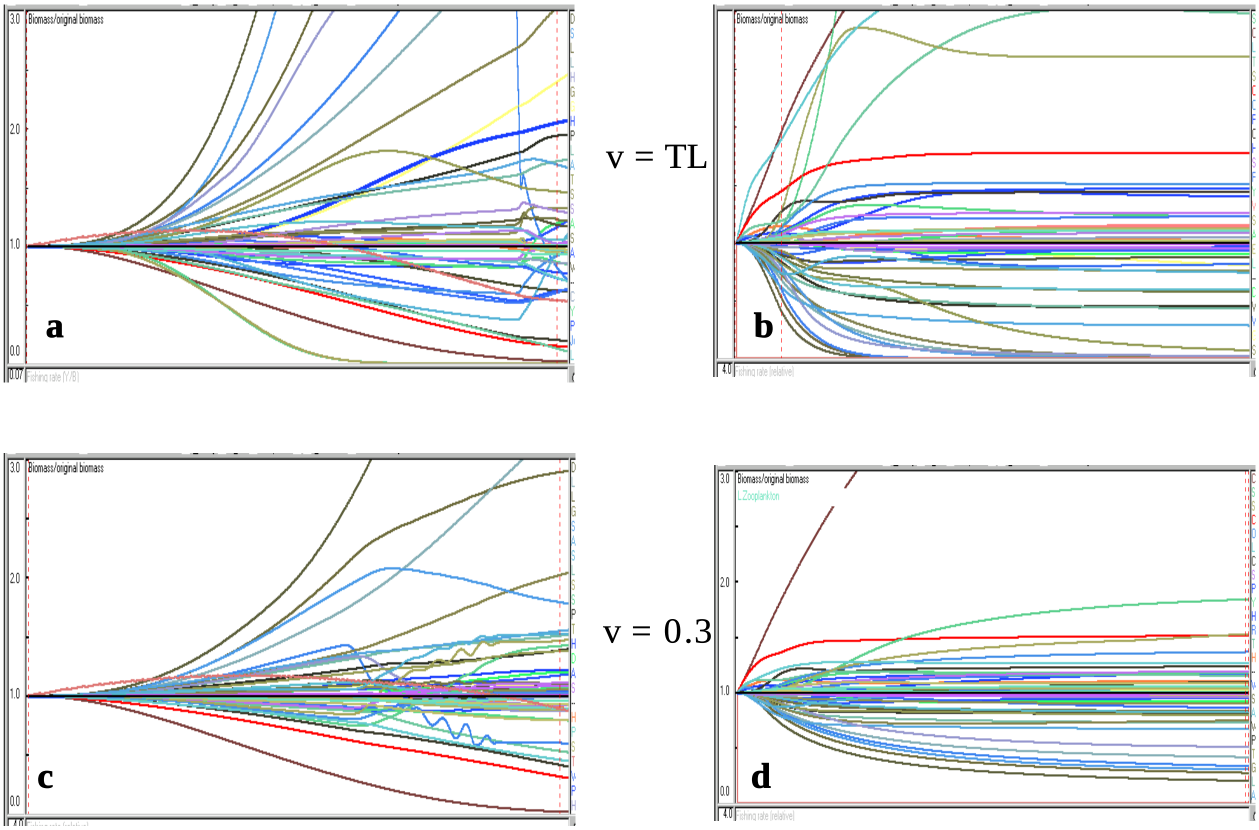

Figure 2. Results from simulating the 1900 Newfoundland model for 100 years without fishing (b and d) and with an increase in fishing mortality of 1% per year (a and c). Vulnerability settings by trophic level with an upper limit of 0.8 and an lower limit of 0.2 are shown in a and b, while c and d show default vulnerability settings at *Ecosim* (0.3).

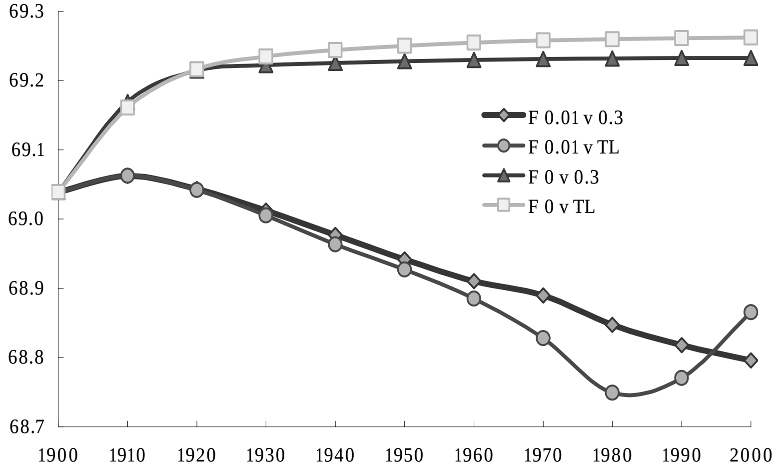

Figure 3. Resilience (ratio of overhead to development capacity) in the four scenarios, without fishing and with an increase of 1% in fishing mortality per year over 100 years and with vulnerability set at *Ecosim* base (0.3) or by trophic level (of prey) with a range of 0.2 - 0.8.

of Newfoundland (2J3KLNO) constructed for the time period 1900-1905 (Heymans and Pitcher 2002) was subjected to two fishing regimes. First, fishing was eliminated totally, and second fishing mortality was increased by 1% each year, for each of the species fished in the 1900-1905 model. The simulations were run for 100 years, and at 10 year intervals a new *Ecopath* model was created, re-imported into *Ecopath* and its network analysis properties calculated, without balancing these models. The ratio of overhead to development capacity (Φ/C) was plotted against time for both scenarios.

When fishing was eliminated from this model, some species seem to go extinct due to the high vulnerability parameters used in the *Ecosim* simulations. For the policy search simulations the vulnerabilities were set equal to trophic level (by prey), with the maximum v = 0.8 and minimum v = 0.2 (Ainsworth 2004, this volume). Resetting the vulnerability parameters to 0.3 (*Ecosim* baseline) eliminated these extinctions. This model was then also subjected to the 1% increase in fishing mortality, to give four scenarios for testing the hypothesis that (Φ/C) is related to the resilience of the system.

Resilience methodology

For the purposes of 'Back to the Future' these policy optimizations were run for 50 years, and at year 50 a new *Ecopath* model was created, re-imported into *Ecopath* and its network analysis properties calculated, without balancing. The resilience obtained from these final models were then compared to the base model resilience to see if they changed markedly from the base model, indicating if the policy regime chosen have increased or decreased the resilience of the system.

RESULTS

The four scenarios are shown in Figure 2 (a-d). Figure 2 show the results of the 1900 Newfoundland model simulated without fishing and with vulnerability settings at trophic level and at *Ecosim* base (0.3), and with an increase in fishing mortality of 1% per year over 100 years, with vulnerability settings at trophic level and at *Ecosim* base (0.3). The ratio of overhead to development capacity (Φ/C) was hypothesized to be analogous with the resilience of the system (Ulanowicz 1997, Ulanowicz 1980, Ulanowicz and Norden 1990). This ratio was plotted against time for all four scenarios in Figure 3.

DISCUSSION

From Figure 1a and 1c it is evident that increasing fishing mortality in the Newfoundland model drives the ecosystem to instability (especially in the case of the higher vulnerability settings, v = TL, Figure 1a). In Figure 1c the changes in the ecosystem are not as severe, due to the reduced effect of the vulnerability parameters, but the system is still dramatically affected. Removing fishing from the ecosystem causes some species to increase and some to decrease (Figure 1b and d), with some extinctions, when vulnerabilities are set equal to trophic level. The model does however stabilize in both instances within 20 or 30 years.

All things being equal, it would be expected that the resilience of the system should increase at first when a stressor such as fishing is taken from the system, up to an asymptote where the resilience of the system would not be affected. From the results in Figure 2 it is evident that the overhead/development capacity (Φ/C) ratio does increase as expected in the first 20 years, when fishing is eliminated from the system.

Conversely, the (Φ/C) ratio decrease when fishing mortality is increased (after an initial small increase). In the case of the vulnerability parameters being set to *Ecosim* base (0.3) this decrease is nearly linear over time. In the case of the vulnerability parameters of each prey being set to trophic level (range 0.2-0.8), the (Φ/C) ratio decrease more dramatically over the final half of the simulation, and is at its lowest level in 1980, just prior to the system crash (Figure 1a). After the crash in the 1980s, the (Φ/C) ratio increase again to levels similar to that of 1960.

According to Ulanowicz (1986), the (Φ/C) ratio shows the increase in freedom (disorder, strength in reserve) as oppose to the organization and specialization of the system. Thus, as the fishing mortality increase, and the ecosystem seem to have quite a few species increasing in biomass (Figure 1a and c), the system seems to become more specialized and organized, while loosing freedom and resilience (Figure 2). However, the system is unable to sustain this specialization, and it crashes (Figure 1a), after which its resilience start to increase (Figure 2). These results therefore support the assumption that the overhead/development capacity (Φ/C) ratio is an indication of resilience of this ecosystem.

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Note: *This paper is an earlier version of:*

Heymans, J.J. (2003) Comparing the Newfoundland marine ecosystem models using information theory. Pages 62-71 in Heymans, J.J. (ed.) Ecosystem models of Newfoundland and Southeastern Labrador: Additional information and analyses for 'Back to the Future'. Fisheries Centre Research Reports 11(5): 79pp.

MODIFYING KEMPTON'S SPECIES DIVERSITY INDEX FOR USE WITH DYNAMIC ECOSYSTEM SIMULATION MODELS

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ABSTRACT

The Q-90 statistic, a variant on Kempton's Q index, is used to measure the effects of hypothetical harvest strategies on the biodiversity of the restored Lost Valley ecosystem. The statistic represents the slope of the cumulative species abundance curve between the 10 and 90 percentiles. In applying Kempton's method to *Ecosim* results, functional groups are considered 'species' and their biomass, sorted into bins, is analogous to the number of individuals (as when compared to field sampling studies). A Visual Basic algorithm generates an annual Q-90 value based on *Ecosim*'s output CSV file; this allows us to monitor biodiversity over the course of the simulation. Comparing the biodiversity trajectory generated by different harvest strategies, this technique provides us another method to evaluate the success of the harvest plan from an ecological perspective. This methodology is meant to complement previously described economic valuation procedures.

The 'Lost Valley' approach (Pitcher 2004 this volume, Pitcher *et al.* 2004) assumes that conservation efforts have restored the marine ecosystem to some historical level of abundance. Through *Ecosim*'s policy search routine we have generated strategies to harvest the restored system according to a variety of ecological, economic and social priorities (see Ainsworth *et al.* 2004, this volume).

Using gaming scenarios, *Ecosim* returns suggested fishing efforts for each gear type in the base *Ecopath* model that will harvest the ecosystem sustainably over the course of the simulation and maximize benefits according to the desired objective. In this paper we develop a procedure to monitor the effects of those harvest strategies on the biodiversity of the restored system over time.

