INTRODUCTION TO THE METHODOLOGICAL CHALLENGES IN 'BACK-TO-THE-FUTURE' RESEARCH

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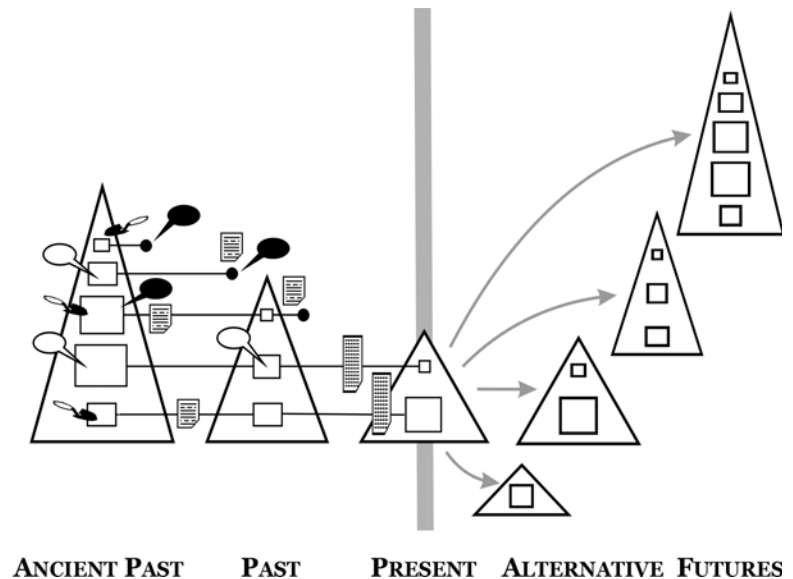*Fisheries Centre, UBC*

ABSTRACT

Many of the concepts in the Back-to-the-Future research process are new and so new methods, and modifications to existing methods, have been required for analysis, modelling and prediction of marine ecosystems and their fisheries. Methodological issues have been encountered in describing and modelling past ecosystems, in devising an ecosystem approach to determine sustainable fisheries, in devising a rational basis for choosing appropriate restoration goals and in attempting to maximise consent and compliance through encouraging a sense of ownership of policy by stakeholders. This paper summarises these issues, and introduces each of the new methods later to be described in detail in papers in this report. Results from case studies of the BTF process are contained in a separate report.

INTRODUCTION

Back-to-the-Future (BTF) is a science-based restoration ecology aimed at the creation of truly sustainable food and wealth from capture fisheries and aquatic ecosystems (Pitcher *et al.* 1999). The fisheries are embedded in aquatic ecosystems that, by quantitative analysis and with the agreement of stakeholders, trade-off wealth and food with a specified degree of retention of their unexploited biodiversity and trophic structure. Hence, BTF uses past ecosystem states as candidates for adoption as policy goals for the future (Figure 1, Pitcher 2001). In practice, the policy goals are subject to a number of practical constraints from species, habitat and climate changes (Haggan *et al.* 2003). The six logical steps in the BTF process are outlined in Table 1 (Pitcher 1998, 2004a, Pitcher *et al.* 2003).

Many new concepts have been developed as a part of the BTF research sponsored by *Coasts Under Stress* (CUS), and so it is not surprising that existing methods have not been adequate to express them. This report contains descriptions of the new methods that have been developed, along with papers of a general methodological nature from CUS research partners. BTF case studies and results are the subject of a separate publication.

The new methods can be divided into four groups: methods required to describe and model past ecosystems, ecosystem-based methods to determine sustainable fisheries, methods that set out a rational basis for choosing appropriate ecosystem restoration goals, and finally, practical techniques that attempt to secure compliance and consent through participation.

1. METHODS OF MODELLING PAST ECOSYSTEMS

The present-day ecosystem is represented by mass-balance and dynamic simulation modelling (at present using *Ecopath* with *Ecosim*; Walters

Figure 1. Diagram illustrating the 'Back to the Future' concept for the restoration of past ecosystems. Triangles at left represent a series of ecosystem models, constructed at appropriate past times, where vertex angle is inversely related and height directly related to biodiversity and internal connectance. Time lines of some representative species in the models are indicated, where size of boxes represents relative abundance and solid circles represent local extinctions. Sources of information for constructing and tuning the ecosystem models are illustrated by symbols for historical documents (*paper sheet symbol*), data archives (*tall data sheet*), archaeological data (*trowel*), the traditional environmental knowledge of indigenous Peoples (*open balloons*) and local environmental knowledge (*solid balloons*). Alternative future ecosystems, restored 'Lost Valleys', taken as alternative policy goals, are drawn to the right. (Diagram modified from Pitcher *et al.* 1999 and Pitcher 2001.)

ANCIENT PAST   PAST   PRESENT   ALTERNATIVE FUTURES

Table 1. Stages in the 'Back to the Future' process for the restoration of fisheries and aquatic ecosystems. Workshop phases are in italics. Modified from tables in Pitcher (1998) and in Pitcher *et al.* (2003).

Stage	Goals	Steps
1	Model construction of past and present aquatic ecosystems	Assemble present-day mass-balance and ecosystem simulation model
		Assemble preliminary past models using compatible structure and parameters
		Search and score data archives, historical documents, archeological information
		Workshop of scientists knowledgeable about system
		Interviews for traditional environmental knowledge, and for fisher's opinions and behaviour
		Assemble and standardize historical and interview scores database
		Assemble and test suite of ecosystem simulation models
		Workshop of scientists and managers to compare and standardise ecosystem models (may need to return to this step after preliminary results)
2	Evaluation of ecological, economic and social benefits that could be gained from each system	Determine sustainable fisheries with which to exploit reconstructed ecosystems ('Opening the Lost Valley')
		Challenge model scenarios with uncertainty
		Challenge model scenarios with climate changes
		Ecosystem simulation scenarios under anticipated conditions
		Workshops to evaluate policies with fishing communities
		Critique and evaluate 'Lost Valley' fisheries scenarios and adjust where required
		Searches for optimal mix of fishing gears
		Determine Optimum Restorable Biomasses (ORBs) for 'Lost Valley' scenarios
		Quantify risks to ORB policies
3	Choice of system that maximises benefits to society	Identify trade-offs among economic, ecological and social criteria
		Ecological economic evaluations including analysis of risks
		Workshops with communities, managers, scientists, NGOs, and government
		Participatory policy choice
4	Design of instruments to achieve this policy goal	Model exploration of MPAs, effort controls, acceptable quotas, times and places for fishing
		Evaluation of costs of the desired management measures
5	Participatory choice of instruments	Community and stakeholder discussion and choice of instruments to achieve policy goals
		Workshops with communities, managers, scientists, NGOs, and government
		Participatory policy choice
6	Adaptive management: implementation and monitoring	On-going monitoring, validation and improvement of model forecasts using adaptive management procedures
		On-going participatory guidance on instruments and policy goals

et al. 1997) using techniques that have received a degree of approval by marine ecologists (e.g. Whipple *et al.* 2000). This modelling is a far from trivial task, especially if fitting to time series of fisheries and survey data is undertaken. Moreover, highly migratory species like salmon, that exhibit lifetime shifts between different ecosystems, are included in ecosystem models with difficulty (see Martell 2004, this volume).

Models for past ecosystems are assembled using scientific archival data, archeological data, historical information, and local and traditional environmental knowledge. Scientific data derive mainly from published scientific papers, although material from unpublished reports and archives can often be valuable. Archaeological data has a similar set of sources (see Orchard and Mackie 2004, this volume). Historical information is gathered mainly from relevant books, letters, trade accounts and other historical documents, although, unlike science and archaeology, where searchable databases are the norm, finding and locating historic material can be quite hard. In some cases, translations are required. Local and traditional environmental knowledge, on the other hand, is rarely published and often has to be derived largely from oral sources through interviews and discussion held in coastal communities (see papers by Ainsworth, Simeon and Pitcher *et al.* 2002c, this volume).

Once found, all these data have to be assembled into a relational database together with evaluations of its scope and quality, to ease retrieval of relevant information for the models. (The CUS BTF project database will be described by Erfan in a later report.) Even so, a significant task is systematising the way in which information is collated for use in the models. The reason is that, once documented, information has to be expressed in a form that can be used in building ecosystem model structure, in setting parameters, or in shaping dynamic responses to changes. Although presence and absence of a species is easily dealt with, the models require us to know actual biomasses, size and growth parameters, and items in the diet.

Information about the local fisheries, with analyses and surveys, and about local aquatic fauna and flora is relatively easily found, especially as an output of 'science workshops' comprised of research partners and local scientists with expert knowledge of the area and the taxonomic groups. One of the principal

problems here is data that has been gathered on either a very small or a very large scale compared to the area of focus (see Haggan 2004, this volume). Another issue often requiring a lot of work is the concordance of measurement units, since specialists on different taxa often work in very different fields. Scientists who generously make the relevant information available, often from a lifetime's work on a group of organisms, are encouraged to publish a paper in one of the BTF reports so that they retain a recognised ownership of material that otherwise would easily vanish into model simulations.

