CHALLENGING ECOSYSTEM SIMULATION MODELS WITH CLIMATE CHANGE: THE 'PERFECT STORM'

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ABSTRACT

When ecosystem models of the past are constructed, appropriate climate regimes need to be incorporated. Likewise, the effects of possible future climate changes on ecosystem structure and function must be included in forecasts of sustainable fisheries in reconstructed ecosystem. This paper examines how these issues might prejudice the BTF policy process. We show examples of models driven by inter-annual climate indices or by direct indicators of primary production.

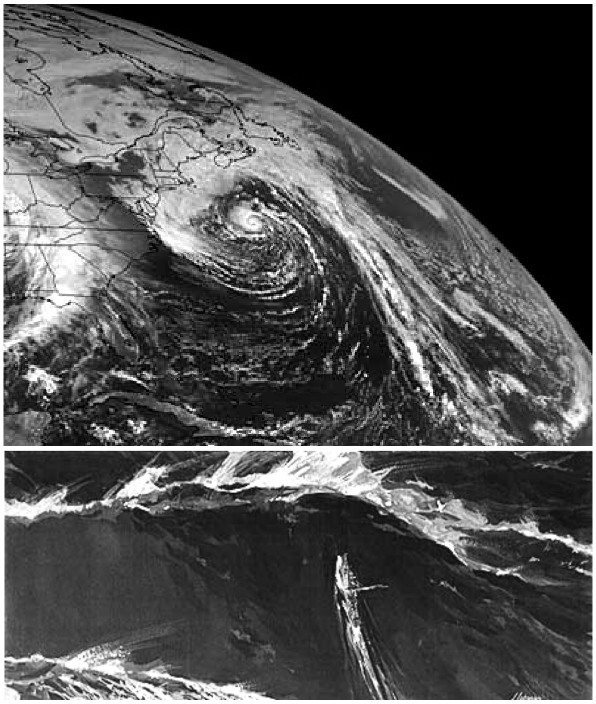

Figure 1. On November 1st 1991, the 'Perfect Storm' in the north-western Atlantic was accurately forecast by meteorologists (*top panel: composite radar picture, NOAA*) and many lives were saved, even though the swordfish vessel, the *Andrea Gail,* sank with all hands (*lower panel: pre-production watercolour from film*) because they ignored the warnings (Junger 1997). The science of weather forecasting is pretty good these days, although, in October 15th 1987 the British meteorological office was blamed for failing to predict the most damaging storm (18 people died) to hit southern Britain since 1703 (26th November, 8000 people died; Sutton 2003). Likewise, it may be both encouraging and hazardous to attempt to forecast the state of marine ecosystems under the influence of inter-annual climate fluctuations, climate-induced regime shifts and one-off catastrophic events.

Alterations in ecosystem structure due to climate change represent a major challenge to Back-to-the-Future (BTF) investigations. Climate changes that need to be addressed in BTF ecosystem simulations span time scales ranging from short inter-annual fluctuations to the major long-term shifts that result in ice ages. There are two aspects to the problem and each of them forms the basis of one of the most common criticisms of the BTF approach. First, the reconstruction of past ecosystems to use as future policy goals may be prejudiced if those past ecosystems existed under different climate regimes. Secondly, BTF relies on forecasts made by sustainably fishing restored past ecosystem states in which simulations are projected into the future – the 'fished Lost Valley' scenarios – and so these forecasts may not be viable unless likely climate change is taken into account. The two aspects of the problem differ fundamentally in scientific terms. Past climate changes are inherently knowable, can be estimated from reported observations, and these estimates, if poor, can be improved. Future climate changes, in common with all scientific forecasts such as weather forecasting, are unverifiable until the specified future time is reached, and so the best we can do is to project a series of likely scenarios, some of which may be more likely than others. In its most serious form the 'climate change' criticism goes something like this. Even if the climate of past times is well understood and the ecosystem models of the past adjusted accurately to take account of those changes, past ecosystem states cannot be used for future policy goals. We can expect climate to induce differences among past ecosystems, the present day ecosystem from which we have to commence the reconstruction process, and the projected future. This paper aims to analyse these issues and assess the degree to which the BTF process might, in practice, be prejudiced by them.

Types of climate change

Oceanographic influences on the living organisms in marine ecosystems are mediated ultimately through temperature and ocean circulation currents. Proximal factors driven by these changes affect thermocline depth and the upwelling of nutrients from sediments that determine phytoplankton production. Freshwater runoff, driven by rainfall, and ice melt, driven by temperature, can also have a profound influence

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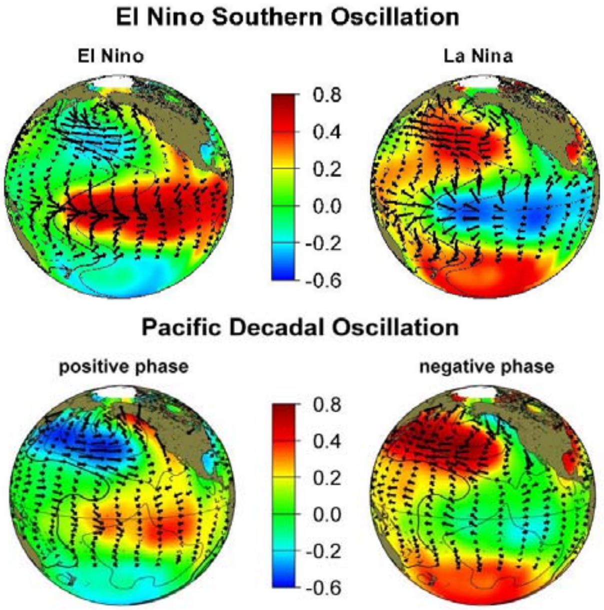

Figure 2. Temperature contour and water flow diagrams showing ENSO and PDO from a North Pacific perspective. Colour-scaled values are degree Celsius deviation from long-term mean (from NOAA).

on inshore marine ecosystems. Changes in ocean currents are important in the physical dispersal of planktonic larval stages of fish and invertebrates. Temperature changes can affect fish physiology directly, but can also determine global wind patterns, that in turn affect ocean currents. Hence, a complex of climatic factors affects the templates of habitat offered by the marine environment to the suites of organisms that compose its ecosystems (Review: Barange 2002).