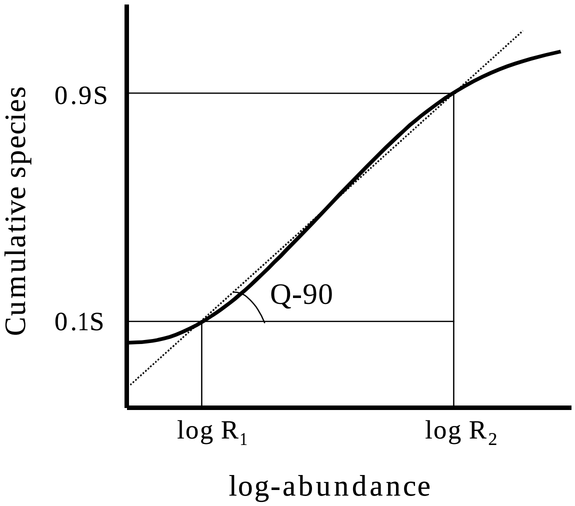

Figure 1. Representation of Q-90 statistic. S is number of functional groups in reference model; R_1 and R_2 are lower and upper 10 percentiles of the species abundance distribution. Modified from Kempton and Taylor (1976).

METHODS

Biodiversity Index

Species diversity is measured here by the Q-90 statistic, a variant on Kempton's Q index (Kempton and Taylor, 1976). Kempton's Q index describes the slope of the cumulative species abundance curve. The index is robust against changes in sample size (provided that very small samples are avoided), is not dependant upon the assumption of a particular species abundance model, is not biased by very abundant or very rare species, and expresses both speciosity and evenness (Magurran 1988).

Q-90 statistic

In the case of field sampling, Kempton and Taylor suggest using the inter-quartile slope of the species abundance curve in order to circumvent problems arising from the inclusion of tails (which may be long and include a high number of low-abundance species). In applying this methodology to *Ecosim*, tails become less of a problem since we there are almost no low abundance functional groups in the base model. Our Q-90 statistic therefore represents the slope of the cumulative species abundance curve between 10 and 90 percentiles, rather than quartiles (Figure 1). Each functional group in the model represents one "species" and the biomass of the functional groups, sorted into bins, serves

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Ainsworth, C. and Pitcher, T.J. (2004) Modifying Kempton's Biodiversity Index for Use with Dynamic Ecosystem Simulation Models. Pages 91–93 in Pitcher, T.J. (ed.) Back to the Future: Advances in Methodology for Modelling and Evaluating Past Ecosystems as Future Policy Goals. Fisheries Centre Research Reports 12(1): 158 pp.

as a proxy for the number of individuals in that species. The statistic is defined by the following relationship:

$$Q90 = \frac{\frac{1}{2} n_{R_1} + \sum_{R_1+1}^{R_2-1} n_R + \frac{1}{2} n_{R_2}}{\log(R_2 / R_1)}$$

Where n_R is the total number of functional groups with abundance R; R_1 and R_2 are the representative biomass values of the lower and upper 10 percentiles in the abundance distribution; n_{R1} and n_{R1} are the number of functional groups that fall within the R_1 and R_2 bins, respectively.

The lower and upper 10 percentiles are chosen such that:

$$\sum_{1}^{R_1-1} n_r < 0.1 \bullet S \leq \sum_{1}^{R_1} n_r$$

and $$\sum_{1}^{R_2-1} n_r < 0.9 \bullet S \leq \sum_{1}^{R_2} n_r$$

Where S is the total number of functional groups in the model.

Applying Q-90 to Ecosim output

Ecosim returns the functional group biomass data for each simulation year in a comma delimited text file (CSV). However, at present, the program does not permit extinctions; it instead returns a low non-zero value for critically depleted groups. Therefore, every harvest scenario will contain the same number of functional groups as in the base model. To increase the sensitivity of the index to group depletions, a filter is passed over the biomass profile each year of the simulation. If the biomass of a given functional group falls below a reference value, that group is considered "extinct" and is omitted from the Q-90 calculation – this will reduce the measured biodiversity of the system. In evaluating Back-to-the-Future past and present ecosystems, the undepleted biomasses found in the most pristine ecosystems (typically represented by pre-contact models) are chosen as reference values, and an arbitrary fraction of that biomass defines the extinction threshold. The threshold is typically set to 60% of the unfished biomass, but this value may be reduced when evaluating severely depleted systems. For example, the present-day Newfoundland ecosystem has been more heavily depleted compared to its pre-contact counterpart than has Northern British Columbia (these models are described in Ainsworth *et al.* 2002). A lower extinction threshold is therefore required in the former to improve the resolution of the biodiversity index.

Description of the algorithm

A Visual Basic algorithm reads biomass from *Ecosim*'s output CSV file and converts the monthly data into annual averages. A user-defined number of bins are established that represent 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. If any group falls below its reference biomass, it is omitted from the procedure. Bins may be linear or logarithmic; in the case of the latter each bin is 10% larger than the previous. The upper and lower 10 percentiles are determined as the bins in which 10% and 90% of the functional groups occur. The Q-90 statistic is calculated and plotted for each year in the simulation.

The statistic is most useful for evaluating *Ecosim* output created from the same or similar static models. For instance, the affects of alternative harvest strategies on the same *Ecopath* model may be evaluated, or the affects of analogous strategies on several related base models (e.g. models representing different time periods, but containing equivalent groups).

CONCLUSIONS

The modified Kempton's Q statistic provides a convenient means to judge the affects on biodiversity of a hypothetical harvest strategy and allows us to monitor one aspect of ecological health over time. In terms of the Lost Valley, this technique complements two other ecological valuation methodologies: the ascendancy index of Heymans (2004) and Cheung and Pitcher's (2004) technique to estimate sub-extinctions within composite functional groups. Together, these methods can monitor ecological consequences of a proposed Lost Valley harvest strategy, and when paired with the economic evaluation described in Ainsworth and Sumaila (2004), allow us to thoroughly evaluate the harvest strategy. Once we have described the economic and ecological attributes of a given Lost Valley scenario, we are able to provide management with an objective tool to weigh potential benefit with the costs of restoration.

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AN INDEX EXPRESSING RISK OF LOCAL EXTINCTION FOR USE WITH DYNAMIC ECOSYSTEM SIMULATION MODELS

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ABSTRACT

This paper derives a least squares empirical relationship to enable prediction of the likelihood of local extinction (= extirpation) of the most vulnerable species that has been grouped with other species into one functional component ('box') in a dynamic ecosystem simulation model (*Ecopath*-with-*Ecosim*).

INTRODUCTION

The effect of fishing has become a conservation concern, following cases of local extinctions and extirpations of marine species as result of fishing (Dulvy *et al* 2003, Sadovy and Cheung 2003). Restoring a marine ecosystem from its current over-exploited state and sustainable management of the rebuilt system is an innovative way to prevent fishing from driving marine species to extinction (Pitcher *et al.* 2004). An approach termed 'Back to the Future' (BTF), which integrates ecosystem modelling, socio-economics analysis, community participation in policy exploration and evaluation (Pitcher 1998, Pitcher and Pauly 1998, Pitcher *et al.* 2004), is being developed. It aims at restoring depleted marine ecosystems back to a previous lower exploited and healthy state, which can provide long-term ecological, social and economic benefits to the present and future generations.

The BTF approach relies strongly on the use of ecosystem modelling tools, *Ecopath* with *Ecosim* and *Ecospace* (*EwE*) (Walters *et al.* 1997). In the model, biota in marine ecosystem are modelled as functional groups. Therefore the model does not directly address issues relating to biodiversity change in the ecosystem, except at the functional group level. Particularly, extinction (regionally or globally) of a species within a functional group would not be revealed. Therefore, the risk of species extinction or extirpation associated with fishing cannot be explicitly dealt with when evaluating different policy options to restore and exploit the ecosystem. However, this could be overcome by developing an index which can indicate the extinction risk of species within the functional groups under different fishing patterns.

Table 1. Attributes of growth rate and productivity that are related to vulnerability of marine species to extinction as suggested from published literature (Musick 1999, Roberts and Hawkins 1999).

Related attributes/parameters	Vulnerability to Extinction High	Vulnerability to Extinction Low
Intrinsic rate of increase (r)	Low	High
Longevity (tmax)	Long	Short
Natural mortality rate (M)	Low	High
Production biomass	Low	High
Von Bertalanffy growth (k)	Low	High
Fecundity	Low	High
Age or size at sexual maturity	Old or Large	Young or Small
Reproductive frequency	Semelparity	Iteroparity

Life-history characteristics of a species or functional group relate to their risk of extinction. Previous studies identified growth rate and productivity as important characteristics that affect the vulnerability of marine species to extinction (Musick 1999, Roberts and Hawkins 1999), and these factors can be further subdivided into a number of attributes (Table 1). These attributes can generally be incorporated in the production rate and production biomass of a population. In *Ecopath*, production rate and production biomass are explicitly expressed as the production to biomass ratio (P/B), and the biomass of each functional group. Therefore, under certain fishing rates and other factors being equal, it is expected that P/B ratio should negatively correlate with the extinction risk of a population, or positively correlate to the time required for it to become extinct. Moreover, rare low biomass species are suggested to be more vulnerable to extinction (Musick 1999), and the P/B ratio is negatively correlated to extinction risk (Table 1). Therefore, it is expected that species with lower initial biomass will be more vulnerable to extinction.

If the above propositions hold, species with different P/B ratios and initial biomasses, which have been grouped together in the functional group of a model, should become extinct at a different rate if they are subjected to a similar intensity of fishing. We also expect to see an empirical relationship between the time when each species becomes extinct and the P/B ratio and initial biomass of the species. Moreover, by assuming that the change in overall biomass of the model group is a function of the change in abundance of each species within the group, change in group biomass can be used as indicator

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Cheung, W-L. and Pitcher, T.J. (2004) An Index Expressing Risk of Local Extinction for Use with Dynamic Ecosystem Simulation Models. Pages 94–102 in Pitcher, T.J. (ed.) Back to the Future: Advances in Methodology for Modelling and Evaluating Past Ecosystems as Future Policy Goals. Fisheries Centre Research Reports 12(1): 158 pp.

Table 2. Parameters for sub-groups investigated in the hypothetical *Ecosim* model. For further details see text.