For the CUS BTF project in Newfoundland and British Columbia, the output from an extensive process of consultation with the science community has been presented in detail in four reports (Ainsworth *et al.* 2002, Pitcher *et al.* 2002a, 2002b, Heymans 2003), where information essential to the modelling process, such as geographical scope, biomass, relative fishing mortalities, diets and other ecological information are assembled.

In the absence of local publications on these topics, as is often the case, interviews, conducted under suitable partnership agreements, are the best way to gather LEK and TEK information for use in the modelling. Ainsworth (2004, this volume) reports on methods used in interviews designed especially to gather material that can be used in ecosystem modelling for the CUS BTF project. A report on a community workshop is presented in Pitcher *et al.* (2002c, and see Power *et al.* 2004, this volume).

For ease of comparison, the structure of the past and present ecosystem models should be similar, although of course biomasses and fluxes can be vastly different. Global extinctions of species cause some technical difficulties in modelling. When species have gone locally extinct ('extirpation'), this creates some difficulties (see Pitcher 2004d, this volume). Some practical solutions found in the CUS project are presented by Heymans and Pitcher (2004, this volume).

Another frequent problem is that reconstructions of the ancient past may suggest the presence of large numbers of top predators that are too numerous to be supported by what are thought to be realistic levels of forage organisms (Pitcher 2004c, this volume).

Representing changes in ecosystem structure over long periods of time represents a major challenge. Clearly, the effect of shifts in climate has to be accommodated in the forecasts as much as

possible (see Pitcher and Forrest 2004, this volume). But early periods of depletion by human exploitation also had significant impacts on ecosystem structure and function. Recent reconstruction work by Jackson *et al.* (2001) shows what may be possible in this respect.

Ideally, the timing of the series of ecosystem models for BTF may depend on the locality, the dawn of quantitative documentary evidence, and major shifts in resource and ecosystem history such as the introduction of new fishing gears, damming of rivers and collapses of fish stocks. But because of the large amount of work involved in drawing up each ecosystem model, the gaps in time between a series of BTF models may be quite large. So an ideal choice of the time snapshots to use as BTF models is generally constrained by the resources available for the research. This raises a significant methodological problem in that failure to cover important changes that occurred within these time gaps can prejudice the choice of appropriate policy goals at the end of the BTF process. In the event, the choice of the time periods to model in a BTF analysis is something of a compromise.

In many cases, additional informative models might be drawn up for pre-modern humans in the late Pleistocene post-glacial era. Although such ancient ecosystems would be unlikely to ever become practical policy goals, they have the advantage of providing a 'pristine' baseline against which all more recent changes might be assessed. In fact, for some areas of the world only recently colonised by Europeans, such as Australia, New Zealand and the Pacific coast of America (Diamond 1997), models of 'pre-contact' ecosystems may serve this purpose well.

In models of the distant past, the estimation of the size and impacts of ancient fisheries presents many problems. Although the history of fishing technology is quite well known from archaeology and from traditional knowledge, its likely fishing power may be estimated, and ancient diets may be calculated, nevertheless, the size of the human populations that engaged in fishing is often hard to assess. Estimates of ancient human population sizes are often the subject of controversy among archaeologists and anthropologists. In one of the recent volumes from this CUS BTF project, Heymans (2003) presents an example of what may be done with ancient diets and fisheries. It is emphasised, however, that the aboriginal fisheries in the ecosystems are described only to provide an accurate picture of the ancient ecosystem, and they would not necessarily be chosen for a future restoration policy. This issue

is discussed in more detail below.

Finally, many of these problems may be eased if we were able to run a past model forward to simulate its change into a more recent ecosystem. Performing this using *Ecosim* requires a great deal of data on fisheries and climate (see Stanford 2004, this volume), but has been possible for some ecosystems that have undergone rapid change, such as the Gulf of Thailand (Christensen 1998). Unfortunately, to date, attempts to do this with both BC and Newfoundland ecosystem models have been only partially successful (Heymans 2003). Heymans and Pitcher (2004, this volume,) summarise the construction of models of the past in relation to the ecosystems researched for the CUS BTF project.

2. METHODS FOR DEVISING SUSTAINABLE FISHERIES

A marine ecosystem restored to some semblance of its past state might be thought of as a 'Lost Valley'[1]: an ecosystem discovered complete with all of its former diversity and abundance of creatures (Pitcher 2004b, Pitcher *et al.* 2004). The BTF process aims to describe a series of such 'Lost Valleys' as a set of potential restoration goals.

Since a 'Lost Valley' has to be fished sustainably, we have to ask how this might be achieved? Using the same fishing fleet as today in order to fish a restored ecosystem is generally not a viable option since massive depletion would soon ensue. Nor is it realistic to expect the fishing gear and methods of former times, including those of 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. The BTF process aims to devise such criteria. For example, a candidate fishery designed with the criteria could be challenged by assessing its conformity with the FAO Code of Conduct for Responsible Fisheries (FAO 1995) using a rapid appraisal technique (Pitcher 1999).

After applying the criteria in this way to design an 'ideal fishery' for a particular location, ecosystem simulations (using the *Ecosim* policy search interface; Walters *et al.* 2002) can be used to find the relative fishing mortalities that should be used by each gear type in the 'ideal' fishery to achieve sustainable catches over a long time period, usually 100 years.

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 (see papers by Ainsworth *et al.,* Pitcher and Forrest 2004, this volume).

3. METHODS FOR CHOOSING ECOSYSTEM RESTORATION GOALS

Once we have snapshot of what a set of alternative restored ecosystems, complete with their sustainable fisheries, might look like, the remaining issue to solve is to find an objective way to choose a rational policy goal from among them. This may be done by comparing the benefits that will accrue to society from each alternative future represented by a fished 'Lost Valley' ecosystem. In order to show the full range of options that may be considered, included in this process is the present day ecosystem (albeit with fisheries designed to be sustainable), and perhaps an ecosystem even further depleted (Figure 1).

One fundamental way to evaluate the benefits of alternative restored ecosystem is the net present economic value of their fisheries, information that is readily estimated from the *Ecosim* simulations mentioned above. A modification more in accord with ecological economics is to estimate present value using intergenerational equity calculations (see Ainsworth and Sumaila 2004a, this volume).

Purely economic considerations, however, are rarely considered sufficient for modern policy making. Therefore, in the BTF process we also estimate the relative impacts on biodiversity (see papers by Ainsworth and Pitcher, Heymans, and Chueng and Pitcher 2004, this volume) and social factors such as the likely number of jobs and their diversity (see Ainsworth and Sumaila 2004b, this volume). For a proper evaluation, the costs of restoration have to be considered alongside the benefits. This part of the evaluation system is not yet completed for the CUS BTF research and the issue is discussed further below.

4. PARTICIPATORY AND ADAPTIVE POLICY IMPLEMENTATION

Implementing a policy goal that has been chosen using any science-based process, including BTF, is, of course a much more difficult matter. When fishing communities and other essential stakeholders actively participate in the policy

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[1] We are grateful to Dr Daniel Pauly for suggesting this term in 2001. (See Pitcher *et al.* 2004, this volume)

Table 2. Summary of integral participatory elements from local fishing communities in the BTF process. TEK = traditional ecological knowledge, LEK = local ecological knowledge. All stages are intended to work in concert with science-based decision making.

Model development phase	TEK: in model construction
	LEK: in model construction
	TEK/LEK/Community: model credibility and validation
Policy development phase	Community choices: how to rebuild
	Community choices: choice of best benefits to cost ratio for policy goal
	Community choices: choice of acceptable and sustainable fisheries
Operational phase	Consent and compliance
	Monitoring

agenda, compliance and consent may be high (Hart and Pitcher 1998). For example, Haggan (2000) identifies 4 elements as critical to participation: recognition of the scope of the problem and our collective responsibility whether fishers, scientists, managers or policy makers; respect for different systems of knowledge; agreement to share knowledge in the interest of conservation and restoration; and, commitment to share in the benefits of restored systems.

In BTF the aim is to encourage a greater chance of success because a sense of ownership of the process is fostered and developed from the earliest stages of the work. The BTF process includes community participation in building models of the past (see Simeon 2004, this volume), in the choice of sustainable fisheries and in the evaluation of the costs and benefits of alternative restoration goals (see Power *et al.* 2004, this volume, Pitcher *et al.* 2002c). Moreover, the cognitive maps shaped by awareness of past abundance and diversity develop in BTF process may serve to assist consent and compliance with a restoration agenda (Pitcher and Haggan 2003). Participatory elements that are integral to three phases of the BTF process are summarised in Table 2.

Once management aims to make progress towards a specific BTF past state, the use of quantitative adaptive management (e.g., Walters 1986) is the wisest course, in order to try to avoid the disasters that a changing environment and imperfectly understood ecology can throw at any management plan.

CONCLUSIONS

Policy goals that reflect an approach of restoration ecology may be chosen using the BTF procedures outlined here and presented in more detail in subsequent papers in this volume. But a number of methodological challenges raised by BTF remain unresolved at this stage.

The way in which historical information is turned into inputs for the ecosystem modelling could do with considerable improvement. Better semi-quantitative assessment of relative biomass, diet and sizes needs to be devised. Our historical data need a more rigorous and replicable transduction into the quantitative data needed for modelling. For the CUS BTF research, a first step in this respect will be published by Ainsworth (2004) in the forthcoming 'results' volume.