Time scales of climate change

These climate influences occur over a range of time and spatial scales. Seasonal climate changes are those with which we are most familiar and, especially in polar regions, can have a dramatic effect on the structure and functioning of marine ecosystems. In this paper the ecosystem modelling in which we are interested is based on annual changes in biomass, and so seasonal changes are not considered further here, although they can be incorporated into *Ecosim* modelling (S. Martell, unpublished). Inter-annual changes include more-or-less random fluctuations in temperature and ocean currents from year-to-year, whose variance is characteristic of a particular geographical location. It is this variance that is most likely to increase under the influence of a global warming trend.

Inter-annual changes also include major ocean forcing such as El Niño (male child), named because the main effects occur at ‘navidad’ (Christmas). Its primary effect is to shift the equatorial current in the tropical Pacific to a greater or lesser degree, with a time span for its effects of 6-18 months. Spring warming of the sea to the north of Indonesia causes the Eastward warm equatorial current to increase. This current then swings poleward off South America to displace and overlay the cold northerly Humboldt current, with origins in the Antarctic ice melt, whose upwelling normally drives exceptional marine production off Peru. Exactly what triggers El Niño to start is not yet known, although the Earth’s spin is reduced by the mass of less dense warm water. The opposite effect, La Niña, gave rise to the concept of ENSO (El Niño Southern Oscillation: Figure 2). Although based in the central and southern Pacific, ENSO’s influence extends to the North Pacific, Indian and Atlantic oceans. Records up to the 1970s indicated major ENSO events occurring about once every 15 years, but in the past two decades they have become up to three times as frequent.

Medium-term, quasi-cyclic changes occur over larger ocean regions on decadal time scales, for example the Pacific Decadal Oscillation (PDO: Figures 2, 3) with a period of 20-30 years. These are major shifts in currents and temperatures,

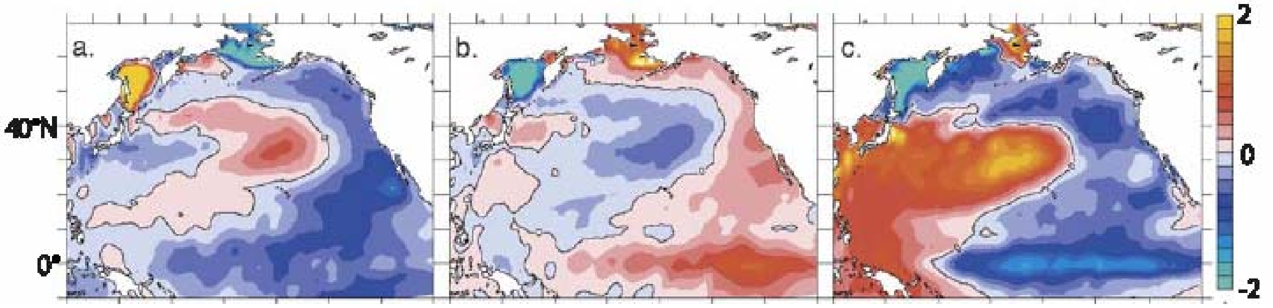

Figure 3. Recent changes in the Pacific Decadal Oscillation (PDO). Clear evidence of decadal regime shift revealed by sea surface temperature anomalies in the North Pacific. (NOTE: coloured figure may not show up well in grey scale). Panels show left-to-right, (a) 1970-1976, cool phase of PDO; (b) 1977-1983, warm phase of PDO; (c) 1999-2003, strong cool phase of PDO. These temperature changes are paralleled by sea level pressure and wind patterns. (Diagram from Peterson and Schwing 2003.)