Functional Group	Scenarios	Sub-group	P/B				Biomass (t km^{-2})			
Apex predators	3 sub-groups	Apex 1	1.099	1.041	0.983	1.157	0.018	0.015	0.012	0.009
		Apex 2	1.157	1.157	1.157	1.157	0.018	0.018	0.018	0.018
		Apex 3	1.215	1.273	1.331	1.157	0.018	0.021	0.024	0.027
	7 sub-groups	Apex 1	0.636	0.983	0.810	1.157	0.005		0.008	
		Apex 2	0.810	1.041	0.926	1.157	0.006		0.008	
		Apex 3	0.983	1.099	1.041	1.157	0.007		0.008	
		Apex 4	1.157	1.157	1.157	1.157	0.008		0.008	
		Apex 5	1.331	1.215	1.273	1.157	0.009		0.008	
		Apex 6	1.504	1.273	1.388	1.157	0.010		0.008	
		Apex 7	1.678	1.331	1.504	1.157	0.011		0.008	
Mesopelagics	3 sub-groups	Meso 1	0.557	0.546	0.516	0.607	0.71	0.73	0.743	0.844
		Meso 2	0.607	0.607	0.607	0.607	0.844	0.844	0.844	0.844
		Meso 3	0.637	0.668	0.698	0.607	1.266	1.055	0.945	0.844
	7 sub-groups	Meso 1	0.334	0.516	0.425	0.607	0.308		0.362	
		Meso 2	0.425	0.546	0.486	0.607	0.326		0.362	
		Meso 3	0.516	0.577	0.546	0.607	0.344		0.362	
			0.607	0.607	0.607	0.607	0.362		0.362	
		Meso 1	0.698	0.637	0.668	0.607	0.380		0.362	
		Meso 2	0.789	0.668	0.728	0.607	0.398		0.362	
		Meso 3	0.888	0.698	0.789	0.607	0.416		0.362	
Benthic fishes	3 sub-groups	Benthic 1	0.07	0.071	0.072	0.074	0.440	0.450	0.455	0.463
		Benthic 2	0.074	0.074	0.074	0.074	0.463	0.463	0.463	0.463
		Benthic 3	0.096	0.089	0.081	0.074	0.602	0.556	0.509	0.463

of species extinctions.

In this study, the above hypotheses were tested by comparing results obtained from simulations of a hypothetical *Ecopath* model. Species within a group were split into individual groups in the model and results obtained from *Ecosim* simulations were compared with those obtained from a model without sub-dividing the functional group. An empirical model was then developed to calculate an extinction index that can be used to approximately estimate the occurrence of a species extinction event.

METHODOLOGY

A hypothetical *Ecopath* model, supplied with the software, the 'ocean-test model', was used to generate simulation results (details of the ocean-test model are summarized in Annex 1). The functional groups: apex-predators, mesopelagics, benthic fishes, and large-zooplanktons were split into three and seven sub-groups. Each sub-group was assumed to be a composite species of the corresponding functional groups, with the same diet composition, production to consumption ratio (P/Q), fished at the same intensity, but with varying biomass and P/B ratios (Table 2).

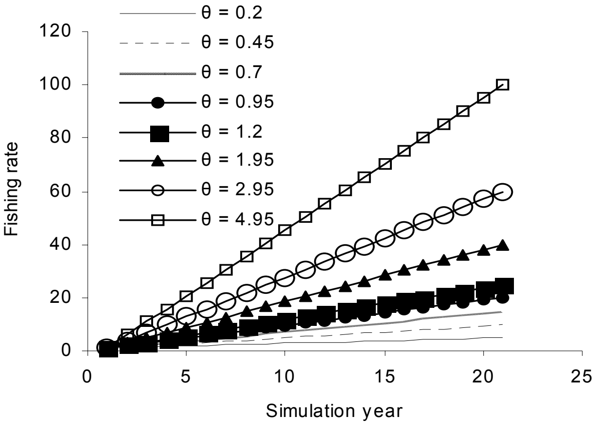

Figure 1. Patterns of fishing rate (%) in the *Ecosim* simulation of the 'Ocean-test Model'. The average increases in fishing rate are shown in the legend.

Ecopath models developed for each of the above scenarios were simulated under a range of fishing patterns in *Ecosim* (Figure 1). The time-series of biomass changes in each simulation were recorded. The results were expressed as a ratio of the sum of biomasses of all sub-groups at simulation time t (B_t), to the sum of biomass of these groups in the *Ecopath* base model (B_e). We recorded this B_t/B_e ratio and the simulation time when each of the sub groups became extinct (B_{ext}/B_e). Extinction of a sub-group was defined as when its biomass was reduced by more than 99% its initial base model level.

Input parameters of the models and simulations were plotted against the B_{ext}/B_e values and evaluated with regression analysis. The independent variables include the P/B ratios (Φ_i), the biomass (B_i) of individual sub-groups (i), the standard deviations of the P/B ratios (δ_j), the biomasses (γ_j) of the sub-groups (j), and the average rate of increase in fishing rate (θ). A regression model with B_{ext}/B_e as the dependent variable was developed from simulation results of the apex-predators, mesopelagics and benthic

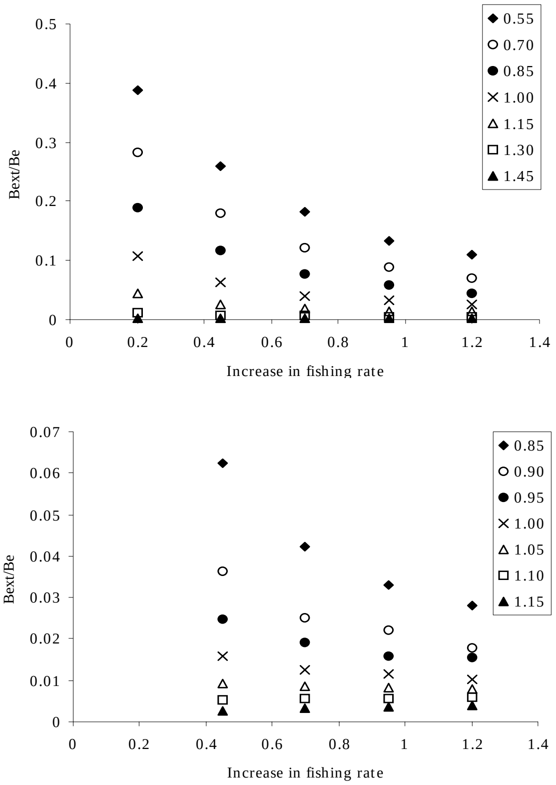

Figure 2. Plots of the ratio of model group biomass to initial biomass at extinction, Bext/Be (y-axis) resulting from different increases in fishing rate, θ (x-axis). Species (sub-groups) with different P/B ratios within the model groups are shown in the legend. Upper panel shows results from apex predators group with a standard deviation of the P/B ratio = 0.125. Lower panel shows mesopelagics group with sd = 0.066.

fishes. Results from simulations of the large-zooplanktons were not include in the regression as they were used to test the validity of the regression model.

RESULTS

A total of 433 data points were generated from the *Ecosim* simulations of the apex-predators, mesopelagics and benthic fish model groups. Analysis of the data suggested that the observed B_{ext}/B_e obtained from the simulations could be explained by four components in the regression model.

(1) *Fishing rate component (G)*

From the data, there is a consistent relationship between the increase in fishing rate (θ) and the observed B_{ext}/B_e. Data obtained from simulations of the apex-predators and mesopelagics are shown as examples in Figure 2, which can be fitted with a logistic model:

$$G = [a/(b^* \theta + c)] + d^* \theta \qquad (1.1)$$

where a, b, c and d are coefficients determining the shape of the relationship, θ is the average rate of increase in fishing rate, and G is a function of B_{ext}/B_e:

$$B_{ext}/B_e = f(G) \qquad (1.2)$$

(2) *P/B component*

The shape of the curve from equation 1.1 varies with the P/B ratio (Φ, normalized to the mean P/B ratio of the model group, Figure 2). Therefore, it is suggested that Φ is a function of coefficients a, b, c and d. The simulated data suggest that Φ is non-linearly related to coefficient a, and linearly to coefficients b, c and d (Figure 3). As such, it is assumed that:

$$a = m_1/\,\Phi + n_1 \qquad (2.1)$$
$$b = m_2{}^*\,\Phi + n_2 \qquad (2.2)$$
$$c = m_3{}^*\,\Phi + n_3 \qquad (2.3)$$
$$d = m_4{}^*\,\Phi + n_4 \qquad (2.4)$$

where m_i and n_i are constants.

Setting the model group standard deviation of P/B ratios (δ) and mean P/B ratio (α) as independent variables while Φ and other factors are kept constant, we found that a second degree polynomial of δ and α are functions of B_{ext}/B_e. So:

$$B_{ext}/B_e = f(\alpha^* \delta^2) \qquad (2.5)$$

(3) *Biomass component (H)*

The two sub-models above cannot fully explain the results obtained from the simulations when biomasses of the sub-groups are independent variables. A plot between the biomass (normalized to the mean) of the sub-groups (B_i) and B_{ext}/B_e of the corresponding model groups suggests a non-linear relationship between the two. The shape of this relationship is affected by the increase in fishing rate (θ) (Figure 4). Hnec we get:

$$H = s^*/[(1-B_i/t)/(B_i{}^* \theta)] - v \qquad (3.1)$$

where s, t and v are constants, and H and the standard deviation of the sub-group initial biomasses (γ) are functions of B_{ext}/B_e:

$$B_{ext}/B_e = f(H^* \gamma) \qquad (3.2)$$

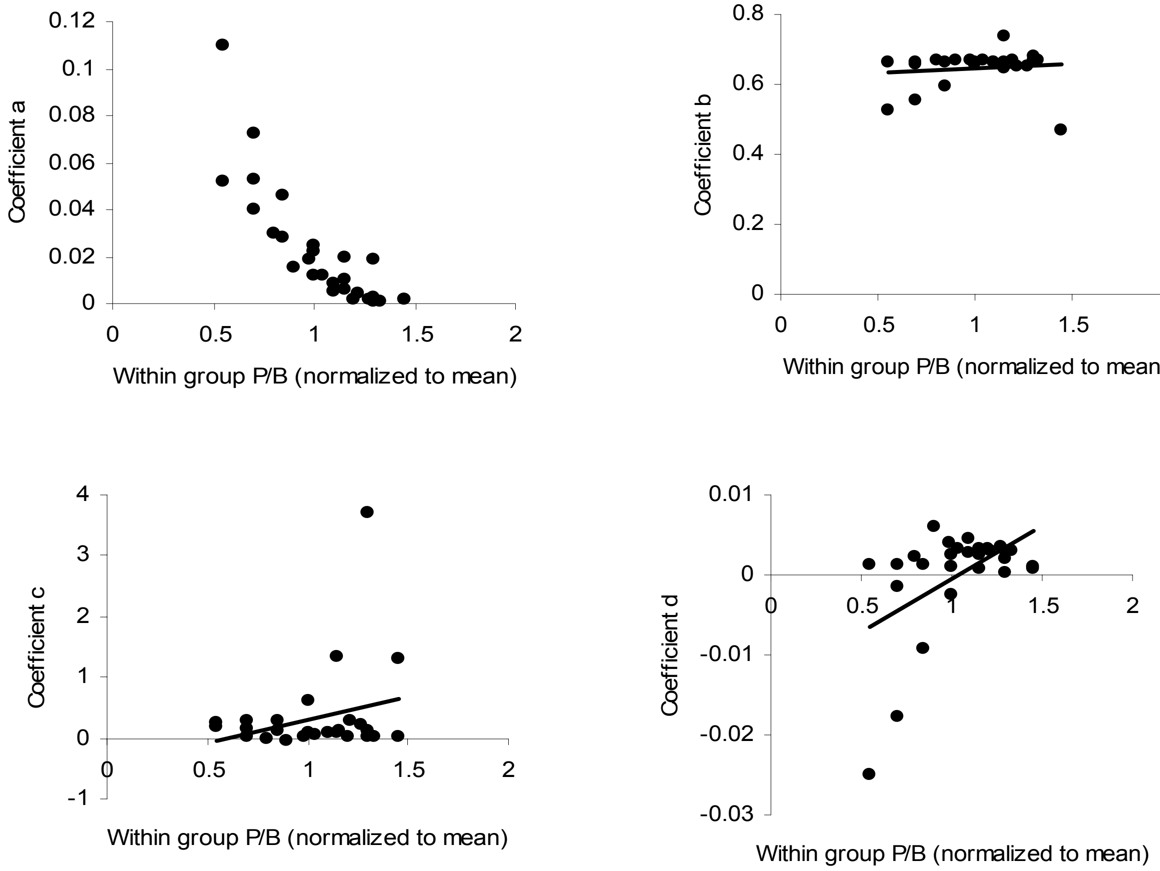

Figure 3. Plots of the coefficient for the P/B component model against P/B ratio of the sub-groups representing within-group species. Coefficient a has a non-linear relationship with the P/B ratio, while the relationships between coefficients b, c, and d with the P/B ratio are less clear, but are assumed to be linear.