BTF has an advantage in not relying exclusively on complex stock assessment (Walters 1998), although such work can help in the tuning of the ecosystem models. At present, the quantitative ecosystem modelling used for BTF to date relies almost exclusively on *Ecopath* and *Ecosim* techniques. Yet many of the assumptions in this modelling system, while plausible, remain unvalidated. Of especial concern are the *Ecosim* 'vulnerability' parameters, to which specific results often appear very sensitive (see Ainsworth 2004, this volume). Moreover, these parameters not only shape predator-prey interactions (which they do in an entirely credible fashion for a former evolutionary ecologist), but also pre-determine the scope for further biomass growth in relation to current abundance. For any series of 'time-shot' BTF ecosystem models, this creates a conflict between the need to compare the outcomes of various fisheries options while other parameters remain fixed, and setting parameters correctly for biomasses that were closer to unexploited levels in the past. These modelling problems have yet to be resolved.

As pointed out by Heymans and Pitcher (2004, this volume), past ecosystem models may resemble the actual past as a Picasso resembles reality. An important question is whether our comparative restoration policy scenarios can be made robust against such distortions. A deeper insight of the dynamics of ecosystems under change will be required before we can answer this question.

A broad participation by scientists, researchers, stakeholders, government, managers, NGOs and the public is critical for the success of any restoration policy that might be set up under the BTF banner. Yet we have barely scratched the surface of the deep issues raised by the need for this level of participation in the BTF policy searches and analyses. Nor have we enough experience of asking fishing communities to choose what kind of future they might wish to aim for. We are not yet sure how to convey the uncertainty in our work, which to many may seem arcane. Perhaps 'barefoot ecologists', the equivalent of rural development generalists for

fisheries, as envisaged by Jeremy Prince might be able to help (Prince 2003).

The intention is to give BTF players a clear cognitive map of a future ecosystem that resembles one from the past, to which all may agree and aspire (see Pitcher 2004a). And so, to date, BTF analysis has not considered the costs of achieving each restoration, because this may divert attention from that ultimate goal. (Although it is noted that it may be logically argued that the true policy goal cannot be known until a full cost-benefit analysis is performed.) The fundamental problem here is that estimating the costs of restoration may depend on precisely what techniques are adopted, and the actual instruments may themselves generate conflict (for example, MPAs set up adjacent to a traditional fishing community – see Lucas 2004, this volume - or reduced quotas for some sectors as fisheries are modified to become more sustainable). Again, these important issues remain to be resolved.

The logistics of mounting a quantitative, robust and credible BTF analysis are considerable. The sheer cost, in money and time, of assembling an inter-disciplinary team to gather, validate and analyse the historical, archaeological and ecological information needed for BTF is formidable. Moreover, like other synoptic work involving whole ecosystems, the scope of BTF work appears to be far outside the capacity of one graduate student thesis. In this project, it has therefore been gratifying to see modest financial support from *Coasts under Stress* augmented by enormous enthusiasm and commitment from the team of graduate students, postdoctoral researchers and research partners who have helped with the research reported here.

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Synoptic Methods for Constructing Ecosystem Models Of The Past

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Abstract

This paper gives a brief description of the steps that need to be taken when constructing a model of a past ecosystem. It is important to know what question is going to be asked of the model, as that affects all subsequent steps. To construct a model of the past it is important to know the area, time periods, species to include, what data is available, how to handle the calculation of *Ecopath* parameters for species that have a different age structure from the present day, and finally how to make and test the assumptions needed in such a endeavor. Assumptions that have to be made lead to uncertainty, which may be examined using the emergent properties of the ecosystem.

Introduction

Models of the past are constructed for comparison with present day models. They provide baselines for the emergent properties of these ecosystems. For the *Coasts under Stress* Back-to-the-Future project we aim to assess the effect of long term trends in the social and environmental health of regional ecosystems on the environment and on human health. The question asked was:

> 'How can local ecological and scientific knowledge help us to understand changes in environmental, community, and individual health in ways that will help develop better strategies for future ecological recovery?'

In this paper the methodology of constructing models of the past will be illustrated by using two examples from the CUS BTF project: Northern British Columbia (including the Hecate Strait and Queen Charlotte Sound), and Eastern Newfoundland /Southeastern Labrador (NAFO Div. 2J3KLNO) (Figures 1 and 2), as defined in Pitcher *et al.* (2002) and Ainsworth *et al.* (2002).
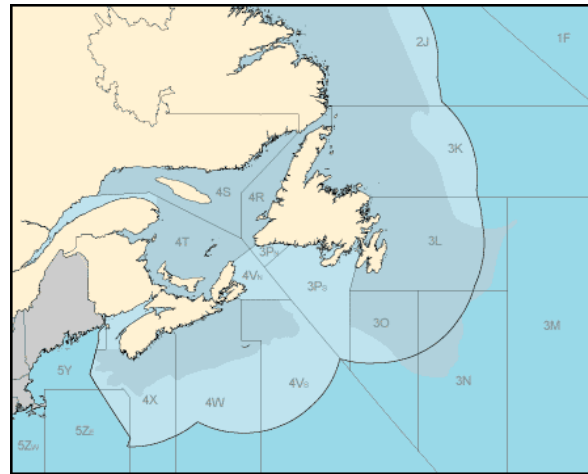
Figure 1. NAFO divisions of the east coast of Canada, showing the areas used in the CUS-BTF study (Divisions 2J, 3K, 3L, 3N and 3O).

Methodology Used in Constructing Models of the Past

The steps involved in the construction of past models include: 1) defining the system, 2) choosing the time periods you want to model, 3) data gathering (on catch and biomass mostly), 4) which species to include, considering extinctions and incorporating species that are not well studied even at present, 5) calculating the energetic ratios for models of the past, and finally 6) making other assumptions for species where we have no better information.

Defining the system

To define a model of the past you have to define the boundaries of your ecosystem. The chosen system should preferably be contained in a natural or oceanographic feature, with a single climate. Generally a larger area is preferable as it increases the chances of having any historical information. By and large the international jurisdiction of the area does not matter, as the jurisdiction would have changed over the course of time. Usually the ecosystem is defined based on current knowledge of the system. For instance, in both the northern BC and Newfoundland models we defined the system based on current knowledge and more recent models constructed for these ecosystems (Heymans and Pitcher 2002a, and Ainsworth *et al.* 2002).

In Newfoundland the area chosen (Figure 1) was similar to that used in prior models of the system (Bundy *et al.*, 2000, and Heymans and Pitcher 2002). The area chosen included the DFO-NAFO divisions 2J3KLNO and incorporated the Labrador Current and the Grand Banks, as they

Figure 2. Map of the West coast of Canada showing the Hecate Strait and Queen Charlotte Sound, both in the study area.

are interconnected and for some species they are managed as a unit. The area chosen for the northern BC model included both Hecate Strait and Queen Charlotte Sound (Figure 2). The area was chosen to answer particular questions, thus some inshore marine waters were included in the model area as salmon had to be included.

Choosing time periods

When choosing a time period it is advisable to choose times pre- and post pivotal gear changes or exploitation levels. Time periods pre- and post the start of formal recorded data are often important, and, finally, the time periods depend on the questions that are asked. For the CUS BTF project the question was to assess the longer term trends in the health of local and regional ecosystems.

The time periods chosen for the northern BC model were 1750, 1900, 1950 and the present day. (Ainsworth *et al.* 2002). The 1750 model was chosen as it was pre-European contact, while 1900 was prior to large scale commercial fisheries and the resumption of whaling. By 1900 the number of First Nations people were drastically reduced, which had a positive effect on Steller sea lions, although the sea otters were already locally extinct by that time. The 1950 model incorporates the large scale purse seine fishery for herring, the

collapse of pilchard and the start of DFO's catch data series, while the present day (2000) model was initially based on a model constructed by Beattie (2001) but using more recent data.

In Newfoundland the time periods chosen were 1450, 1900-1905, 1985-1987 and 1995-1997 (Vasconcellos *et al.* 2002). The 1450 model was pre-European contact, 1900 was prior to the large scale Grand Banks fisheries, but after the large scale whaling that took place in that area. The 1985-1987 model was based on the model constructed by Bundy *et al.* (2000) and was prior to the groundfish collapse based on reliable data, while the 1995-97 model was after the groundfish collapse but did not have the same quality of data as the 1985-87 model.

Data gathering

Information on past abundances, catches, etc. are not easy to obtain in normal scientific literature. However, building models of the present day gives a blueprint for models of the past. Information on past abundance and catches are generally found by looking at historical documents, or by using expert opinion of fisheries biologists on virgin population of key species. It is also possible for marine mammal or seabird experts to make 'educated guesses' on how many animals must have been in the system at a certain time. There are also archaeological and anthropological information available to assist with presence/absence of species, as well as the utilization of marine species by First Nations or European settlers. Finally, Traditional and Local Ecological Knowledge (TEK/LEK) can also be useful for models that are within their time frame, i.e. models that go back about 50 years.