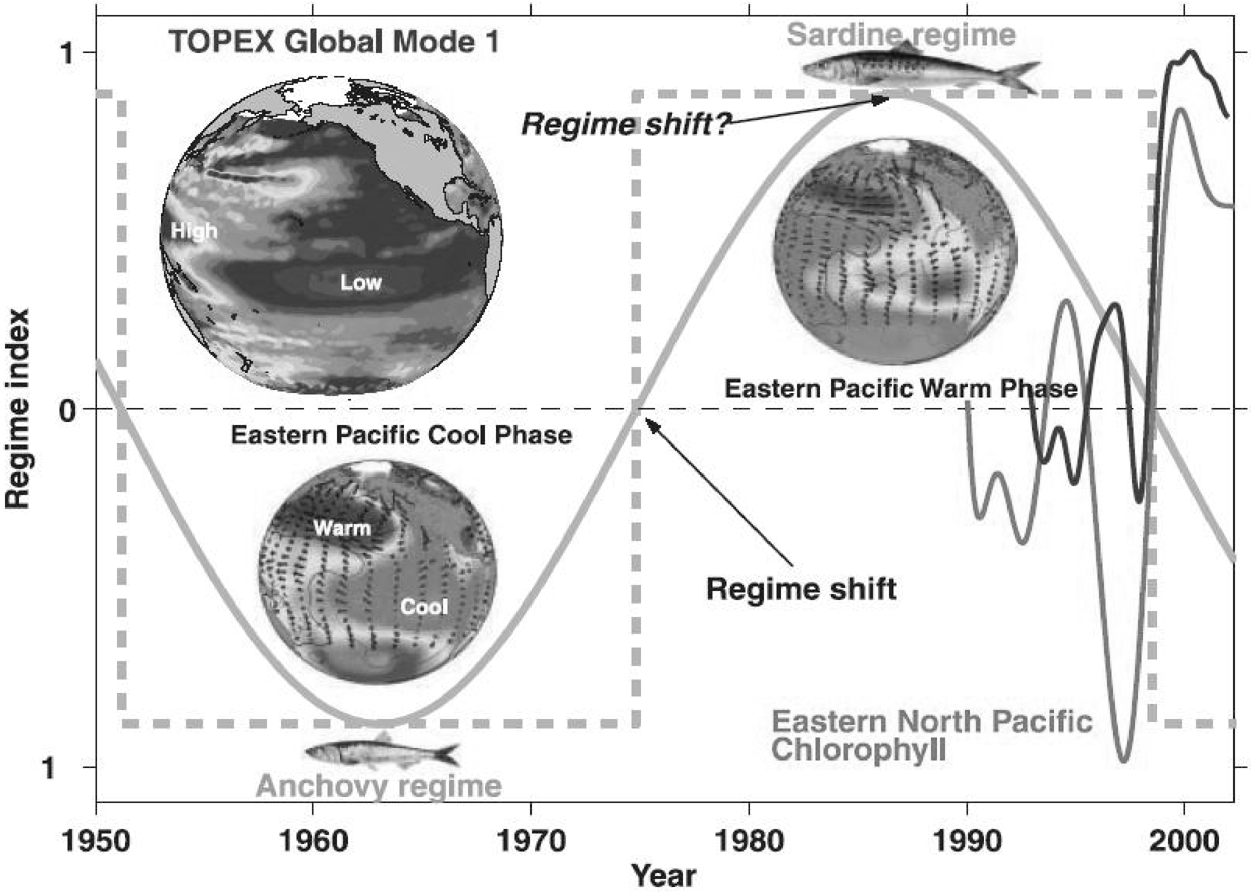

Figure 4. El Viejo/La Vieja marine climate regime analysis for the Pacific. In cooler conditions, anchovies dominate (La Vieja), while in warmer regime (El Viejo) sardines are abundant. Spatial SST and atmospheric circulation anomalies are shown for each regime (globes). Note that the eastern Pacific is out of phase with the central North and South Pacific. Some indices suggest rapid shifts (dashed line), whereas others are gradual (solid line). Low sea surface height (TOPEX) and high chlorophyll (California Current) in the cool anchovy regime mean a shallow thermocline/nutricline. Associated basin-scale current systems support recent stronger California Current and a weaker Kuroshio Current. (Diagram from Chavez *et al.* 2002).

sometimes occurring rapidly between relatively stable periods. A number of longer-term cycles have been suggested (e.g. Klyashtorin 2001), the most compelling of which is an approximately 62-year cycle in the Pacific, termed El Viejo/La Vieja [old man/old woman] (Chavez *et al.* 2003) (Figure 4). Very long-term climate trends can lead to ice ages and consequent sea-level changes. In addition, there is good evidence for a dramatic human-made recent global warming trend (Figure 5).

Four questions are critical to the use of climate influences as a part of the BTF process. (1) Can we drive and/or tune past models using time series of climate or surrogate climate data? (2) Are these models stable? (3) Are the observed biomass dynamics realistic and do they emulate observed regime shifts? (4) Can we determine and 'lock on' to the appropriate state of ecosystem for the model of a past time period?

HOW DO CLIMATE CHANGES AFFECT AQUATIC ECOSYSTEMS?

The vast majority of papers in the 'effects of climate on fisheries' literature describe climate impacts on a single species at single geographical location, and only a few deal with populations of the same species over a wider geographical area. As might be expected, a fair number of well-argued publications supported by solid data cover the impact of climate changes on fisheries recruitment. There are also a small number of synoptic, global-scale analyses of climate-induced changes to groups of fisheries of interest such as the small pelagics. There are very few attempts (e.g., Barange 2002) to deal with the integrated effects of climate on whole ecosystems, and even fewer attempt to compare ecosystem-scale effects over wide areas. This paper therefore includes a review of recent publications that shed light upon the ways in which climate changes alter fish

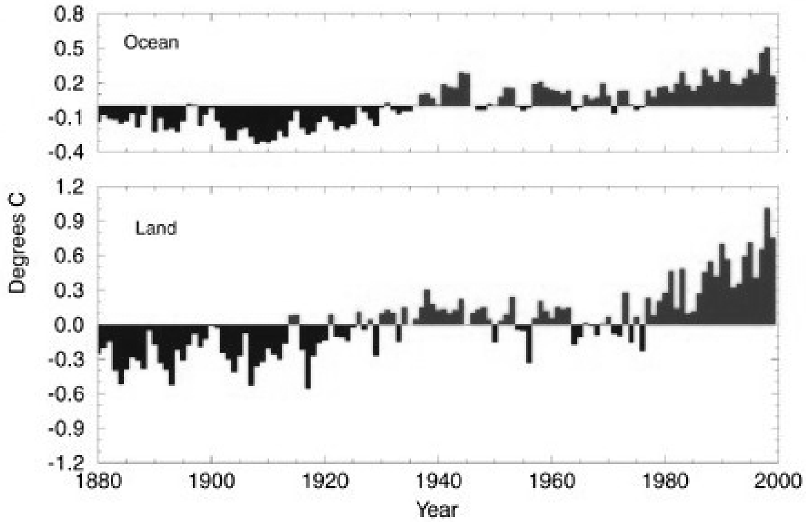

Figure 5. Global warming trend shown in average annual global temperate anomalies, 1880-2000, from land and sea. Source: National Climatic Data Center, NOAA, USA.

populations and fisheries within an ecosystem context.

It seems that shifts in ocean climate regimes can alter ecosystem structure quite quickly, and these may be faster at lower trophic levels (Barange 2002, Hare *et al.* 1999). Changes in wind patterns affect oceanic circulation, salinity, thermocline depth and primary production; changes in the distribution and abundance of predators and prey influence fish, marine mammal and bird populations (Barange 2002). Sometimes changes can affect similar species within a single domain in opposite ways. Surprisingly, some hold that climatic regime shifts can have opposite effects on the same species in different ocean domains (Benson and Trites 2002).