(4) Extinction index component (B_{ext}/B_e)

The above results support our proposition that the risk of extinction, or the time at when extinction would occur, (expressed as B_{ext}/B_e) is dependent on the P/B ratio, initial biomass and fishing rates. Summarizing from equations 1 to 3, we suggest that:

$$B_{ext}/B_e = c_1 * G * \alpha * \delta^2 + c_2 * H + c_3 * \gamma + c_4 \qquad (4)$$

where c_i are constants, and B_{ext}/B_e must be greater than or equal to zero.

All the constants from equations 1 to 3 were obtained by fitting equation 4 to the results generated from the test simulations using a least squares method (Table 3). The coefficient of determination of the best fit is 90.2% (Figure 5a). When the biomass-dependent component is separated from the model (Figure 5b), the model explains over 97% of the variations in these groups from the ocean test *Ecopath-with-Ecosim* model.

Table 3. Values of the coefficients in the local extinction empirical model. The coefficients are estimated by fitting the model to the observed simulation data using least squares.

Coefficient		**Value**
P/B component		
	m_1	31.336
	n_1	-14.989
	m_2	-6.780
	n_2	9.458
	m_3	1.157
	n_3	0.361
	m_4	-2.757
	n_4	1.124
Biomass component		
	s	1.11
	t	1.186
	v	0.0153
B_{ext}/B_e component		
	c_1	0.131
	c_2	0.224
	c_3	0.00143
	c_4	0.00991

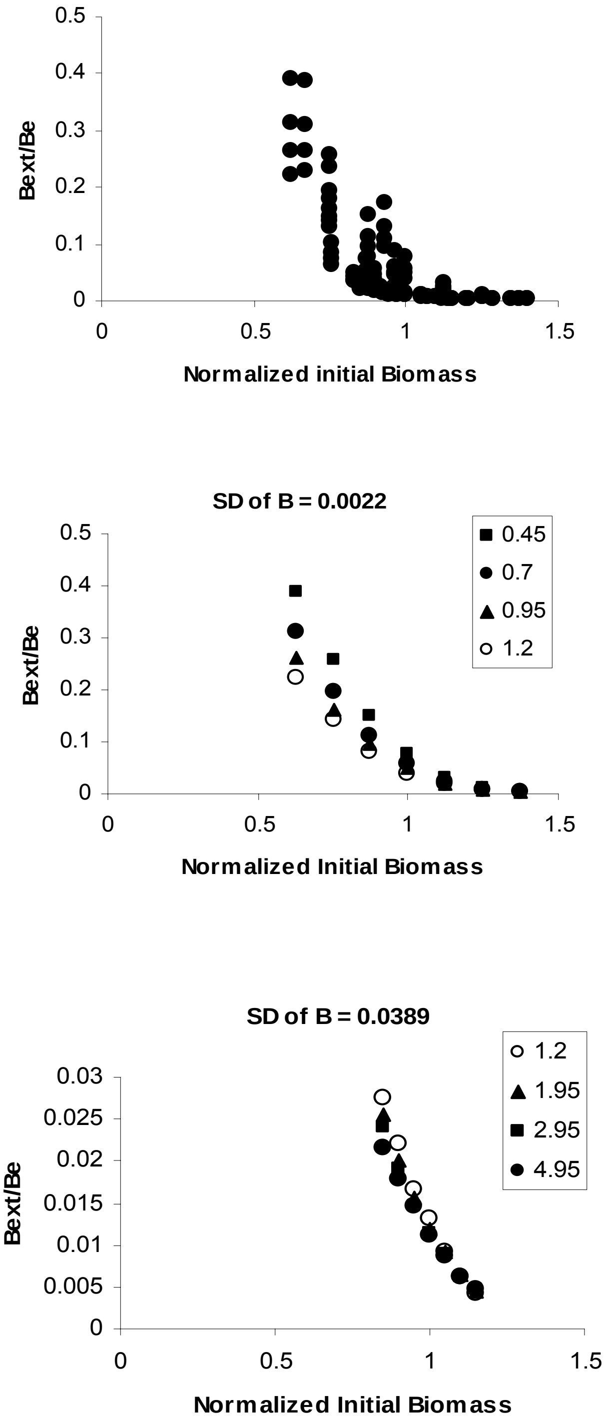

Figure 4. Plots of ratio of biomass at extinction to initial biomass of the within group species (Bext/Be) against the initial biomass of the species (normalized to the mean biomass of the functional group) (a) Top: all simulations data from different biomass, P/B ratio and fishing rates (N=151); (b) Middle: data with the standard deviation of biomass of the sub-groups (SD) = 0.0022; (c) Bottom: with SD = 0.0389.

Applying the model

Therefore, it is suggested that if the biomasses of the sub-groups (=species) are assumed the same, only the non-biomass dependent component should be used.

The estimated B_{ext}/B_e can then be used to determine when a species within the functional group may go extinct, given the conditions above. For example, in the hypothetical example, large zooplanktons were fished with a constant fishing rate of 18 times the *Ecopath* base fishing rate for 21 year. The extinction time for the 10 within functional group species that have different P/B ration, were predicted by the model (Figure 5).

The algorithm also applies to scenarios in which the increases in fishing rate are not constant. In a hypothetical scenario (Figure 7a), first, local peaks of fishing rate were identified and the average change in fishing rate (θ) between consecutive peaks was calculated, including the first peak from the original values at simulation time zero. B_{ext}/B_e of the species within the model group were obtained from each calculated θ, and compared with the simulated biomass of the functional group to see if extinction occurs before the particular peak of fishing rate was reached (Figure 7b).

Probability of local extinction

In many *Ecopath* models that have been constructed, information on the P/B ratio and biomass of individual species within a functional group are often unavailable. Therefore, B_{ext}/B_e could not be estimated using equation 5. Here, a surrogate approach was developed in which B_{ext}/B_e can be estimated by using a Monte Carlo approach to sample the P/B ratios of the species within a model group. A given range of P/B ratios and a pre-specified mean P/B ratio is all that is required. Since there is no evidence that P/B ratios within a model group are normally distributed, a rectangular prior distribution of P/B ratios is used. However, other distribution types can be employed according to different model structures.

A probability distribution of the percentage of species going extinct can be obtained from the Monte Carlo simulation. Moreover, the average time at when the first within-group species extinction occurs can be estimated. Distributions of the likelihood of extinction of the functional group 'Large Demersal Non-reef Associated Fishes' in a Hong Kong ecosystem model (1950s) are shown as example (Figure 8, Buchary and

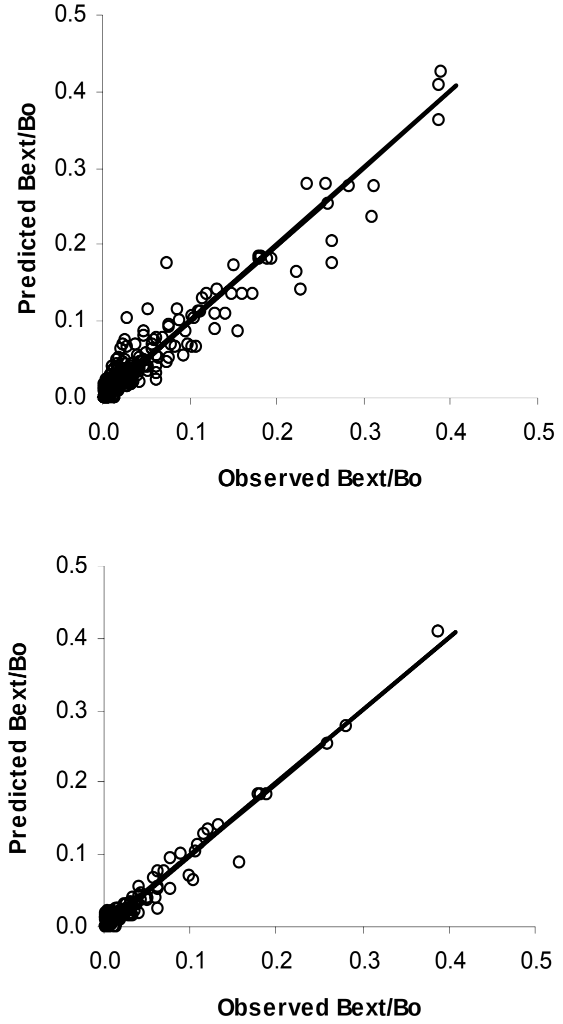

Figure 5. Comparisons between the ratio of biomass at extinction to initial biomass of within functional group species (Bext/Be) as predicted by the empirical models (equation 4) and Bext/Be observed from the *Ecosim* simulations; (a) Top: all data points (N = 432) R^2 = 0.902; (b) Bottom: data points with equal initial within group species biomass (N=281) R^2 = 0.972. The solid line represents exact agreement between the predicted and observed data.

Cheung, *unpublished*). The ecosystem was exploited under three sets of fishing rates that were found form optimality searches to maximize the ecological, economic and social values from the fisheries (see Ainsworth *et al.*, Pitcher 2004, this volume, Pitcher *et al.* 2004).

DISCUSSION

The model developed in this study is an attempt to estimate species extinction in the trophic mass balance model in which occurrence of species extinction in dynamic simulations are masked by the aggregation of species into functional groups. One of the problems in the latter is that when fishing strategies are evaluated in terms of their ecological, social and economic benefits, the ecological impact of fishing on a particular functional group may not be significant. However, it may pose serious threats to the survival of one or more of the within group species which possess characteristics rendering them vulnerable to extinction. Therefore, the extinction model can be used as one of the ecological indicators in evaluating the ecological effect of fishing strategies.