Data for building the models of Newfoundland (Heymans and Pitcher 2002a, 2002b, Pitcher *et al.* 2002a) were obtained mostly from historical documents and books that summarize changes in that ecosystem: (Lescarbot 1914, Howley 1915, Lewis and Doutt 1942, Wright 1951, Mercer 1967, Mowat 1984, Reeves *et al.* 1985b, Crosby 1986, Montevecchi and Tuck 1987, Cushing 1988, Pastore 1992, Hewitt 1993, Ryan 1994, Pope 1995, Turgeon 1995, Marshall 1996, Lear 1998, Hiller 2001, Cridland 1998, Whitridge 2001). For the calculation of the pristine population and catch of cod, a reconstructed time series obtained from (Hutchings and Myers 1995) was useful. The Internet was useful for obtaining information on historical populations. In Newfoundland the Heritage Site of Newfoundland and Labrador (www.heritage.nf.ca) contains information on the fishing industry, First Nations and European

settlement in the area.

Data for building the models of northern BC (Ainsworth *et al.* 2002) were obtained from historical documents, such as Hudson's Bay Company records (Hammond 1993), as well as other historical records (Lord 1866, Chambers 1872, Anderson 1879, Dawson 1880, Mowat 1886, ANON 1892, Osgood 1901, Freeman 1904, Babcock 1910, Alexander 1912, Thompson 1916, Newcombe 1917, Muir 1935, Carrothers 1941, Akrigg 1975, Kenyon 1975, Jacobsen 1977, DeWhirst 1982, Vancouver 1984, Reeves *et al.* 1985a, Webb 1988, Gregr 1999, 2002, Mackie *et al.*, 2001). Data for building the model of 1950 was also obtained from interviews done in Prince Rupert and surroundings (see report in Pitcher *et al.* 2002). Both historical data and interview data for this system was collected in a database searchable on the web at: www.fisheries.ubc.ca/projects/btf/ (see Erfan, results volume).

Data on sport fish catches are rarely recorded in official 'catch statistics', but can be considerable (e.g., Pitcher 2003, Pitcher and Hollingworth 2002). In Northern BC some estimates have been made using interview and other techniques (Forrest 2002).

Data on catches made by First Nations are generally hard to find (e.g., Irwin 1984). However, in northern BC an estimate of salmon catches by First Nations were made by (Hewes, 1973) and assumptions had to be made for the catch of eulachon and marine mammals. In Newfoundland the catch of marine animals by First Nations was hard to calculate, as the Beothuk people of Newfoundland were extirpated by 1829. Assumptions had to be made about how much the people of Newfoundland would have eaten. With the help of an Ingeborg Marshall, an anthropologist from St. Johns, their consumption of marine resources were calculated by apportioning likely catches between marine mammals, salmon, and other marine resources ((Renouf 1999, Marshall 1996, Heymans 2003).

Which species should be included?

The species to be included usually depends on the question asked, what species have gone extinct, locally or globally, and what species migrate through the system. The question asked implies that some species would be important as single groups in one model vs. being able to combine them in other groups. For instance, in Newfoundland it was necessary to put Greenland cod and lobster into their own compartments, as

the question asked pertained more to the human interaction and inshore system than to the offshore system.

Likewise, the importance of migratory species such as migratory salmon and transient killer whales become more important in the northern BC model, as these are important in the policy arena of that system. There are two other important considerations that need to be taken when deciding which species to include, namely extinctions and species that are not well studied.

Extinctions

Local and global extinctions make the inclusion of certain species very difficult. For comparison between emergent properties of ecosystem models it is important for the groups to have the same number of compartments. Similarly, simulations that span two different models would need all the compartments to be included in both models. Thus, it is important to include species that have become extinct during the course of the modeling exercise. These species are usually included by adding a very small biomass ($1*10^{-6}$ t.km^{-2}) in the models where they are essentially extinct (see Pitcher 2004, this volume).

An example of a local extinction in northern BC is the sea otter, which became extinct before the 1900 model. Pristine population estimates are given by (Kenyon, 1975), and were used for the estimation of sea otter biomass in 1750, but by 1900 and the subsequent models of 1950 and 2000 biomass was assumed to be $1*10^{-6}$ t.km^{-2}.

Three species have become extinct in Newfoundland since European contact: walrus and grey seals have become locally extinct, while the great auk is globally extinct (see Pitcher 2o04, this volume). No estimates of walrus or grey seal biomasses were available for 1450, but estimates of rookery sizes and whelping patches were given in the controversial book by (Mowat, 1984), which had to be used in lieu of any other data. By 1900 both these species were locally extinct in Newfoundland, and their biomass estimates were therefore assumed to be $1*10^{-6}$ t.km^{-2}.

The extinction of the great auk was easier to model. Although there were at least 100,000 nesting pairs of great auk in Newfoundland at the time of European contact, they were extinct by 1830 (Burke *et al.* 2002, Sarjeantson 2001, Montevecchi and Kirk 1996). However, seabird biomass and impact is so small that they are usually grouped into functional groups. The great auk was therefore grouped with the piscivorous

birds, and as such no assumption had to be made about their biomass, other than the assumptions made for bird biomass in general (see Pitcher 2004, this volume).

Species that are not well studied

In all ecosystem models there are some species that are very poorly studied. Incorporating them is usually problematic, and very little data are usually available for non-commercial species. Examples of these species are the rockfish in northern BC (a guild of over 30 species) and Greenland cod in Newfoundland. There are many species of rockfish in northern BC, but until very recently, very little data was available on these species. In the present day model therefore, they were broken down into inshore rockfish, planktivorous and piscivorous rockfish. (Ainsworth *et al.* 2002, Foulkes in prep.). There are no historical estimates of biomass, production, etc. for these species, or for Greenland cod in Newfoundland, so their biomasses are estimated by *Ecopath* by assuming that their P/B and Q/B ratios would be similar in the past as they are today.

Energetic parameters

The other parameters needed for constructing *Ecopath* models are also be different in models of the past. Parameters such as the P/B and Q/B ratios are often smaller in populations that have many more older fish, that produce and consume less than a population that consist mostly of younger smaller fish.

The P/B ratio is usually assumed to be:

$$P/B = F + M \qquad (1)$$

where F is fishing mortality and M is natural mortality. Fishing mortalities in most models of the past are generally small, but where estimates of catch and biomass are available, they should be added to the estimate of natural mortality calculated below (Palomares and Pauly 1998):

$$\log M = 0.0066 - 0.279\,(\log L_\infty) + \\ 0.65431\,(\log k) + 0.4631\,(\log T) \qquad (2)$$

where L_∞ is the population asymptotic length of the Von Bertalanffy growth function (and is usually greater in populations of the past), k is the Von Bertalanffy growth parameter, and T is temperature in °C.

The Q/B ratio is calculated from an empirical formula published by Palomares and Pauly

(1998):

$$\log Q/B = 7.964 - 0.204\,(\log W_\infty) - 1.965T^* \\ + 0.083A + 0.532h + 0.398d \qquad (3)$$

where T^* is the temperature in °Kelvin, A is the tail aspect ratio (generally obtained from *Fishbase*), h = 1 for herbivores and 0 for all other groups, and d = 1 for detritivores and 0 for all other groups.

W_∞ is estimated from the length weight relationship:

$$W = a + L^b \qquad (4)$$

where the a and b parameters are obtained from *Fishbase*.

Estimating natural mortality and consumption parameters for juveniles are more challenging, therefore in most instances these parameters for juveniles were assumed to be 1.5 x that of the adults. Sometimes it was not possible to estimate both the P/B and Q/B ratios for a group, and then the gross efficiency (GE) was assumed to be 0.2 and the P/B or Q/B was calculated by *Ecopath*.

Assumptions

Constructing models of the past involves making many assumptions for biomass, catch, etc. Also, there is generally no data available on past diets and one has to assume that the diet in the past was similar to that of the present. Usually, when balancing the model the diet is the first parameter that is changed. Thus, starting with today's diet and assuming that most species are generalists that would feed on similar species, the diet of the past is changed to balance the model. The assumption that past diets were not very different is vindicated by a paper showing that, in field studies on Georges Banks (Gulf of Maine), diets of many species changed in proportion as much as would be expected from the change in abundance and species composition (Link and Garrison 2002).

CONCLUSIONS

Constructing models of the past is not an exact science. Often the model obtained would seem closer to a Picasso painting than to reality (Figure 3, Heymans and Pitcher 2002a). In an abstract Picasso painting the parts of the whole are all present, but are not realistic in proportion or placement, and this creates the interesting reaction desired by the artist. In a painting by a

great Renaissance master like Raphael, in contrast, things look as they do to the eye, although in fact subtle artistic artifice is employed to achieve this effect.