Fish growth is often affected directly if water temperature alters (e.g., halibut; Clark *et al.* 1999), but there is usually little attempt to partition this effect into a direct metabolic influence and indirect effects mediated though the food web. Extreme temperatures can directly affect the physiology of migrating salmon (Hinch *et al.* 2002).

It is well documented that climate shifts can have a serious impact on fisheries (e.g., Japan; Kawasaki and Omori 1995), especially when they coincide with overfishing, as in the classic collapses of the Monterey sardine in the 1950s and the Peruvian anchoveta in 1971 (see accounts in Pitcher and Hart 1981).

And, more recently, recruitment of cod in the heavily overfished North Sea appears to be threatened by climate warming trends (O'Brien *et al.* 2000).

Time series of catches and other data often suggest synchronous changes over large ocean basins, suggestive of climatic and oceanographic factors at work in determining abundance. For example, using catch time series several centuries long from the Mediterranean and adjacent Atlantic areas, Ravier *et al.* (2001) demonstrated 7-fold fluctuations in abundance, and synchronised 100-year and 20-year cycles, in the traditional tuna 'tonnara' trap fisheries that formerly caught bluefin tuna on their annual spawning migrations Moreover, coherent patterns observed across large regions of the Pacific demonstrate the strong role of climatic forcing in determining the size fish populations (Hollowed *et al.* 2001). Catch records suggest that warmer years and regimes may lead to higher fisheries production (e.g., sablefish; King *et al.* 2001) in higher latitudes (e.g., Beamish 1993).

Fishery catches, however, may be influenced by a number of factors and so other means have been explored to examine climate-linked changes. For example, nitrogen isotopes in lake sediments demonstrated large changes over 300 years of Alaskan sockeye salmon abundance related to climate (Finney *et al.* 2000). Similar analyses

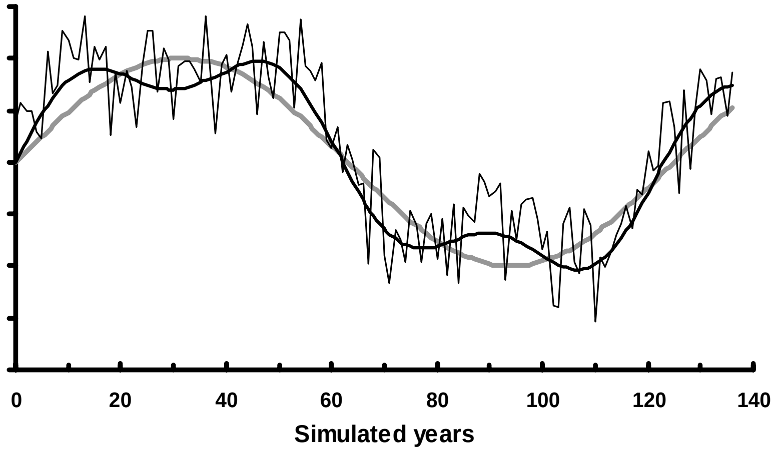

Figure 6. Simulated climate influence time series constructed from 62 and 20-year cycles (sine waves), and ENSO anomaly (triangular probability distribution). Note that coincidence of two cycles can lead to 'rapid shift of regime', and to 'stable plateau' periods. Authors' simulations show that these effects depend on relative wavelengths and starting point.

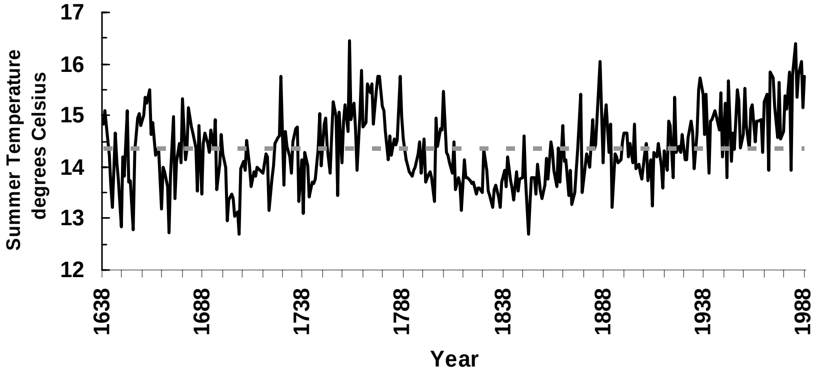

Figure 7. Summer temperatures in northern British Columbia as reconstructed from pine tree rings. Dotted horizontal line shows mean. Note recent warming trend. Data from Szeicz and MacDonald (1995).

spanning 2,200-years reveal very large shifts in abundance resulting from climatic forcing, far exceeding the decadal-scale variability recorded from catches during the past 300 years. For example, salmon declined from 2100—1200 BP, but were more abundant from 800—100 BP (Finney *et al.* 2002). On equally long time scales, the abundances of 1200 years of Pacific sardine and Northern anchovy off the California coast (Baumgartner *et al.* 1992) alternate with the salmon, giving some clues as to the ocean mechanisms at work (Finney *et al.* 2002). The regime of high clupeid abundance (2000–800 BP) is confirmed by archeological studies (Tunnicliffe *et al.* 2001).

Alheit and Hagen (1997) describe an example of long-term climate forcing of European herring and sardine populations. In the Skagerrak, since the 10th century, there were nine boom periods for inshore herring fisheries, each lasting several decades. Otherwise, the herring fishery was very small. Some other European herring fisheries coincide (English Channel and the Bay of Biscay), whereas others (Norwegian herring and sardines) alternate with these periods, apparently driven by negative anomalies in the North Atlantic Oscillation index (more sea-ice in the Arctic, cold European air and water temperatures, fewer westerly winds).

In Norwegian waters, the North Atlantic Oscillation index relates to recruitment of North East Arctic cod (Godø 2003), while sea surface temperature is linked to Barents Sea capelin and Norwegian spring spawning herring stocks, although heat flux, ice cover and heat transport are also important variables (Stiansen *et al.* 2002).