It is encouraging that the outcome of our empirical model agrees with existing views about the characteristics and factors that affect the vulnerability of extinction in marine species (Roberts and Hawkins 1999; Dulvy *et al.* 2003). For example, species with a lower production rate and production biomass will be more likely to go extinct first. The model fits well with the simulation data used to develop the model and provides reasonable predictions of extinction events within functional groups.

It should be noted, however, that the empirical model is based on numerous assumptions and approximations. The data which is used to fit the model is generated from a hypothetical *Ecopath* and *Ecosim* (*EwE*) model. *EwE* has its own sets of assumptions (Walters *et al.* 1997), and therefore the uncertainty of the results given by the empirical model will be magnified. It would be desirable if empirical data from known cases of marine species extinction could be used to develop the empirical model. However, reported cases of marine species extinction are insufficient to undertake the analysis in this study.

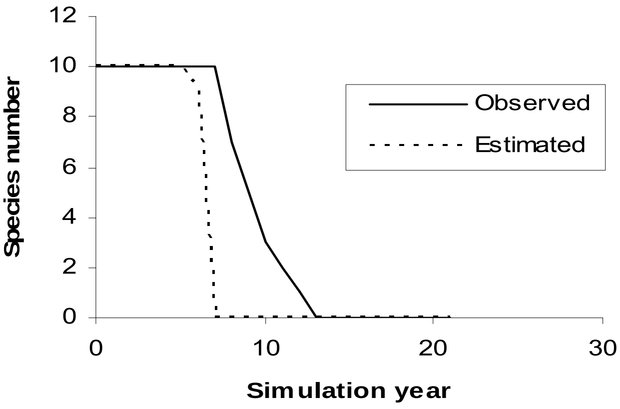

Figure 6. The change in species richness of the 'large-zooplanktons' group under a constant fishing rate 18 times the initial rate (10 species initially). Species were considered extinct if they dropped below 99% of their initial biomass. The solid line is the observed change from *Ecosim* simulation, while the dotted line represents the changes in species numbers predicted by our extinction model.

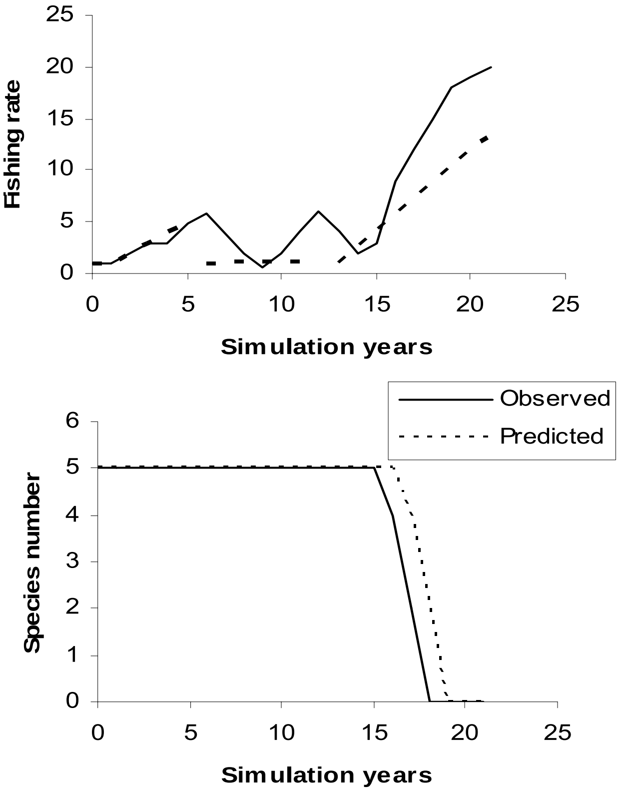

Figure 7. The change in species richness of the Apex predator group under fluctuating fishing rate. (a) Top: the change in fishing rate in the *Ecosim* simulations (solid line) and the between peaks increase in fishing rate (θ) (dotted line); (b) Bottom: species were considered extinct if they drop below 99% of their initial biomass. The solid line is the observed change from *Ecosim* simulation, while the dotted line represents the changes predicted by the extinction model.

EwE is determined by many more parameters than those being modelled in this empirical model. For example, vulnerability factors, which determinate the rates of exchange between vulnerable and non-vulnerable biomass of each functional group, are not taken into account in the empirical model (see Ainsworth 2004, this volume). Therefore, results obtained from the empirical model are only approximate.

Application of the empirical model should be restricted to fishes and invertebrates. Since the model is developed based on an extinction criterion of 99% reduction from the initial biomass, this may be too conservative for higher marine vertebrates or even for some marine fishes and invertebrates (Dulvy *et al.* 2003). Revision of the model can be undertaken should the extinction criteria be adjusted.

Furthermore, there are others factors which will affect the extinction vulnerability of marine species. For instance, degradation of critical habitats, as a result of destructive fishing, will have direct threats to the survival of the species (Musick 1999; Roberts and Hawkins 1999; Dulvy *et al.* 2003). Such factors are not taken into account in the empirical model.

Because of the assumptions and approximations of the empirical model, it does not produce, and should not be seen as producing, accurate prediction on the time and likelihood of species extinction under a given fishing intensity. Other more rigorous analytical method, such as the various population viability analyses (Boyce 1992; Brook *et al.* 2000), can be used if more accurate predictions are sought.

On the other hand, the small number of parameters required for the empirical model allows a convenient application, in particular for ecosystems where fisheries and ecological data are insufficient or species diversity is high which renders it difficult to model individual species as separate functional groups.

Moreover, the model can be used as an indicator to compare the possible effects of different fishing strategies in affecting species extinction risk. This is particularly useful in conducting 'Back to the Future' analyses in which alternative fishing strategies are evaluated and compared for their possible ecological, social and economic benefits and risks that can result.

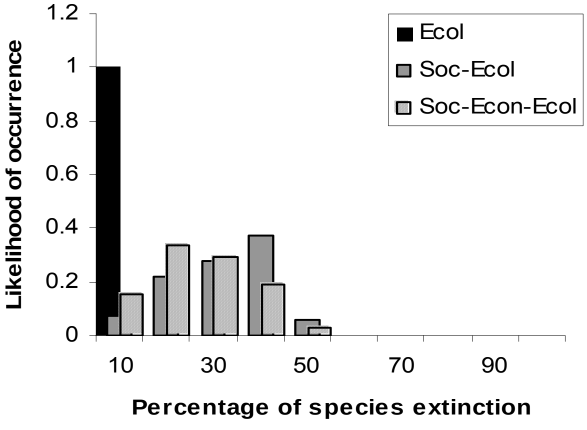

Figure 8. Distributions of the likelihood of extinction within the functional group 'Large demersal non-reef associated fish', which consists of 25 species, from three *Ecosim* simulations of a Hong Kong 1950s model. Fishing rates in the three scenarios aimed to maximize the ecological benefits (black), social and ecological benefits (shaded) and social, economic and ecological benefits (diagonals) from the ecosystem (from Buchary and Cheung, unpublished).

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ANNEX

Annex Table 1. Basic input and output parameters for the "Ocean test model". Bolded values were estimated from *Ecopath.*

Group name	Trophic level	Biomass (tkm^{-2})	P/B	Q/B	Ecotrophic efficiency	Fishery catch (tkm^{-2})
Apex predators	**4.26**	0.055	1.157	14.951	**0.930**	0.020
Mesopelagics	**3.35**	2.533	0.607	2.748	**0.912**	0.147
Epipelagics	**3.27**	0.516	1.991	9.230	**0.960**	0.020
Benthic fish	**2.67**	1.388	0.074	0.324	**0.861**	0.020
Benthopelagics	**2.61**	0.600	0.104	0.431	**0.942**	0.020
Zooplankt.large	**2.60**	9.864	0.466	2.684	**0.827**	0.020
Benthos	**2.05**	4.772	0.108	0.382	**0.590**	0.020
MicroZooplankt.	**2.00**	2.434	19.812	96.561	**0.456**	0.020
Phytoplankton	**1.00**	0.900	393.435	-	**0.695**	0.000
Detritus	**1.00**	1.000	-	-	**0.011**	0.000

Annex Table 2. Diet composition matrix of the "Ocean test model".

No.	Preys\Predators	1	2	3	4	5	6	7	8	9	10
1	Apex predators	0.048								0.048	
2	Mesopelagics	0.100	0.100	0.100						0.100	0.100
3	Epipelagics	0.752	0.050							0.752	0.050
4	Benthic fish				0.150						
5	Benthopelagics					0.150					
6	Zooplankt.large	0.100	0.250	0.400		0.200				0.100	0.250
7	Benthos				0.400	0.050		0.050			
8	MicroZooplankt.		0.600	0.400			0.600				0.600
9	Phytoplankton			0.100			0.400		1.000		
10	Detritus				0.450	0.600		0.950			

HOW DO WE VALUE THE RESTORATION OF PAST ECOSYSTEMS?

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The restoration of depleted/degraded marine ecosystems can be seen as a re-investment in natural capital that entails high short-term costs for benefits that, at least in business terms, will come in the distant future. This time gap between costs and benefits makes it particularly important to determine the costs and benefits of such a project through time to help determine the value of the undertaking. Determining the costs, and especially, the benefits of marine ecosystem restoration is quite challenging. This is because the benefits can be many and diverse; and they may accrue to both current and future generations. Proper valuation of ecosystem restoration will require the extension of current valuation methods, and the development of innovative new approaches. Sumaila and Charles (2002) suggest key questions and issues that need to be addressed regarding the value of restoration include:

What are the benefits (economic, ecological, social, cultural)?
What are the costs?
Over what time frame are benefits and costs measured?
What is the intergenerational flow of these benefits and costs?
How do we deal with discounting of future benefits and costs?
What about equity issues - do the benefits of restoration reach those who suffer the costs?
Who receives the benefits (fishers, First Nations, general public …?
Who incurs the costs (fishing industry, impacting industries, e.g., logging, pollution, urban growth, taxpayers?
What about the differing levels at which benefits and costs occur: individuals and corporations (e.g., resource users), communities, regions?

In measuring benefits, we must take into account all types of benefits (and costs), including consumptive uses (fishing, mineral extraction, etc.); non-consumptive uses (e.g., observation of wildlife, notably through tourism); non-use/existence value, the inherent value placed on the very existence of the ecosystem; and option value, the value placed on maintaining the marine ecosystem for possible future economic uses (see Sumaila and Bawumia 2092).

In measuring benefits, we must take into account the direct net benefits accruing from all relevant economic activity, e.g., fisheries, tourism, extraction of non-renewables, the non-use benefits, existence value and option value; all of these must be measured at the appropriate scale – the individual, as well as social and community benefits, including the spin-off benefits that may arise in the regional economy (e.g., increased post-harvest activity as a result of a more productive fishery).