Similarly, an ecosystem model obtained by reconstructing the past incorporates most of the important groups and species that were present at the time period chosen, but the lack of information, and the quality of the available information influences the model. To counteract this problem it is advisable to describe the information and assumptions as well as possible, and to perform uncertainty

Figure 3. Paintings of attractive young women by Picasso and by Raphael. The Picasso creates a fascinating and shocking effect because all the right elements of the weeping woman (the artist's mistress) appear in a pastiche, but are placed in unexpected locations. The Raphael, on the other hand, appears as a luminous and accurate representation of a smiling young woman (the artist's wife), and whose artifice to deceive the eye is more subtle. The am is for ecosystem models of the past, perhaps initially comparable to a Picasso, to become more like a Raphael. *Pablo Picasso: Weeping Woman, 1937, etching/aquatint/drypoint. 49 x 69 cm, Museum of Modern Art, New York; Raphaello Sanzo: School of Athens (detail), 1510, fresco, 8 x 5.5 m, Vatican Museum, Rome.*

estimations where possible. Testing for the effect of different input data on the emergent properties of the ecosystem is a valuable way of checking uncertainty. This needs to be done for the models of the *Coasts Under Stress* BTF project. Additionally, putting the errors for the main parameters into the *Ecopath* model can help later when the ecosystem model is used in simulation mode and the effects of parameter uncertainties on alternative polices can be checked. The aim eventually is to have ecosystem models of the past that encourage the familiar comfort of a Raphael rather than the shock of a Picasso.
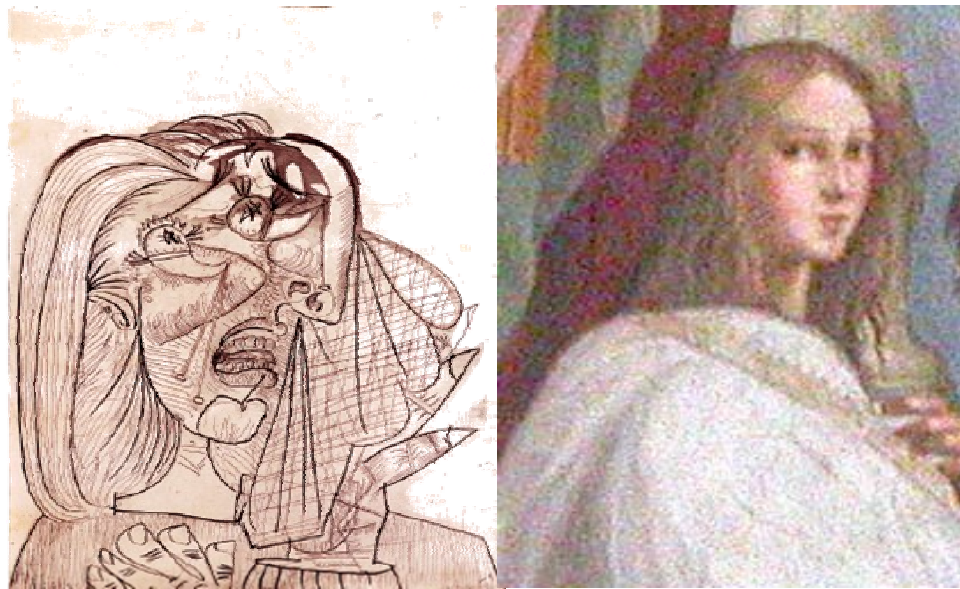
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WHAT WAS THE STRUCTURE OF PAST ECOSYSTEMS THAT HAD MANY TOP PREDATORS?

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Figure 1. Top predators, like this blue marlin, may have been so abundant in ancient ecosystems that a very large amount of prey forage fish was required to support them. This paper discusses the ecosystem modelling issues raised by this possibility.

ABSTRACT

Analyses of the ancient past, from historical sources, archaeology and reconstructions, suggest the presence of large numbers of top predators where few are found today. In mass-balance ecosystem models, these animals are not generally able to be supported by what are thought to be realistic levels of forage organisms. This paper examines the logic of this issue, and suggests that the problem may be resolved by evidence from archaeology and stable isotope analysis. In the past, more species may have occupied the forage fish niches and the diet of top predators near to carrying capacity may have been wider due to intra-specific competition.

Historical sources (e.g., examples in Mowat 1984) and attempted reconstructions (e.g., coastal ecosystems: Jackson *et al.* 2001; predatory fish: Myers and Worm 2003; sharks: Baum *et al.* 2002; whales: Roman and Palumbi 2003) all suggest that past ecosystems had many more large and long-lived top predators than we find today. Analysis of archeological remains also often suggests large predatory species where few are present today, for example, bluefin tuna along the whole western Canadian coast (Tunnicliffe *et al.* 2001, and see discussion page 139 this volume) and the North Sea (Mackinson 2001), and large old individuals of species in regions where they are represented by smaller, younger members today (e.g. cod and saithe at Skara Brae neolithic settlement, Orkney; Barret *et al.* 1999, Childe 1931, Clarke 1977). Moreover, compared to the present day, fishery exploitation was low in the ancient ecosystems (e.g., aboriginal fisheries in Newfoundland, Heymans 2003, Lucas 2004, this volume).

The issue in question here is that, when such large amounts of top predators are inserted into a mass-balance ecosystem model, a very large amount of amount of prey organisms is required as food to maintain all these animals. The
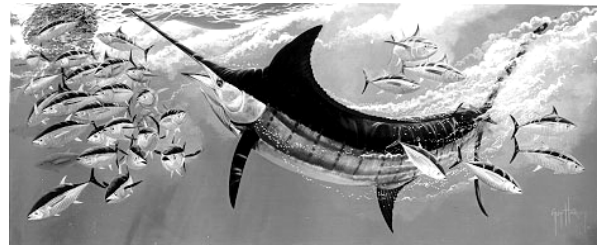
resulting biomass of forage animals is thought to be unrealistic compared to present day levels. We may ask if, in fact, this issue is some kind of artifact of the ecosystem modelling method, or a genuine conceptual problem?

On the modelling side, we may note that the P to B ratio of large old individuals of a species is far lower than the ratio characteristic of exploited populations today, and so adjustments in this respect are now routine in the creation of ecosystem models of ancient systems (see Heymans and Pitcher 2004, this volume, and e.g., Ainsworth *et al.* 2002). Nevertheless, even with reduced P/B ratios, surprisingly large forage fish biomasses can still result.

Some simple answers to the problem offer themselves first.

1. There were not so many top predators. The high abundance of top predators may actually be a false impression, based on anecdotes of those impressed by local patches of high abundance? For example, in the accounts of the first European visits to Newfoundland (Pope 1997, Williams 1996) we find what at first sight appear to be exaggerated references to cod so abundant that buckets full of the fish could be scooped up with little effort. Such reports may have been aimed, in part, at reassuring the late 15th Century financiers of expeditions to the New World that future gains would be considerable, as indeed they were. It is, however, a reasonable conclusion from the considerable archeological and documentary evidence that in ancient coastal ecosystems there were indeed large amounts of top predators, both in terms of species and in terms of large, old individuals, numbers and biomass. In addition to the references cited above, the work of Jackson *et al.* (2001) is perhaps the most significant in this respect.

2. A high biomass of forage fish is acceptable. A second simple answer is that a high abundance of

Figure 2. Discovered after a violent storm in 1850, Skara Brae, Orkney Islands, Scotland is the best preserved Neolithic village in northern Europe and offers a unique window into the lives of the fishers and farmers who lived there between 5,100 and 3,450 BP. Photograph shows a house with a stone dresser (rear wall) around which are three tanks for preparing fish bait. Middens from the site contain bones from huge cod and saithe (Barrett *et al.* 1999).

prey needed to feed these top predators may be actually acceptable. Biomasses in excess of 40 tonnes per km² are quite possible for small forage fish in upwelling or otherwise highly productive ecosystems (V. Christensen, pers. comm.). These fish may be highly productive, especially after a successful recruitment, feeding directly on blooms of phytoplankton and small zooplankton, with B/P ratios in excess of 3 in some cases. Although very high forage fish biomasses may be sporadic due to volatile recruitment, long-lived predators are presumably buffered against the fluctuations.

In some cases though, these simple answers may not be sufficiently convincing. Two more complex answers are discussed below.

3. *The diet of abundant competing predators was broader in the ancient past.* Populations of fish near their carrying capacity are not only comprised of large old individuals compared to exploited populations, but they are also characterised by high levels of intra-specific competition for food, space and other essential resources. This competition leads them to occupy all suitable habitat including the fringes of their normal range (MacCall 1990). For our purposes, the concept may be extended from the physical habitat to elements of their trophic niche. Competition at high population densities may lead to less successful individuals eating all manner of unlikely prey at the fringes of the normal diet. Hence, for this reason the breadth of

the top predator population's diet may have been much wider than under 'normal' exploited conditions under which data on diets has been gathered today. In a mass-balance model of the ancient past, therefore, diet might be broadened to more species of likely prey animals, reducing the high biomasses of any one species required to support the abundant predators.

4. More forage fish species were present in the ancient past. A similar argument concerns the number of species of forage fish present ancient ecosystems. Where today forage fish often occur in single-species 'wasp-waist' ecosystems, in the past more species may have been present. According to Odum's ratchet (Pitcher 2001), species with low P/B ratios become locally extinct first under the joint influence of climate and exploitation (Dulvy *et al.* 2003, Christensen and Pauly 1997). Even today, several species of less abundant non-commercial small pelagic fish co-occur with dominant species such as herring, capelin and mackerel. In some areas, the biomasses of small non-commercial forage fish are not even surveyed (e.g. sand-lance in British Columbia). Hence, today's species composition for this group of forage fish may not be a reliable guide to the food web that existed in ancient ecosystems. Since both #3 and #4 entail adjustments to the diet matrix of the mass-balance model, both arguments may need to be taken into account.