Climate influences on recruitment are often a very important mechanism. In the North Sea, 22 years of data on climate during larval stages explained more than 70% of recruitment variability leading to models that could forecast recruitment in the summer of the spawning year (Svendsen *et al.* 1995). In coho salmon climate factors determine cohort strength; faster growing fish better survive the first winter at sea when upwelling nutrients have led to better plankton feeding conditions in the previous summer and autumn (Beamish and Mahnken 2001). Rodhouse (2001) shows how squid recruitment is correlated with synoptic oceanographic data. For example, in the eastern Pacific coastal upwelling system catches in a squid fishery for *Dosidicus gigas* are linked to the El Niño (ENSO) cycle. Twenty-fold fluctuations in mackerel recruitment in the Gulf of St. Lawrence were related to copepod abundance, which was negatively related to climate expressed as freshwater discharge (Runge *et al.* 1999). Many prawns recruit like this too.

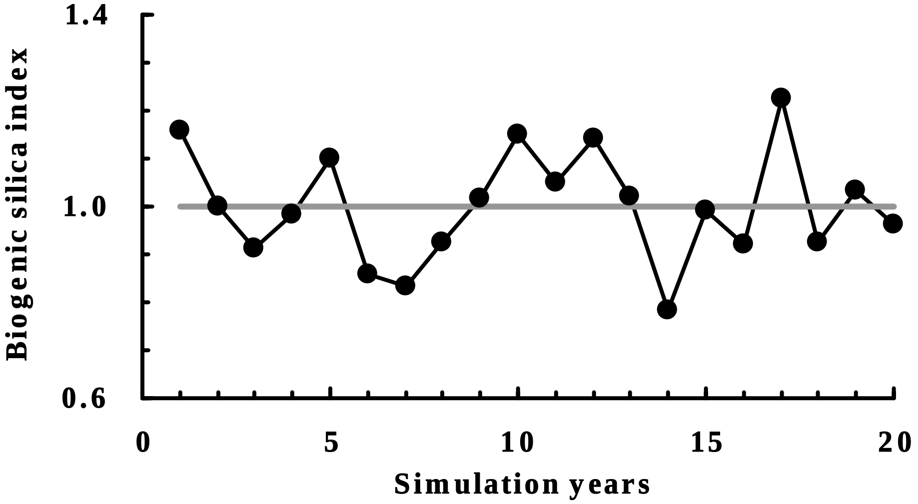

Figure 8. Twenty-year time series of data used to drive primary production in dynamic ecosystem model forecasts of Lake Malawi. Data is based on published time series of biogenenic silica in lake sediments, in randomized order, normalised to unit mean, and the variance adjusted iteratively to fit likely extreme lake biomass values. Data from Johnson *et al.* 2001.

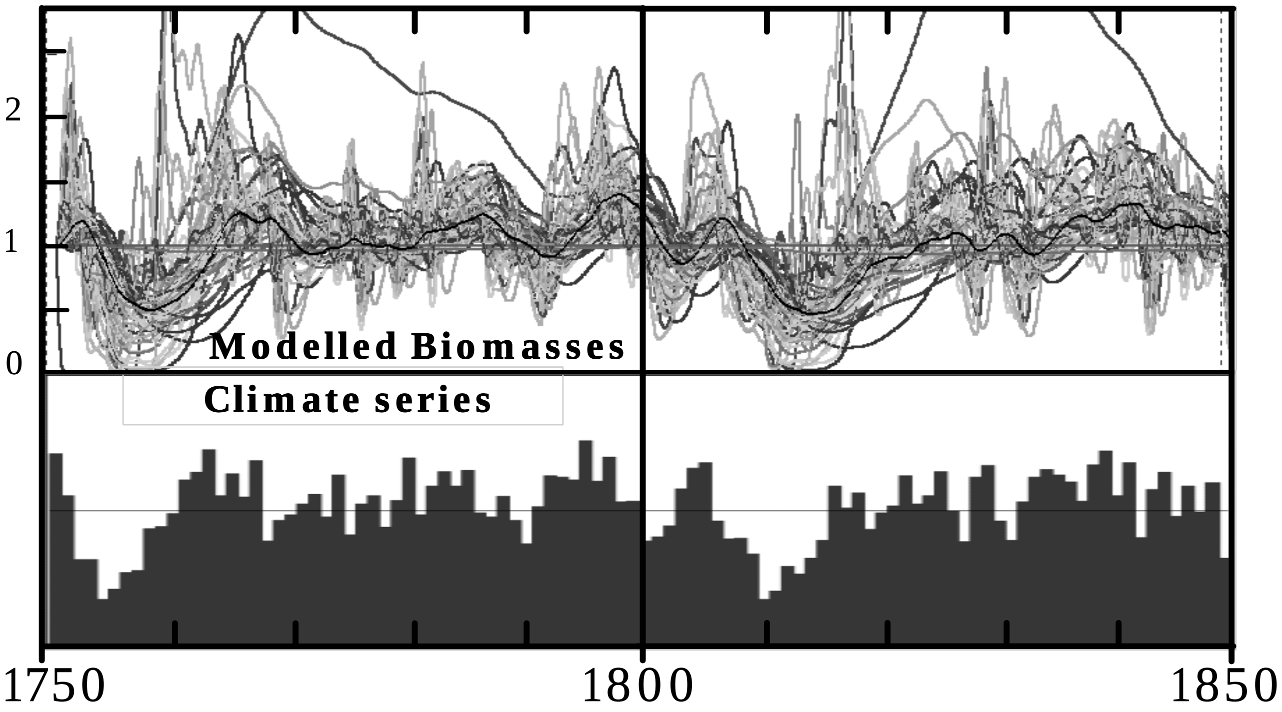

Figure 9. Example of 46-compartment whole ecosystem model driven by 100 years of a marine climate index based on tree ring data. Annual values of climate series shown at bottom panel and were used to drive primary production. Modelled biomass changes relative to starting values shown above and below the unity line in upper panel. Starting model is a reconstruction of Northern British Columbia as it may have been in 1750. Marine climate data from Gedalof and Smith (2001).