Efforts at determining values from environmental and natural resources, in general and ecosystem restoration, in particular, have received some attention recently (e.g., Costanza *et al.* 1997, Weitzman 2001, Sumaila 2001, Sumaila and Walters 2003, 2004, Sumaila *et al.* 2001). An application is described in Ainsworth and Sumaila (2003), but more methods need to be developed.

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ECONOMIC VALUATION TECHNIQUES FOR BACK-TO-THE-FUTURE OPTIMAL POLICY SEARCHES

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ABSTRACT

We use conventional and intergenerational models of discounting to measure the economic success of each BTF restoration scenario in terms of net present value (NPV). The NPV term condenses the flow of future benefits into a single expression, and includes a time component to reflect the interests of an investor: with immediate benefits contributing heavily to the term, and far-off benefits discounted exponentially with time. The intergenerational model of discounting considers the needs of future generations better than the conventional model by including the arrival of new stakeholders each year. These entrants bring a renewed perspective on future earnings, partially resetting the discounting clock. Future work will weigh the economic success of each restoration scenario against the costs of achieving restoration.

INTRODUCTION

The economic cost associated with restoring the marine ecosystem to some level of its former diversity and abundance must be weighed against the additional benefit that the restored system would tender. Although costs and benefits may be measured in ecological and social terms as well (other papers in this volume consider these), we argue that economic considerations will take centre stage in determining the feasibility of any actual long-term conservation agenda (Ainsworth and Sumaila 2004).

We have therefore developed methodology to rank the *Lost Valley* ecosystem restoration goals, and their associated optimal harvest profiles, in terms of net present value (NPV) offered by the conventional and intergenerational (IG) approaches to discounting (Sumaila and Walters, 2004). The NPV term condenses the flow of future benefits into a single expression, while introducing a time component that reflects the interests of an investor: weighing immediate benefits heavily in the calculation, and discounting far-off benefits exponentially with time.

However, under the conventional discounting model, the future stock condition is worth so little in net present value (at any practicable level of discounting), that there emerges a tendency to focus on short-term benefit. Ainsworth and Sumailia (2003) postulate that this effect may have contributed to the Atlantic cod collapse. Therefore, we also value the BTF scenarios under the intergenerational discounting (IG) model of Sumaila and Walters (2004), which takes into account the needs of future generations better than the conventional model. The IG formula considers a continuous interlacing of generations, where devaluation of future benefit is counter-weighted each year by the addition of 1/G stakeholders, where G is the human generation time. The new entrants bring with them a renewed perspective on future earnings, partially resetting the discounting clock. Thus, the intergenerational approach will assign a high value to harvest scenarios that spread out benefits over several decades, while the conventional approach will favour scenarios that provide immediate profits at the expense of the standing resource.

Results from this analysis are presented in Ainsworth *et al.* (2004a) for northern BC evaluations, and Heymans *et al.* (2004) for Newfoundland. For information on *Back to the Future* (BTF) optimal policy search methodology, refer to Ainsworth *et al.* (2004b).

METHODS

Fishing mortalities per gear type, which are determined by an optimal policy search routine for each restoration period, fleet structure and harvest objective, are held constant in a 50-year dynamic *Ecosim* simulation. The resulting time series of absolute biomass is used to calculate landings (since they are not directly reported in the output CSV file). We assume the ecosystem reaches equilibrium after 50 years of harvest. The end-state values of biomass and harvest are then maintained for another 50 years in steady state. The first half of the simulation represents a development phase in the newly opened *Lost Valley* fishery, the second half represents a settlement phase.

Total catch per functional group for each year is converted into gross income by multiplying landed tonnes by wholesale market price. BC prices per functional group (Table A1) are based

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Ainsworth, C. and Sumaila, U.R. (2004) Economic Valuation Techniques for Back-To-The-Future Optimal Policy Searches. Pages 104-107 in Pitcher, T.J. (ed.) Back to the Future: Advances in Methodology for Modelling and Evaluating Past Ecosystems as Future Policy Goals. Fisheries Centre Research Reports 12(1): 158 pp.

on SAUP (2002), and modified by Pitcher (pers. comm.). These were converted to US dollars assuming an exchange rate of US $0.63 per Canadian dollar. For the BC models, price is also affected by gear type according to the estimated multipliers in Table A1. Newfoundland price per functional group is based on average Atlantic Canada values from 1995-1999 (DFO, 2002), and are shown in Table A.2 in US dollars. For Newfoundland models, functional group prices are the same for all gear types.

Non-market prices for northern BC (Table A.3) were obtained from Beattie (2001). These refer to estimated revenues from wildlife viewing, scuba diving and kayaking in the case of marine mammals and from sporting operations in the case of recreational species. Non-market values were not included in the Newfoundland models.

Cost is subtracted from calculated gross income to determine profit. Costs are assumed equal to 60% of gross income, based on DFO (1994). The annual profits over 100 years are then condensed into a single figure, the net present value (NPV), according to the following discounting methods.

Conventional discounting calculates NPV according to:

$$NPV = \sum_{t=0}^{T} \frac{NB_t}{(1+\delta)^t}$$

Where NB_t is net benefit in year t, δ is the discount rate and T is 100, the total number of simulation years.

Intergenerational discounting (Sumaila and Walters 2004, 2003, Sumaila 2001) employs the following relationship:

$$NPV = \sum_{t=0}^{T} \frac{NB_t}{(1+\delta)^t} \cdot \left(1 + \frac{t}{G}\right)$$

where G is human generation time (~20 years). For all discounting operations, discount rate was taken as 4% per year.

DISCUSSION

Optimal harvest profiles that slope upwards (with most harvest occurring late in the dynamic simulation) perform relatively better under intergenerational discounting than profiles that slope downwards (where most benefit is taken early). The former situation should correspond to optimal harvest profiles based on the 1950 and 2000 model baselines. Their conservative optimal fishing patterns, delivered by the policy search routine, will allow these depleted systems to rebuild, and the greatest harvests will be taken late in the simulation. The latter situation should correspond to harvest profiles based on the 1750 and 1900 baselines. As these represent more pristine ecosystem conditions, their optimal fishing patterns will aggressively mine the system in order to increase productivity; greater harvests will be taken early.

We expect the most lucrative policies to be identified by the optimal policy search routine under the economic objective, followed by the social objective, the mixed objective, the ecological objective and the portfolio log-utility objective (Walters *et al.*, 2002). We also expect the pre-contact ecosystem to generate greater benefits than 1900, 1950 or 2000 systems, since it contains the highest levels of abundance (Ainsworth *et al.*, 2002).

Future work will apply these results to a cost-benefit or cost-effectiveness analysis in order to weigh potential benefits against the costs of restoration (Ainsworth, in prep).

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For discussion after the oral presentation of this paper, see page 138.

ANNEX TABLES

Table A2. Prices (US$ per kg) for Newfoundland fisheries.

Group Name	$/kg
Cod (> 35 cm)	1.02
Cod (≤ 35 cm)	1.02
American plaice (< 35 cm)	0.85
American plaice (≤ 35 cm)	0.85
Greenland Halibut (> 65 cm)	1.17
Greenland Halibut (≤ 65 cm)	1.17
Yellowtail Flounders	0.85
Witch flounder	0.85
Winter flounder	0.85
Skates	0.25
Redfish	0.36
Transient mackerel	0.31
Demersal Bentho-Pelagic Piscivores (>40 cm)	2.06
Demersal Bentho-Pelagic Piscivores (≤ 40 cm)	2.06
Demersal Feeders (> 30 cm)	0.88
Demersal Feeders (≤ 30 cm)	0.88
Lumpfish	3.23
Greenland cod	1.02
Salmon	0.50
Capelin	0.17
Herring	0.12
Transient Pelagics	8.62
Small Pelagics	0.31
Shortfin squid	0.32
Arctic squid	0.32
Large Crabs (> 95 cm)	3.15
Small Crabs (≤ 95 cm)	0.57
Lobster	7.50
Shrimp	1.72
Echinoderms	1.56
Bivalves	1.03

Table A3. Non-market values used for northern BC models.

Group Name	Value/unit biomass
Mysticetae	0.8
Coho salmon	9.85
Chinook salmon	13.13
Inshore rockfish	0.27
Shallow water benthic fish	0.01
Infaunal carnivorous invertebrates	0.01

Table A1. Prices for northern BC fisheries. *Prices in $US. Price per species has been increased by the gear-type multiplier (bottom row).

Group Name	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						2.48	2.48				1.65	
Coho salmon						1.44					0.96	19.15
Chinook salmon						3.7					2.47	49.39
Ratfish	2.09	2.09										
Dogfish	0.35	0.35				0.35						
Pollock	0.31											
Eulachon		1.26									1.26	
Adult herring				0.29								
Adult POP	0.81											
Inshore rockfish	0.81				0.81	0.81		8.06				16.13
Adult picivorous rockfish	0.81					0.81						16.13
Adult planktivorous rockfish	0.81					0.81						
Juvenile turbot					0.2							
Adult turbot	0.2	0.2			0.2							
Juvenile flatfish					0.73							
Adult flatfish	0.73	0.73			0.73							
Juvenile halibut					2.56							51.16
Adult halibut					2.56						2.56	51.16
Adult Pacific cod	0.67				0.67							
Adult sablefish	0.63				0.63							
Adult lingcod	1.06				1.06			1.06				21.29
Shallowwater benthic fish		0.52	0.52	0.52								
Skates	0.14	0.14			0.14							
Large crabs	4.54								4.54			
Small crabs									3.64			
Commercial shrimp		3.07	3.07									
Epifaunal invertebrates										1.42		
Gear-type multiplier	1	1	1	1	1	1.5	1.5	10	1	1	1	20

AN EMPLOYMENT DIVERSITY INDEX USED TO EVALUATE ECOSYSTEM RESTORATION STRATEGIES

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ABSTRACT

We develop a social equity index based on the Shannon-Weaver entropy function for use in BTF optimal policy investigations. The index measures employment diversity across fishing sectors and ranges from from zero to one, where zero indicates no diversity (all fishing effort is concentrated in a single sector) and one indicates maximum diversity (fishing effort is distributed evenly among all sectors). This employment diversity index complements the social utility measure delivered directly from *Ecosim*: total employment.

INTRODUCTION

The employment diversity index presented here, after the methodology of Attaran (1986), was used to evaluate the simulated harvest profile offered by various restoration scenarios described in Heymans (2003) and Ainsworth (2004). 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:

$$D(E_1, E_2, \ldots E_N) = -\sum_{i=1}^{n} E_i \log_2 E_i$$

where,

n = the number of (possible) fishing sectors active in the ecosystem,

and,

E = the proportion of total employment that is located in the ith fishing sector.

The measure is normalized across sectors with respect to their maximum possible diversity so that $D(E_1, E_2, \ldots E_n)$ ranges from 0 to 1. D=0 indicates that all fishing activity is concentrated in a single sector, and 1 indicates the maximum possible employment diversity, with all sectors contributing equally to employment (all *Ei* equal).