How can these issues be resolved? One approach is to look for archaeological evidence of the relative abundance of forage fish species (e.g., van Neera *et al.* 1999). Here, care must be taken to apply a series of strict rules concerning the interpretation of archeological fish bones as being representative of what was present in the wild in ancient ecosystems (see Orchard and Mackie 2004, this volume). For example, values may be distorted by selective fishing, by taphonomic factors affecting relative preservation status, and, since forage fish are generally small, ineffective screening of middens for small bones (see discussion page 138). In some cases, accurate modern analyses based on bone collections that were made in the past may be prejudiced by inadequate preservation, provenance or stratigraphy (i.e., "problems of collection, retention, curation and context", see Leach and Davidson 2001).

Another helpful investigation would be to examine the breadth of ancient fish diets using stable isotope analysis on archeological remains.

Finally, it would be worthwhile to investigate the effect of the structure and breadth of the forage fish diet of top predators more rigorously and systematically using the *Ecopath* auto-balancing facility (Kavanagh *et al.* 2004).

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For discussion following oral presentation of this paper, see page 149.

THE PROBLEM OF EXTINCTIONS

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ABSTRACT

The extinction of species causes problems when, to enable comparison of emergent properties, a series of ecosystem models constructed through time must have a similar structure. Global extinctions are irreversible and approximate representations of such species in models of ancient ecosystems relies on historical and archeological information about their ecology, diet and growth. As a short-cut to preserve model structure, extinct species may sometimes be grouped with species of a similar function in ecosystem models. Local extinctions ('extirpations'), on the other hand, are potentially reversible by natural recolonisation, or by human re-introduction. Ecosystem modelling therefore needs to be able to capture this reversibility by explicitly including such species. Currently, it is especially difficult to model the effects of keystone species, such as sea otters, whose biomass level directly alters habitat structure.

Global extinctions of species, such as the great auk in the North Atlantic (Montevecchi and Kirk 1996), or Steller's sea cow in the North Pacific (Anderson 1995), mean that there is little choice but to eliminate these species from future restoration goals in the Back-to-the-Future process. Local extinctions (= 'extirpations'), on the other hand, are potentially reversible by natural recolonisation or by human re-introduction. But for comparison between the emergent properties of the series of whole-ecosystem models in BTF, it is important for all of the models to have the same number of compartments, although of course biomasses and fluxes can be vastly different. Similarly, ecosystem simulations that span two or more different models need all the compartments to be included in both models. Extinction of species makes this comparison difficult. What can be done in ecosystem modelling, therefore, when species have become locally or globally extinct? How may these factors be accommodated in the suites of ecosystem simulation models used in BTF? Before answering these questions, I review marine species that have become globally or locally extinct in our two CUS BTF ecosystems.

Figure 1. "Actually, there were three arks. The one with dinosaurs and other extinct forms sank due to overcrowding. The one with marsupials was blown off course and landed in Australia." A brave attempt to explain extinctions and biogeography. See www.christianforums.com/t40474&page=2.

GLOBAL EXTINCTIONS

The great auk, *Alca impennis,* was a large flightless, pelagic species of the *Alcidae* (auks) in the North Atlantic, and the original recipient of the name penguin (pen-gwyn, meaning 'white head', the winter plumage, in Welsh and Gaelic), a name later transferred to an entirely different order of Southern hemisphere birds (*Spheniscidae*). Hunting by humans, usually at island breeding sites, rendered the piscivorous great auk extinct by 1844 (Figure 2). Although the bird had been eaten for thousands of years by coastal peoples, in the late 18th and early 19th Centuries great auks were harvested for food, feathers and eggs on an astounding scale. For example, during the Napoleonic wars, Britain mitigated a blockade of Grand Banks' cod by importing shiploads of great auks from the
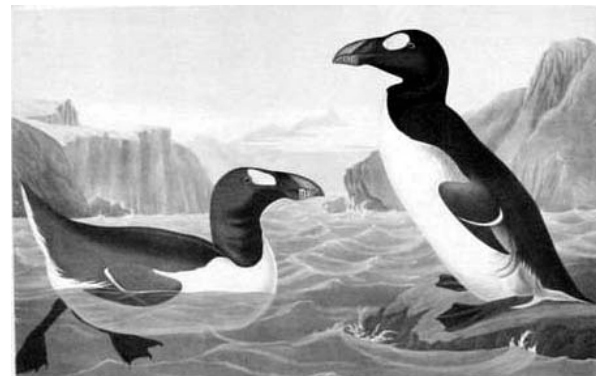

Figure 2. The flightless North Atlantic 'Penguin', Garefowl, Spearbill or Great Auk, *Alca impennis*, a 70cm, 5kg seabird once harvested by the shipload throughout the North Atllantic, and hunted to extinction by 1844. *John J. Audubon, chromolithographic print, The Birds of America, 24 x 36. San Joaquin County Public Library, USA.*
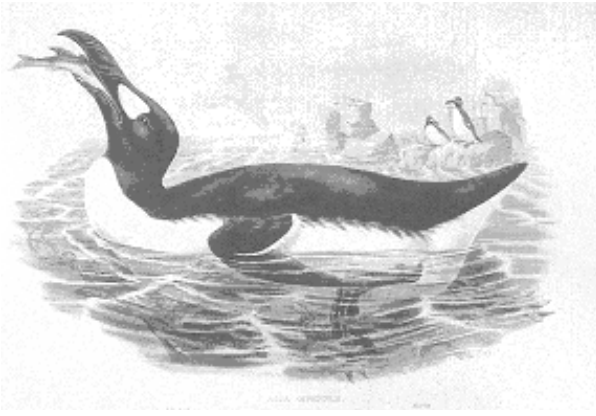
Figure 3. The great auk eating an adult capelin. Few North Atlantic seabirds eat such large prey today. *W. Imp, J. Gould and Whart 1840, coloured lithograph, 38.1 x 54.8 cm, J.H. Fleming Library, Ornithology Collections.*

islands off Iceland. Soon, there was a boisterous and increasingly lucrative international trade in diminishing numbers of great auk eggs, skins and skeletal remains in late Victorian times. Despite unconfirmed reports of sightings up to the early 20th century, the financial incentives brought about by this trade likely ensured that there are no surviving colonies. For example, the last pair of birds seen in Iceland was killed for sale, together with their egg (3rd June, 1844).

The classic study of the biology and demise of the great auk was published by Grieve (1885), and a recent book provides a thorough review (Gaskall 2000). Using data from archaeological remains in middens and from the sites of what appears to have been industrial-scale processing, Sarjeantson (1996) shows how the flightless great auk was wiped out, while the gannet, which was also exploited heavily but can fly, has avoided the same fate. Evidently, there was a population of at least a million birds in the North Atlantic before 1830 (Montevecchi and Kirk 1996), and middens suggest a far greater population over a wide range from Florida to the Bay of Biscay throughout the North Atlantic, and even the Mediterranean, in pre-historic times.

Although there is no quantitative data, fish species eaten by the great auk, which could dive to a depth of at least 10 metres, can be reasonably well deduced from some contemporary descriptions (see Grieve 1885). The diet likely consisted of pelagic fish such as capelin (Figure 3), herring and sandlance offshore, and large scuplins and juvenile cod when feeding inshore during the breeding season. For ecosystem modelling, metabolic parameters for this large bird might be taken as similar to the larger

southern hemisphere penguins.

Hence, there is certainly data enough to include great auks explicitly in mass-balance models of ancient North Atlantic ecosystems and to make preliminary biomass estimates based on diet and the other *Ecopath* parameters. But, in most cases, seabirds have such a small biomass and impact in marine ecosystem models that they are usually grouped into functional categories, such as invertebrate eaters, piscivores, inshore ducks and the like (Burke *et al.* 2002). In fact, in the CUS BTF North Atlantic models to date, the great auk has been grouped with other piscivorous seabirds (Heymans *et al.* 2002b, Davoren *et al.* 2002). This means that, provided the great auk's diet and metabolic parameters are represented in the appropriate functional group in the model, no special assumptions have to be made (Heymans *et al.* 2002a). This device also has the advantage that the group structure of the series of BTF ecosystem models remains the same over time. But the trick has the disadvantage that the possible impacts of the great auk's extinction on ecosystem structure cannot be explored. Since the great auk was clearly major predator of medium-sized fish, this would be an interesting topic to explore in the future.