The recruitment – climate relationship may be quite complex. For example, the spring phytoplankton bloom can vary by up to 6 weeks in Newfoundland, driven by the amount of colder, fresher water from glacial runoff. First-feeding cod larvae have a precise dietary requirement: the nauplii of a copepod *Calanus finmarchicus* in the spring bloom. 'Match/mismatch' feeding conditions drive cod recruitment success, and global warming may prejudice recovery of depleted cod stock by creating long runs of ‘mismatch’ years (Conover *et al.* 1995). In a similar way, climate-driven fluctuations of sardine, hake and mackerel populations in the northern part of the California current appear to be linked to specific diatoms required by sardine larvae (McFarlane and Beamish 2001).

In the Pacific North-west, the PDO cycle has a strong influence on sockeye, pink, chinook, and chum salmon, herring and halibut, especially juveniles (Clark *et al.* 1999, Beamish and Bouillon 1993, Mantua *et al.* 1997). The El Viejo/La Vieja cycle describes an alternation between warm eastern boundary currents favouring sardine and colder conditions favouring anchovy regimes. Moreover, in this system the transitions between different regimes are relatively abrupt, but may be out of phase in different parts of the Pacific (Figure 4).

Other less obvious ecological effects are sometimes found. For example, a long-term trend for warmer water has stabilized the water column in Lake Tanganyika, reducing mixing, so that primary production is reduced by 20% and fish production by 30% (O’Reilly *et al.* 2003).

Fish species with life spans comparable to or exceeding the duration of adverse conditions may weather out the adverse period of a cycle, but at low population sizes, cascade effects can impede population growth in the good period. Fish with life spans shorter than the duration of adverse conditions can only be managed by linking catches to the environmental conditions, preferable using a delayed response (MacCall 2002).

DRIVING ECOSYSTEM MODELS

Clearly, the effect of climate change has to be accommodated in forecasts using ecosystem simulation models as much as possible. To do this, primary production, and other parameters of

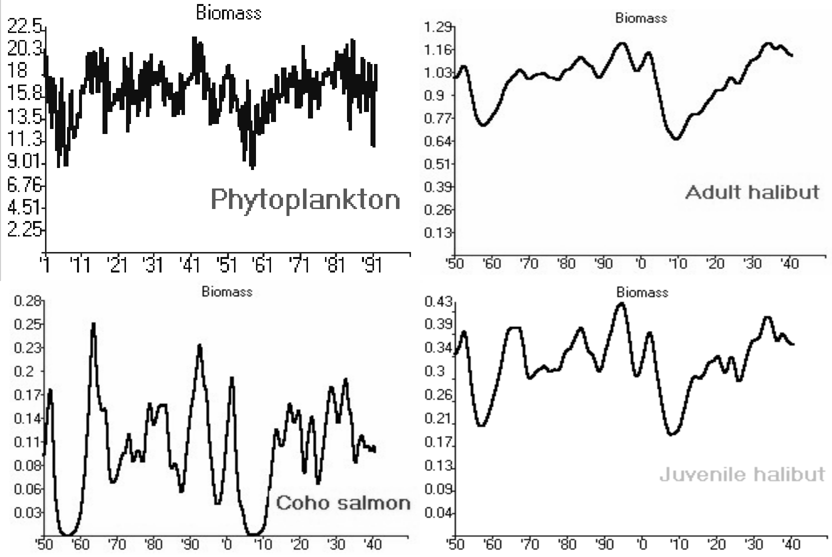

Figure 10. Biomasses of some of the individual groups in the ecosystem model driven by climate time series shown in Figure 9. Top left panel: phytoplankton; Top right panel: adult halibut; lower left panel: coho salmon; lower left panel: juvenile halibut.

ecosystem models, such as stock-recruitment relationships, may be driven in a variety of ways.

Driving models with forcing functions

Although precise forecasts of inter-annual climate changes may never be possible, randomized selections of such data, or functions that emulate past climate changes, can be used to drive forecasts on the basis of likely scenarios. Forcing functions may be based upon empirical inter-annual variation, decadal or longer-period oscillations, or climate proxies such as a local upwelling index. Longer term climate cycles may be included in the forcing function, like the 62-year 'La Vieja/El Viejo' alternation between warm/cold eastern boundary current sardine/anchovy regimes (Chavez *et al.* 2003). All these factors can fairly easily translated into a driving variable for ecosystem modelling (e.g., Figure 6). The algorithm could be modified to take account of large ENSO events that may trigger PDO shifts (Peterson and Schwing 2003).

Residuals between historically measured biomasses and simulated biomasses can be minimized in *Ecosim* models by comparing fits of a range of climate forcing functions. Climate forcing of modelled phytoplankton production may be sufficient, but in some cases climate forcing of recruitment parameters may also be useful for some fish species.

Driving models with climate data

Rather than use mathematical surrogates, tree rings can supply long historic records of inter-annual temperature changes in a region (e.g., Figure 7, northern British Columbia; Szeicz and MacDonald 1995). In the deep sea, growth rings of bamboo coral have been used in a similar fashion (Koslow and Thresher 1996). Sea surface temperature data has been used to drive a biomass model for Japanese sardine (Noto and Yashuda 2003). Sea temperature anomalies successfully improved biomass fits in an *Ecosim* model of the English Channel (Stanford 2004). In small pelagic fish in upwellings world-wide, production rate rather than biomass seems to be the best correlate for climate regimes (Jacobson *et al.* 2001) and hence the best variable to force in single species models. Multi-species modelling driven by temperature time series suggests that species respond differently to climate depending on their position in the food web (Jurado-Molina and Livingston 2002).