Or,

$$D(E_1, E_2, ..E_n) = \left(-\sum_{i=1}^{n} E_i \log_2 E_i \right) \Big/ MAX(D(E_1, E_2, ..E_n))$$

APPLICATION TO ECOSIM

A *Visual Basic* 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 (see Ainsworth *et al.* 2004). 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. Total value per gear type is converted to relative number of jobs using an estimated 'jobs per catch value', as described in Ainsworth *et al.* (2004). Employment per sector (E_i) is then calculated as a fraction of total employment.

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EVALUATING FUTURE ECOSYSTEMS: A GREAT STEP BACKWARD?

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"Those who do not remember the past are condemned to repeat it"
George Santayana (1863-1952)

ABSTRACT

The Goal of Back to the Future is to restore some past level of abundance and diversity. The first objective is to engage scientists, managers, policy makers and the maritime community in developing the best possible computer models of present and past ecosystems. The second objective is to assign ecological and social as well as economic value to past and present systems, so that collaborators can set restoration goals. New valuation techniques, while innovative, use prices and costs from today's fleet to value past systems. This paper asks how we might harness the creative potential of the collaborators to design new fisheries that make sense in terms of the ecosystems and human communities that depend on them. A 'capital/interest' approach is suggested where the biomass essential to maintain productive potential and species of social and cultural importance are considered as natural and social capital, and, as such, not subject to commercial harvest.

'Back-to-the-Future' has strong ethical and participatory elements (Haggan 2000, Haggan *et al.* 1998), one goal of which is to find new ways for a very broad constituency to work on assigning ecological as well as social values when comparing ecosystem states. In brief, ecological value is assigned by giving fish in the water some value relative to those caught. For instance, one could assign equal value to fish in the ocean to those caught (Sumaila *et al.* 2001). Social value is assigned by including the value to future generations (Sumaila and Walters 2004).

One major problem that arose at the December workshop in Prince Rupert related to eulachon, an important food and trade item with high social and cultural value to First Nations. The past ecosystem models presented at the workshop showed very high dollar values for eulachons, derived from the only existing commercial fishery on the Fraser River. First Nation participants made it clear that they did not want a monetary value put on an integral part of their culture and subsistence economy.

How then can we assess, or indeed compare the real value of ecosystem components whose predominant values are non-monetary? This raises the question of 'Ecosystem Justice' addressed by Brunk and Durham (2000) in 'Just Fish: Ethics and Canadian Marine Fisheries' (Coward *et al.* 2000). Sumaila and Bauwumia (Ibid.) argue convincingly that the market cannot guarantee justice for ecosystem components that have no 'monetary value'.

Costanza and colleagues (1997) valued global ecosystem, or 'life support' services such as oxygen production at $US33 trillion/year, or almost double global GNP of $US18 trillion. The Costanza approach is related, as it values quantities that cannot be bought or sold, but is not directly comparable as it assigns dollar values

A 'CAPITAL-INTEREST' APPROACH

It seems to be a given that money is the only 'yardstick' that economists can readily apply. It is certainly a 'currency' that today's decision makers readily appreciate. Those who deal in money have a shrewd idea of the value of capital,. They also see it as something that should be conserved. Consider endowment funding where the interest from a significant capital amount is used to finance ongoing activities, cover core operations and maintain the principal against inflation, or indeed add to it over time. For example, the David and Lucile Packard Foundation dispensed ~$US614 million in 2000 (www.packfound.org) based on capital assets of approximately $9.8 billion. We might then consider the spawning biomass of species necessary to maintain a

Can quotas protect ecosystems?

Quota fisheries are seen by many fisheries managers as a way to protect the desired species. However quota holders have no incentive to protect other ecosystem components. Indeed the scientific uncertainty of existing stock assessment may require quotas that are so conservative that foregone catches could wipe out economic gains (Walters and Pearse 1996). Other authorities (Anderson 1994; Turner 1997) point to high-grading as an inherent problem of quota systems.

Haggan, N. (2004) Evaluating Future Ecosystems: A Great Step Backward? Pages 109-111 in Pitcher, T.J. (ed.) Back to the Future: Advances in Methodology for Modelling and Evaluating Past Ecosystems as Future Policy Goals. Fisheries Centre Research Reports 12(1): 158 pp.

desired ecosystem state as 'natural capital', MPAs would be another way to protect such natural capital. This can certainly be valued (Sumaila and Walters 2004, Sumaila *et al.* 2001), but could be protected by laws and regulations designed to protect resources in perpetuity.

Similarly, we might consider a category of 'social and cultural' capital to protect species such as eulachons and whatever amount of other species are necessary to maintain the culture and existence of First Nations (see Lucas 2004, this volume), and indeed aspects of the lifestyle of other maritime communities. Brody (1988) showed that subsistence hunting by interior British Columbia tribes had significant monetary value by quantifying the cost of equivalent foodstuffs and the value of furs, handcrafts and guiding. Nothing in Brody's work suggests that the tribes would have accepted money in lieu of these traditional activities (see Sumaila 2004, this volume).

FISHING RESTORED ECOSYSTEMS: KEEPING THE OPTIONS OPEN

A second problem arose as a result of using prices and costs from today's fisheries to value past ecosystem states. Hence, we drag existing fisheries structure back with us, ending up with 18 fisheries (16 existing and 2 new ones). This effectively perpetuates today's fleet structure and high degree of specialization where billions of dollars worth of vessels (to say nothing of license values) lie idle for most of the year. It also perpetuates existing divisions, forcing people to defend existing gear types instead of putting their minds to a fresh approach. The unfortunate example picked by the team for the December Prince Rupert workshop (Power, 2003, Power *et al.* 2004, this volume) simply illustrates the problem of forcing people to defend an existing structure rather than having the freedom to design new fisheries (or re-establish ancient methods such as selective trap and weir fisheries) in their home waters. An unfortunate consequence of the valuation approach *as applied* is to negate the opportunity provided by *Back to the Future* to take a new look at how to harvest restored systems.

A better question might be: if we could restore the abundance and diversity of the 1750s ecosystem, how would we harvest it – forgetting that we're 'salmon scientists' or 'halibut scientists' or gillnetters or trawlers or herring or halibut fishermen? Might we not want to consider more local, multi-species fisheries with multi-purpose vessels, where fisheries would be a year-round activity.

What about a form of area licensing that makes sense in terms of the ecosystem and the human communities, rather than an arbitrary line on a map? Such a system would 'vest' the interest in the resource in First Nations and other stable communities that have a long-term interest in maintaining productivity. This is important, as ownership by large corporations, or what Ommer (2000) characterizes as 'footloose' capital runs a real risk that large corporations would see economic sense in catching the last fish and investing the proceeds in ventures that will provide their shareholders with a higher rate of return.

We might also want to concentrate on methods that maximize value rather than volume, for instance, a 6.5 oz can of sockeye branded as 'Copper River Red' sells for $US 8.50 (www.copperriverred.com and see Simeone 2004, this volume), or, the value of live rockfish for the restaurant trade.

There is clear agreement on the need for flexibility in designing sustainable and responsible fisheries of the future. The criteria suggested by the CUS BTF team provide a start (see Pitcher 2004, this volume). But the challenge for *Back to the Future* is to find ways to improve and facilitate this with the participation of local fishing communities.

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INCORPORATING FIRST NATIONS' VALUES INTO FISHERIES MANAGEMENT: A PROPOSAL FOR DISCUSSION

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An attempt to include First Nations values into fisheries management is not an easy task. This is because First Nations values with respect to fisheries are very many and diverse, the values accrue in both direct and indirect ways; in tangible and intangible ways; in monetary terms and non-monetary terms; values can accrue to both the current and future generations (see papers by Lucas and by Simeone 2004, this volume). As daunting as the task of this paper is, we nevertheless have to devise methods and approaches that would enable us incorporate First Nations values into Canada's fisheries management. This is necessary to help us manage Canada's fishery resources in most equitable way.

There are two possible ways of approaching the problem of valuing and incorporating First Nations values into Canadian fisheries management. First, one may attempt to determine all First Nations values from marine ecosystems in dollar terms. Second, one can instead aim to incorporate First Nations values from marine ecosystems without valuing them in monetary terms. Both of these approaches have their advantages and disadvantages. The economic literature is full of methods to help implement the former approach. This implies that there are ample if not adequate tools available for determining values, both monetary and non-monetary from marine ecosystems. This can be counted as an advantage of this approach. A disadvantage of this approach is that First Nations do not believe their values can be adequately captured in monetary terms, and so the approach lacks credibility among its most important constituency (see Haggan 2004, this volume).

The advantage of the latter approach is precisely the fact that it has credibility among First Nations people, because it does not seek to put monetary values on the benefits they derive from marine ecosystem. A disadvantage of this approach is that it is not entrenched in the literature, so new approaches need to be developed to help implement it. The task of this note is to propose a modeling approach that can help us, technically, to include First Nations values into Canadian fisheries management.

PROPOSAL: HOW TO INCORPORATE VALUES WITHOUT VALUING IN DOLLAR TERMS

The proposed approach is based on a simple idea, that is, it imposes First Nations requirements (however, it may be determined) as an extra constraint within the stock dynamics of a single species model, or within a full-fledged ecosystem model. In this way First Nations values from fishery resources are incorporated before any commercial fishing is allowed. This approach actually provides a technical means by which to implement what has been Canadian law for many years – the Canadian Fisheries Act specifically stipulates that once the requirements for conserving Canada's fisheries resources are met, the next priority for Canadian fisheries management is to meet the requirements of First Nations before that of the commercial fishing sector.

To see how this may be incorporated in the stock dynamics of a fish stock, consider the equation below:

$$n_{0,t} = R_t \,,$$

$$n_{a,t} = (sn_{a-1,t-1} - \psi n_{a,t}) - h_{a,t}, \qquad \text{for } 0 < a < A,$$

$$n_{A,t} = [s(n_{A-1,t-1} + n_{A,t-1}) - \psi n_{A,t}] - h_{A,t}, \qquad n_{a,0} \ \textit{given}$$

where R_t is the recruitment of age 0 fish to the habitat in period *t* $(t=1..T)$; $n_{a,t}$ is the stock size of age *a* $(a=0..A)$ fish in period *t*; the parameter *s* is the age independent natural survival probability of cod; ψ is the fraction of the stock of a given age *a* fish in period *t* that is reserved for the First Nations; $n_{a,0}$ denotes the initial number of age *a* fish; and $h_{a,t}$ is the total harvest function for the commercial sector of a given age group in a given year.

Depending on the objective of fisheries management, an objective function with the

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Table 1. Results from the hypothetical model. For further detail, see text.

	Scenario 1	**Scenario 2**
Harvest 1 (1000t)	177 ($131 million)	124 ($46 million)
Harvest 2 (1000t)	196 ($307 million	141 ($156 million)
First Nations (1000t)	235	350
Biomass (1000t)	1210	809

above stock constraint can be computed and/or simulated to determine the appropriate allocation of the harvestable biomass to the commercial sector under the constraint that the allocation to First Nations is met.