In the 18th Century, the North Pacific was the location of two other dramatic global extinctions. In 1741 on the Komandorski islands at the extreme west of the Aleutian chain, Steller found

Figure 4. The extinct spectacled (= Pallas') cormorant, *Phalacrocorax perspicillatus* a 5kg flightless bird found by Steller in the Komandorski islands in 1741. Only 7 museum specimens of this North Pacific penguin-like bird survive and very little is known about it.
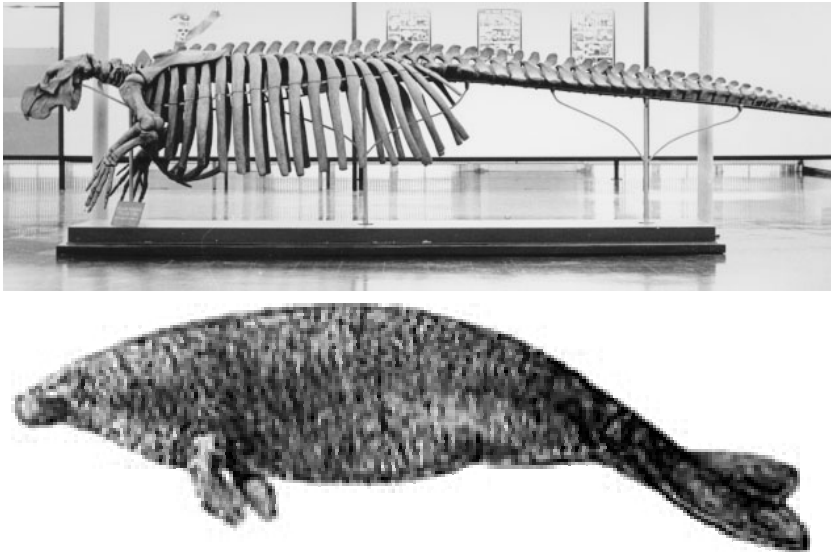
Figure 4. The Sea Cow was discovered by Steller in 1741, all were killed by 1769. *Upper panel:* one of the few extant skeletons in the Helsinki natural history museum. *Lower panel:* likely reconstruction of a Sea Cow (Hans Rothschaur, Germany). These sirenians were 7.5m long, weighed up to 11 tonnes, lived in herds close inshore, and appear to have eaten kelp and red algae.

a flightless spectacled cormorant*, Phalacrocorax perspicillatus* (Figure 4). He also discovered the Sea Cow, or rhytine, *Hydrodamalis gigas,* a large herbivorous sirenian (Figure 5).

Georg Wilhelm Steller, a stern, meticulous German, studied at the University of Wittenberg and then, after a spell as an army surgeon, worked in Russia at the Academy of Sciences in St Petersburg. Steller was 33 years old when he was employed as the naturalist on Janasson ('Vitus') Bering's 1741 expedition from the Tsarina Anna's Russia to the region between Asia and America. Anna had emerged as Empress in 1730 from the turmoil following Peter the Great's sudden death in 1725, and adopted the same expansionist agenda. Bering himself was a Dane serving in the Russian navy. The expedition was a tough call; among the hardships were scurvy, losing the other half of the expedition in a storm, shipwreck, over-wintering on what came known as Bering Island, and having to salvage wood to build a replacement ship[1]. Soon after the shipwreck, Vitus Bering died of scurvy that winter, along with half of his crew. But the tough naturalist Steller impressed the crew by searching out plants to treat scurvy[2]. By the next summer, the survivors began to hunt and eat the sea-cows and

they left the island with barrels of salted sea cow meat[3]. They also hunted and ate the large, flightless cormorant of which Steller wrote, "They weighed 12-14 pounds, so that one single bird was sufficient for 3 starving men."

Immediately after the expedition's return, Siberian fur traders flocked to the Komandorski islands, trapping foxes and sea otters for fur. They used the sea cows, said to be similar to almond-flavoured veal, and the flightless cormorants as a living larder. Sea cow blubber was used for cooking and as lamp oil, the milk of slaughtered cows was made into butter, and the tough hide was used for shoes, belts and skin-covered boats. The animal soon became rare, and although an order prohibiting hunting of the sea cows[4] was sent from St Petersburg to the Komandorski Islands on November 27th, 1755 (Domning, 1978), hunting seems to have continued. The last report of a sea cow being killed was in 1768.

The spectacled cormorant lasted longer, its last stronghold until 1850 being the small island of Aij Kamen (Stejneger 1889).

As well as an island, Bering got a sea named after him (on account of a filing error it seems, see Pitcher 1999). Steller ended up lending his name to an eider duck, a jay, a sea-lion, a rock-trout, an eagle and the sea cow. Also, unexpectedly, his name was used posthumously for Stellerite, a kind of silicate crystal found on the Komandorski Islands in 1909. Hounded by the Tsarist bureaucracy for humane treatment of some prisoners, a drunken Georg Steller died a miserable death in a snow storm at Tjumen, a Siberian town to the east of the Ural mountains, in November 1748, only four years after the expedition. Fortunately his notes (written in Latin under the harsh conditions of the island shipwreck) were preserved, and were retrieved, edited and published by P.S.Pallas (1781)[5],

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[1] Only one man, Sava Starodubtsov, a Siberian carpenter, thought that he remembered how to build a ship. The 46 surviving crew depended on his knowledge for their lives.

[2] Sven Waxell, one the ship's officers, said that Steller, although stern, was "a great botanist and anatomist, well versed in natural science". Steller saved the life of Waxell and his son. He named over 50 new species of animals and plants on the expedition.

[3] A preserved sea cow carcass, and many other specimens, had to be left behind.

[4] In 1754, an envoy of the Tsar wrote that sea cows were being exterminated at such a rate that they would soon be eradicated. Groups of two or three hunters from Kamchatka, the envoy wrote, were "inflicting huge waste and destruction".

[5] And translated into German (Pallas 1781).

Figure 5. 18th century engraving of a Steller's Sea Cow, *Hydrodamalis gigas,* being captured for food on Bering island by a ship's crew in the mid-1700s.

himself a German naturalist of repute working in St Petersburg, with a cat and several birds, including that flightless cormorant, named after him.

Some believe that small colonies of Steller's Seacows still live in remote areas of the northern oceans. In 1962, the crew of a Russian whaler reported seeing six animals that resembled sea cows, feeding in the Gulf of Anadyr, north of Kamchatka. In 1977, a Kamchatkan fisher reported seeing a drifting animal that matched the description of a sea cow (M. Raynal; http://perso.wanadoo.fr/cryptozoo/dossiers/rhytine.htm). Possible reports of sea cows before Steller might lend support to this idea. For example, in 1609 Henry Hudson reported animals that fit the description of sea cows near Novaya Zemlya. There are also reports from Greenland and other Arctic ocean sites But if these earlier reports from pan-arctic sites are correct, sea cow populations must have undergone a serious range collapse in the 17th Century before being described by Steller, or they would surely have been found by the many North Atlantic expeditions of the time. Sea cows were distinctive, large, impressive animals, forming obvious pair bonds, living inshore in small herds

with juveniles, and Steller (1751) even reports them coming to the aid of stricken animals. If their pan-arctic demise was due to recent human predation, there would surely be Traditional Knowledge and Myth concerning these massive social animals among today's native peoples of the arctic.

Archeological evidence places sea cows along the Pacific coasts of Asia and North America as far south as Japan and northern California. Their ease of capture and suitability for providing large amounts of human food would, like other North American megafauna, have rendered them susceptible to the 'clovis' hunting tools of first North Americans 12 to 15,000 years ago (Alroy 2001, Martin 1984). Most of the sites of slaughter and butchering would today lie submerged as a result of rising sea levels after the ice age (see Josenhans *et al.* 1997). It is interesting that the present coastal peoples of the Pacific North-West, whose DNA suggests that they arrived from Asia 6-8000 years ago (Morel 1997), have no knowledge or cultural memory of sea cows. It is likely then, that sea cows were wiped out by hunting very soon after boat-building humans inhabited the Asian shores of the North Pacific 35,000 to 25,000 BP (Erlandson 2001). The abundant food (shellfish, finfish, marine birds – including those flightless cormorants - and mammals) available from North Pacific kelp forests probably attracted early maritime people, and, it is thought, may have facilitated the earliest migrations of people from Eurasia to the Americas. It is possible that the whaling tradition of indigenous people of the North Pacific began with the over-harvest of the predator-naive and defenceless Steller's sea cow, focusing thereafter on cetaceans that were more difficult to harvest (Domning 1972). What Steller discovered on the uninhabited Komandorski islands then, was a living remnant population of one of the Pleistocene megafauna.