Lehodey (2001) has modelled the spatial effect of warmer and cold waters. In the tropical Pacific, ENSO affects a cold tongue of upwelling water that favours high production adjacent to warm unproductive pools. A spatial production model of skipjack tuna uses spawning area, larval and juvenile transport, adult tuna temperature preferences and forage fish prey driven by primary production. Observed movements of skipjack confirm the model results, which show ENSO driving an out-of-phase pattern between the western Pacific region and the cold tongue.

In some cases, biogenic silica deposits in sediments, which track the abundance of diatoms, may accurately reveal the past annual changes in primary production (e.g., Lake Malawi; Johnson *et al.* 2001), and such data has been used to drive forecasts in ecosystem simulation models (Figure 8).

Figure 9 illustrates an ecosystem model, representing a past time (1750) in Northern British Columbia, with phytoplankton production driven by a 10-year time series of marine climate data (transformed tree ring data; Gedalof and Smith 2001). Figure 10 shows separate plots of some of the groups in the model: large climate-driven changes in coho and juvenile halibut

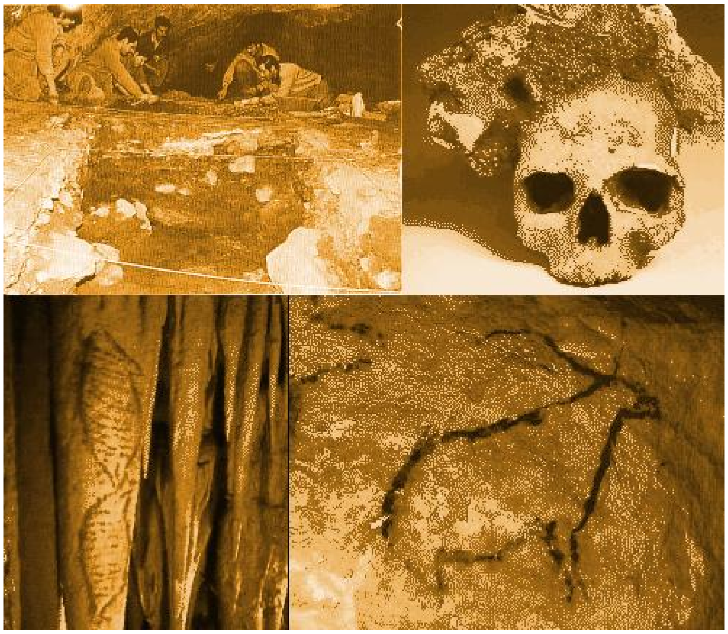

Figure 11. Top left: the first excavations taking place at Nerja, Adalusia, in the 1920s. Top right: human skull from the cave dated at 18,0000BP; Lower left: cave art of food fish, possibly *Pagrus*; Lower right: more cave art food. (From www.cuevadenerja.es).

emulate those expected from the literature (Mantua *et al.* 1997, Clark *et al.* 1999).

Information about specific past times from climate time series might be used to adjust each of a series of BTF ecosystem models to the appropriate contemporary regime conditions. This has not yet been attempted, however, because there is a logical complication. Starting values for the ecosystems of 'sustainably fished Lost Valley' analyses would have to be re-adjusted to a regime appropriate for starting the rebuilding process. Past climate data could at least enable past models to avoid major fluctuations compared to the present. In addition, the problem might be minimised by approximating an average ecosystem state over a period of 10-20 years.

THE 'CAST OF PLAYERS' TECHNIQUE: A SUGGESTION

To emulate changes in species composition in an ecosystem model as climate changes, the modeling system could perhaps be modified to use a 'cast of players', members of which might be brought on-stage and off-stage when conditions are appropriate (Pitcher 2004). In the 'on-stage' condition a species would play its full part in the food web of functioning model, acting as a predator, prey and competitor, and as an actor in any mediation processes, according to its model parameters and diet matrix. When 'off-stage' the species would play no part in the model dynamics. On-stage and off-stage conditions would be set, for example, by the value of an external time series, such as water temperature or a climate index. By bringing on stage members of a food web at different times and temperatures, a large number of intermediate ecosystems might be modelled. The technique could also be used to emulate species re-introductions, or recolonisations, following local extinctions (see Pitcher 2004, this volume).

Some archaeological data sets may provide useful test-beds for the 'cast of players' technique. For example, in the Cueva de Nerja, Andalusia, Spain (Figure 11), human middens reveal the fish that early Mediterranean people were eating over a 9000-year sequence (Morales *et al.* 1994, Rosello-Izquierdo and Morales-Muniz 2001). Early in the sequence, from about 14000 BP, the human diet consisted of a sparid fauna similar to the present, but, during a pluvial period at the end of the last Ice Age between 11000 BP and 9000 BP, humans were eating large cod and haddock, a fauna typical of Norway today. The midden fish bones the show that, by 8000 BP, a typical Mediterranean fauna had returned. The shift from Mediterranean to Nordic and then back to Mediterranean ecosystems might be emulated using the 'cast of players' driven by ancient temperature or climate proxies. Stratigraphic archaeological data could be used to 'tune' the process.

If successful in a trial such as the above, the 'cast of players' technique could also be used to forecast the consequences of global warming in a marine ecosystem by including a set of species from adjacent warmer ocean areas as well as those present today, for example, and then driving the actors on-stage and off-stage with a trend in temperature or climate factors.

CONCLUSIONS

We are now in a position to provide some preliminary answers to the four questions raised in the introduction to this paper. (1) It is certainly possible to make credible attempt to drive and/or tune models of the past using time series of climate or surrogate climate data. (2) Only a few climate-driven whole-ecosystem models have been constructed, all of which have appear to be stable, but far more will have to be done before we are sure of their stability. (3) Whole-ecosystem models have yet to emulate observed regime shifts, and the validity of their biomass dynamics needs more investigation. (4) We do not yet know if we can determine and 'lock on' to the appropriate state of ecosystem for the model of a past time period. Clearly, it is early days for climate-driven whole-ecosystem modelling.