A simple hypothetical example was simulated for a hypothetical single species fishery with three parties (agents or players) that exploit the fish. The three groups are First Nations fishers and two groups of commercial fishers, each with common interests. The stock dynamics of the fish are represented by the above equation.

It is assumed that the management objectives for this fishery are assumed to be twofold. First, allocate a portion of the harvestable biomass to the group of First Nations fishers. Second, allocate the remaining harvestable biomass to the two commercial fishers groups such that the sum of discounted profits they make is maximized. Using assumed biological and economic data, this hypothetical model is run using the software package *Powersim*. The outcome of the simulation is presented in Table 1 for two scenarios of quota allocated to the First Nations group – scenario 1: an average annual allocation of 235,000 tonnes, and scenario 3: an annual allocation of 350,000 tonnes. Table 1 shows the amount of harvest (discounted profit) the commercial groups make annually under the two scenarios. The table also reports the standing biomass under the two scenarios.

Since the numbers are derived from a hypothetical model no practical meaning should be ascribed to them – the exercise is meant only to illustrate how the proposed method may be implemented. It is worth mentioning that this approach can easily be implemented in multispecies and ecosystem models. In particular, it should be straightforward to incorporate this into *Ecopath* with *Ecosim*.

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ABORIGINAL VALUES

Simon Lucas
Chief, Hesquiat Nation, BC, Canada

Thank you. My name is Kla-kisht-ke-is, and I am from what you call the Hesquiat Nation. Within Hesquiat there are nine major groups that make up the tribe. Most tribes in British Columbia are made up like that. As you go up north, the tribes are divided into four clan-type arrangements. My area on the West Coast of Vancouver Island is an important part. It faces the Pacific Ocean.

I want to start off by saying that at one point in our life, River's Inlet was a very major part of the activity of our tribe. Many of the coastal tribes ended up in River's Inlet. The man that taught me a lot, Alec Games, he wrote every day about what he saw. He became a packer. He was packing fish in River's Inlet. It was there that many First Nations got to know each other. They exchanged many songs that are still sung today.

I want to talk about how we see things. If you are in Nuu-chah-nulth territory, the first thing you will hear is all of the tribes and chiefs saying, "everything is one for us". The second thing you will hear is about us as individuals: "all is one for us". Why do we talk that way? It is because of our understanding of the way things are. Our forefathers told us that one of the most important elements in life is rain. It plays a vital role in our territory in terms of the huge rivers and lakes that flow into the ocean. It is the belief of our ancestors that the mountains, the different variety of trees, and the grass and herbal medicines all contribute to the health of the ocean and the banks and food chain that makes up our territory. That is an understanding for us. So people say the centre of our life lies in this ocean. This is where all of our health comes from - all the species you heard about here and how we harvested them.

Kla-kisht-ke-is, Chief Simon Lucas, from the Hesquiat Nation, Vancouver Island, was awarded an Honorary Doctorate from the University of British Columbia in 2002, and is an Adjunct Professor in the Fisheries Centre, UBC, where he has lectured on Traditional Knowledge and traditional ecosystem management. *Photo: Martin Dee, Telestudios, Vancouver.*

As a young child growing up in Hesquiat, I was there until I was five years old. Everyone in our tribe had a canoe or two or three. There were different sizes meant for different kinds of weather. Some were only for fishing - there were huge canoes for traveling and others were for whaling. We had different kinds of canoes. One was 8 feet long and it was just for me. We spent a lot of time in the ocean. It was an important part of the entertainment. My father and grandfather knew when the cod fish were spawning, and after every storm we went walking on the beach. I knew what I was going to find because we were going to find codfish roe. That was important for the family. It was a habit for me to do after every storm. The other thing that my father used to say is that when tides are extremely low, he knew where to go for octopus. So from an early age, I knew how to find it and how to grab and kill it instantly. That was a natural diet for me.

We had huge mussels. One of the things that happened every night is that a guy whose nickname was White Man, one of the things he always did was take mussels in a huge pot. Everyone would go there to feast on mussels every night. The next thing I knew I was living on a place called Addison. That was where I understood my dad's activities. It was there that I went to the ocean, and understood that he always knew where to go for fish like rockfish and cod. In

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my young days, the cod was huge. He knew the migration routes of different stocks of the river.

Also important to understand in terms of value is that the ocean is extremely valuable to our people. In Nuu-chah-nulth where I come from, we do worship that has a lot to do with self-discipline and thanking and conversing with the Creator every day. I knew where to go in the river system. It is still being done there today. One of the rules about it is that you are not to watch anyone else do their worship. That is the law we have. You are not to watch because it is private. The other part of it was to dive into the ocean every day, and it deals with respect and cleanliness. I used hear the term "dirty Indians" when I went to school. I do not understand this because we swim a lot. Hemlock bark turns a little soapy and we used it for the preparation for the hunt of whales.

Sperm whales were hunted for their richness in oil. The ceremony was a one-year affair. You had to have abstinence. You had to be away from your wife and talk to the Creator every day. One of the things that came out of it was self-discipline so there was no glory or dominance over the whale. You are getting it for your grandchildren so they will be healthy, so you ask the Creator that you do not go above your values. That valuable tradition almost got lost when we went into the residential schools. We are lucky that some of our people went underground so that the values of the people were kept. We have people in our tribe who never spoke English and never went to school. There was a guy named Martin John that never spoke English. My grandfather, he could speak English well, but he lived off the resources all his life. Do you know how he died? He died from the common cold when he was 90. That was the only sickness he had. He had perfect teeth and all his hair.

Our values included what we were going to be. We had no fridges and no stores in my community. So I knew everyday there was something new with my dad. If we wanted fresh clams we would have to walk a mile and a half, two miles to get them. My people knew the value of eelgrass. In Hesquiat Harbor, there used to be a massive spawning area there. We had a lot of problems with crab fishermen and long-liners. All they do is apply to the Department of Fisheries and Oceans for a license. There is no education process today about the values that make up that territory when that person goes for a license. There is nothing. All they do is buy a license.

When they first did gillnetting of herring, we went to one of the old chiefs, Felix Michael, and took him to the beach on the first day of the herring fishery. We wanted to see how he was going to react. There were hundreds of nets in his territory that has fed his people for a thousand years. He asked, "What are they doing? Don't they know that the herring are going to spawn and we are going to live off them?" When they go for a license, there is no understanding of value. In that place, Queen's Cove, we used to go to the Chief's territory and see him because behind the reserve was a huge pool, a mile long and a metre deep. There was so much herring there when I was young that we would just take a bucket and scoop them up. That is what he always saw. When that old man died, they took so much herring out that the herring stopped going there.

There is an area in Nuu-chah-nulth territory where three tribes use the banks. What the name of the bank refers to is that there was so much halibut in that bank that every evening the halibut came out of the water to flap their tails in the water. There was so much herring and everything else there. The foreign fishing fleet thought they would never destroy it, but they did. La Perouse Bank became extinct. There was a time you could travel on that bank which was 27 miles from Ucluelet. What happened to that area? Unfortunately our people's intelligence was not well regarded in those days. We have inlets that relate to that area because we know where the fish go.

I lived in Hot Springs Cove. Once in a while the halibut come right close there so we know where they go. There was a time when I went from Hot Springs to a reef where we could get cod and different kinds of snappers. We cannot do that anymore. What they did not realize when they invented drag technology is that they could catch fish in deep water. Technology is a very dangerous weapon to fish. I know because I was in the commercial fishery. I learned the traditional way of fishing, which is traveling along the coast, but when the fishery on sockeye became heavy, people started to phone San Francisco to know what the water temperature in British Columbia will be. The migrating salmon stocks tend to bite at 58 degrees Fahrenheit. When that was found out, they did not have to take their time looking for stocks. They knew where to go immediately.

There was an archaeological dig several years ago. There is an assumption by scientists that if something goes a little wrong, it is terrible. In Addison, they found bluefin tuna in an archeological dig. The people say that our temperature used to change. We had bluefin right in Zeballos. Another tribe said that there were bluefin there. The scientists believe that now, but it would have been nice if they had talked to the First Nations beforehand to see what went on in their territories.

I am seeing erosion of the values that we had. Our elders believe that the herring stock was one of the most important stocks to the ocean. It fertilized the bottom and fed all of the different species that went through our part of the world, like rockfish. So if I had herring, I also had fish that ate herring. Our people were involved in the fishery even though we understood that something was going to go wrong. Fortunately, the Department of Fisheries and Oceans supported our argument in Hesquiat. We wanted no-take on herring in Hesquiat Harbor. We did not need a Marine Protected Area. It is the common mind. We are the ones who understand what is going on. Creating a Marine Protected Area is good, but who is going to enforce it? Who is going to watch it? In our system, I have an oceanfront. In the days of old, like in the 1870s or 1880s, it was still exercised. No one could fish in the area that belonged to me, especially those from other tribes, but people from my own tribe could. Then we were faced with having to be economically competitive in this world so we started to harvest everything we could without conscience.

The other thing that is important in terms of value is what our people are now talking about, which is how much we have changed since Father Perbont came. We did not know he would have so much impact on us. My tribe bought flour there. The ladies were excited about the sacks. They dumped the flour and kept the sacks. They did the same with potatoes - they dumped it right there on the seaweed. After they did that, our people said that there were huge potatoes that grew right on the shoreline. In terms of what has happened to us since then - and it is important for you to know why values are important - our people are leading in every sickness in British Columbia. We went from eating fresh food to eating canned meat every day. This happened over a very short period of time, 150 years compared to the ten to fifteen thousand years that we have been eating fresh seafood. In an archaeological dig that went back 5000 years, they discovered that nothing had changed over the years. All the skulls still had their teeth. The only thing they found was a trace of arthritis. One guy got a hold of our blood under the pretence of studying arthritis, but we are getting the blood back now. But I do not want to give you the impression that what you are doing is not valued.

The leadership is saying that we value technology but we want to combine it with the traditional values. When you look at me and ask where I come from, I come from the West Coast of Vancouver Island. There are lots of things that bother me there now.

We used to be the dominant species over the things that moved in Nuu-chah-nulth. Now there is another dominant species: sea otters. Because of them, there are no more clams and no more urchins. When they replanted sea otters, they did not ask us how to control them. We have decided to harvest the sea otters, but we have to do it in a humane way. There is a loss of sea otters and sea lions around the world, but in our territory there are too many of them. There are problems with people who think differently but who do not live there. We are having problems with the animal rights people who say sea otters have a right to live. Well, we have a right to live too. We do not want people to forget that there is a human aspect to whatever decision is made. We want to be part and parcel of the decisions about our home.

Nuu-chah-nulth is setting a precedent. For 5 years we have discussed what we can do protect the resources in our territory. We had chiefs who realized we could not do it alone, so we got some non-natives involved. We respect their values. I think that we have some ways to go. We went from riding in canoes to fishing in seine boats. We know what happened between the canoe and seine boat. I hope I have given you some information on why values are so important to us.

The Editor did not wish to disrupt Chief Lucas' narrative with citations, but readers may be interested in reading more about some of the topics raised in the following papers in this volume.

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