There is sufficient historical information about sea cow diet, and reasonable inferences about metabolism may be made from extant sirenians, for us to attempt to model them explicitly in a mass-balance ecosystem model (Stejneger 1886). The animals seem to have lived mainly inshore, near to sources of fresh water (Domning 1976). Steller's account indicates that the sea cow fed mainly on soft brown kelps and red algae, with a little sea grass. Anatomical adaptations to the sea cow's mouth and gut seem to fit with this. The huge sea cow gut seems to have been an adaptation to digest large amounts of poorly masticated algae. There were no teeth, only horny lips and upper palate for rasping algae from the
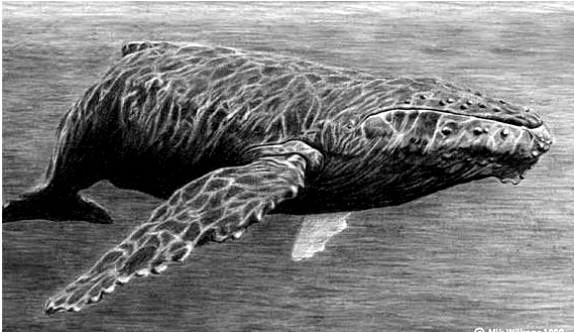
Figure 6. Two hundred Humpback whales, *Megaptera novaeangliae*, were common residents in the Strait of Georgia, BC, before commercial whaling wiped them out early in the 20th century. Nowadays, they may be slowly returning.

rocks. Steller says that large amounts of torn and dislodged kelp floated around sea cow feeding sites. Metabolic parameters for dugongs, 3 metres in length, could be scaled to reflect the slower turnover and larger body size of the sea cow (Pitcher 1998). Sea cow predators would have been mainly killer whales and perhaps cold water sharks. A starting value for sea cow biomass in a model might come from the estimated 5000 population in the area around the Komandorski islands. Assuming an area of 100km by 50 km around the islands, this amounts to an average biomass of about one animal per km^2 in inshore habitats, or about 7.5 tonnes per km^2.

As yet, no-one has attempted to construct an inshore ecosystem model that contains sea cows grazing kelp. In fact, it seems that kelp canopies are remarkably resilient to cropping of the distal fronds (Steneck *et al.* 2002). A multi-million dollar industry of canopy-cropping factory ships sustainably harvest kelp in California with little permanent damage to the kelp forests (Tegner and Dayton 2000). It is therefore unlikely that sea cow grazing of canopies deforested kelp beds. But the large quantities of kelp grazed would have dynamic effect on the kelp forest canopy structure, and would alter strategic cover and hence the survival of many inshore fish and invertebrates. And so, in contrast to most pelagic systems where floating phytoplankton comprises the food of higher trophic levels, these factors would make a sea cow/grazed kelp system structurally similar to many terrestrial ecosystems. Modelling the ecosystems of terrestrial game parks, or even dinosaur ecosystems, would make fascinating work in terrestrial or palaeo-ecology. Changes to the modelling framework to deal with habitat structural elements directly would be required, as discussed below.

LOCAL EXTINCTIONS: ABSENT BUT POTENTIALLY RESURGENT SPECIES

When species have become locally extinct ('extirpation' in conservationist language), one has to allow the possibility that they may return, either through natural migration or though active reintroduction.

An example of natural recolonisation is the humpback whale in the Strait of Georgia, British Columbia. More than 200 humpbacks were resident until wiped out by commercial whaling , a process that was complete by the 1920s (Gregr 2002, Winship 1998, Merliees 1985). Humpbacks now seem to be making a slow return to the Strait (Gregr 2002). Hopefully, simulation models may be able to capture this process of recolonisation. On the other hand, in Newfoundland, almost a quarter of a million walrus were estimated to be resident before exploitation started in 1800 (Mercer 1967), but have shown no signs of returning. Grey seals in Newfoundland have a similar status (see Heymans and Pitcher 2004, this volume). As with the globally extinct species discussed above, estimates of ancient biomass may be based on historical records of breeding sites, or, in the case of whaling, on records of whale kills.

Archeological remains of fish bones in middens show that Bluefin tuna, *Thunnus thynnus*, were at one time distributed along the entire coast of British Columbia and Washington state (Tunnicliffe *et al.* 2001, and see discussion page 139). Traditional Environmental Knowledge concerning weather and seasons for the hazardous spearing of these fast, giant fish

Figure 7. A sea otter, *Enhydra lutris*, eating a sea urchin. Sea otters were common residents along North Pacific coasts before being hunted for fur in the 18th and 19th centuries, and were wiped out in British Columbia. In recent years, sea otters have been re-established on Vancouver Island. Ecosystem modelling of sea otters is tricky because they are keystone species, altering the structure of inshore habitats.

suggest that they were seasonal visitors to coastal habitats depending on weather and conditions (see Lucas, 2004, this volume). However, they appear to be entirely absent from the region today.

Figure 8. Print of an Aleut sea otter hunt at Sanak Island, Alaska. Aleuts have been hunting sea otters for over 2500 years and devised a special whale-bone barbed dart that detaches from a shaft on contact with the otter. (See also Lucas 2004, this volume.)

To accommodate dynamic ecosystem modelling, groups that are present early on, but are later absent, have to be included in some way. As mentioned above, unless the species has been grouped with species of similar function, it is important to include in all time periods species that have become locally extinct over period of the series of ecosystem models. One technique that has been used for the Newfoundland series of CUS BTF models (1750, 1900, 1987, 1995: Vasconcellos *et al.* 2002), is to set biomass for the 'absent' periods to extremely low levels (zero cannot be used as it causes a software failure). For example, a value of $1*10^{-6}$ tonnes/km² has been used for walrus in models of recent Newfoundland ecosystems. At this low level they are essentially extinct (Heymans and Pitcher 2004, this volume). This technique, however, can create some technical problems as, during simulations, the species may undergo an unexpected modelling resurgence if there is enough food for them to do so. It may be possible to 'hold them down' using a biomass forcing function in *Ecosim* (see Martell 2004, this volume and discussion page 149).

An example of active re-introduction of an extirpated species is the sea otter, *Enhydra lutris,* reintroduced to from Alaska to Vancouver Island in British Columbia in the 1990s (see Lucas 2004, this volume). Sea otters became extinct through hunting in BC before 1900 (Kenyon, 1975), but following reintroduction, today have a established a small but increasing biomass in few areas. Sea otter diet and metabolic parameters are well-known (e.g., Bodkin *et al.* 1998, Reidman and Estes 1998) and it is not difficult to incorporate sea otters in ecosystem models (Ainsworth *et al.* 2oo2). The series of models for northern BC should ideally reflect the series of changes: abundant in the ancient past, absent after they were hunted to local extinction for their furs, and then re-introduced. But it is proving hard to include them explicitly in the models for every time period, and in models of restored BTF 'Lost Valley' ecosystems because, at very low biomass, they have 'plenty of food' and tend to undergo a modelling resurgence.

Problems in Modelling Keystone Species

An additional major problem for the BTF modelling here is that sea otters, however, are keystone species, causing large changes in habitat structure (Pitcher 1998, Simenstad *et al.* 1978). They alter the type of kelp available as cover to a suite of juvenile fishes and invertebrates by foraging on kelp-eating sea urchins that themselves graze selectively (Riedman and Estes 1990). The consequence is that inshore kelp ecosystems with and without sea otters have very different habitat structure and a different fauna of inshore fishes and invertebrates (Steneck *et al.* 2002).

When sea otters were extirpated in the Komandorski islands through hunting, this keystone mechanism may have helped to seal the fate of the sea cow: resurgent kelp-eating urchins would have competed for kelp as food (Anderson 1995).

The open canopy habitat known as 'kelp forest' appears to be dependent on the presence of sea otters (Steneck *et al.* 2002). Before human contact, predation by sea otters on urchins prevented overgrazing on kelp forests (Simenstad *et al.* 1978, Estes *et al.* 1998). In Alaska, Aleuts seem to have depleted sea otters as early as 2500 BP, causing the urchins to grow larger (Simenstad *et al.* 1978). From 1700, fur traders hunted sea otters to the brink of extinction, and kelp forests were then destroyed from over-grazing by urchins released from sea otter predation. Then after 1900 in Alaska, legally-protected sea otter populations increased, and the resultant trophic cascade re-established the kelp forest. Recently, however, kelp forests have disappeared again as sea otter populations have fallen prey to killer whales (Estes *et al.* 1998), that have shifted their diet to otters from pinnipeds after the latter populations declined significantly. The reason for the pinniped declines is still open to debate (Rosen and Trites 2000).

The sea otter's keystone effect is mediated

through habitat change that in turn alters feeding opportunities and refuge from predators for inshore fish and invertebrate species (Estes *et al.* 1989). Most of these changes are based on a living biomass acting as complex structured habitat, not on feeding interactions in a food web, and hence a purely trophic web model cannot simulate them. A routine to put 'non-trophic' mediation effects in *Ecosim* has been developed (Christensen and Walters 2003), but it is hard to fit the parameters for the interaction in anything other than a post-hoc fashion. In other words, keystone effects, like the sea otter, may be emulated in *Ecosim*, but not simulated.

The problem here is that spatial complexity and structure of habitats are not modelled explicitly in the EwE dynamic ecosystem system. For aquatic ecosystems this may be acceptable for the majority of cases, except where rooted macrophytes or coral reefs are involved, but it would be entirely unacceptable for most terrestrial ecosystems where plant architecture, both living and dead, provides a template of structured habitat for the vast majority of organisms. Alternative ecosystem modelling techniques, such as 'Atlantis' (Fulton *et al.* 2003), may be more appropriate in representing the effects of 'plant architecture'.

CONCLUSIONS

Extinctions cause problems for dynamic ecosystem modelling. This paper has put forward some suggestions about how these issues may be tackled, but some fresh advances in ecosystem modelling techniques are needed before we can approach species extinctions with confidence. BTF is one of the few fisheries policy analysis systems to explicitly and quantitatively deal with the extinction issue (Pitcher 2002).

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For discussion following oral presentation of this paper, see page 149.