Climate cycles mean that fisheries management aimed at rebuilding stocks may have to use a much longer planning horizon has been typical. MacCall (2002) suggests that, during adverse periods little rebuilding may occur even if fishing is halted, while in favourable periods, depleted populations of large predators allow smaller unfished competitors to thrive, again inhibiting population growth. Consequently, rebuilding of apex predators may require a century or more. Again, for policy work based on whole-ecosystem modelling, these issues need systematic investigation using the new climate-driven simulations.

Climate-change affects species mix and shifts centres of production. Everett (1997) warns that,

> "The positive effects of climate change, such as longer growing seasons, lower natural winter mortality, and faster growth rates in higher latitudes, may be offset by negative factors such as changes in established reproductive patterns, migration routes, and ecosystem relationships. Serious consequences could occur where these factors interact with pervasive over-fishing, diminishing nursery areas, and extensive coastal pollution."

Barange (2002) advises that multi-disciplinary research is required to understand the challenges of climate change. Moreover, as in other areas of fishery management, suitable actions consequent upon accurate forecasts of the effects of climate fluctuations may be hard to implement as policy. All sorts of human constraints may apply, such as lack of understanding, failure to appreciate uncertainties, and unanticipated reactions depending on unequitable benefits (El Niño, Broad *et al.* 2002).

The 'Perfect Storm' in November 1993 was formed when two independent meteorological phenomena occurred together. The storm was accurately forecast and its likely impacts well understood, but when it arrived, those caught up in its fury were both astonished and ill-prepared. It is to be hoped that forecasts of the effects of directional climate change - global warming - on natural ecosystems in the sea will not catch us so unprepared. While recent effort has been rightly focused on the disastrous effects of an era of outrageous and uncontrolled overfishing, it is sobering to realise that climate may bring about equally devastating changes. Finney *et al.* (2002) warns that

> "an unprecedented shift to a very low productivity regime, lasting centuries, can occur even without the influence of fisheries and other anthropogenic impacts".

Today, with both overfishing and climate shift independently caused by human actions, we may have unwittingly set the stage for a 'Perfect Storm' of changes in the ocean ecosystems.

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For discussion following oral presentation of this paper, see page 142.

TUNING ECOSYSTEM MODELS TO PAST DATA

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ABSTRACT

This note sets out how whole ecosystem simulation models may be tuned using past surveys or fisheries assessment outputs.

The question is often asked as to whether these *Ecopath-with-Ecosim* models could actually influence decision-making? Their usefulness for policy is strongly connected to the accuracy of their outputs. Are the predictions that they make reliable and robust? The process of tuning is intended to enable the models to reflect reality.

Ecosystems have enormous complexity and the grim reality for modelers is that capturing this is an impossible task (Oreskes *et al.* 1994). To predict with absolute certainty what the future holds is beyond the capabilities of computer simulations. Conversely, if the models cannot simulate what has already known to have occurred, they are not reliable enough to be used as a predictive tool. Hence, the essence of 'tuning' an *Ecopath* with *Ecosim* model is to run the model through time and compare it's estimates to observed time-series data.

Tuning is an iterative process through which group by group the model moves towards the a better representation of the actual ecosystem. The first stage is to have a balanced base *Ecopath* model that you are confident reflects the time period you have modeled. Simple diagnostic checks on the model, such as setting fishing mortality to zero, or increasing it ten-fold, give an early indication of the validity of the results. Certain groups with a high Ecotrophic Efficiency (EE), whose abundance is controlled primarily by fishing, will rapidly increase if fishing pressure is suddenly reduced. The modeler, being aware of the system, will be able to ascertain whether this increase is reasonable and modify the basic input parameters if necessary.

The aim of my English Channel ecosystem model was to predict into the future using a range of policies. Sufficient data were available to build an accurate contemporary model of the English Channel (Stanford 2002, 2004). In order to have confidence in these predictions it was necessary to build a past model, which would act as an anchor point from which to extrapolate to the present day. A number of the commercially exploited stocks had been assessed since 1973 and this was the year designated for the earlier model. This model was constructed and run from 1973 to 1995 using stock assessment data for fishing mortality. Where this was not available, estimates were provided from experts or similar stocks so that for each exploited functional group there were fishing mortality data.

Where the biomass estimates of the model significantly differed from stock assessment data the English Channel model required modification through one or more of three ways:

1. *The basic input parameters entered into Ecopath could be changed.* Fishing mortality may cause the EE to be close to 1 and an increase in fishing will cause the group to decline. If the 1973-1995 time-series data indicated that the group was more resilient than predicted by the model, increasing the starting biomass or the estimate for production/biomass will dull the impact of fishing.

2. *Vulnerability settings can be altered.* Vulnerabilities are a measure of whether the system is predator or prey controlled (top down or bottom up). Increasing the vulnerability towards 1 means that predators control the system and that prey are constantly available for capture. A group that consumes prey with a high vulnerability is likely to increase rapidly if its predation or fishing mortality is reduced. Conversely the increase will be moderated by lower vulnerabilities. There are three stages to modeling vulnerabilities. The first is to accept the default value of 0.3 for all groups. Secondly, vulnerabilities can be set according to the trophic level of the prey and finally the best method is to assign vulnerabilities on a group-by-group basis that enable the model to closely replicate time-series data (see Ainsworth 2004, this volume).

3. *Forcing functions can be used. Ecopath* with *Ecosim* has the capability of including effects not generated by trophic interactions or fishing. These were particularly significant in the English Channel model because the English Channel is located at the boundary of two bio-geographical regions (Southward and Boalch 1988). Variations in temperature will allow different species to be successful (see Pitcher and Forrest 2004, this

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volume). Hence, a warmer climate will have positive impacts on sole (*Solea solea)* stocks (Henderson 1994) and negative effects on cod (*Gadus morhua*) (O'Brien *et al.* 2001); (Planque and Fox 1998). Although sole fishing mortality increased between 1973 and 1995 their biomass simultaneously increased as warmer temperatures had a positive effect on recruitment. Forcing functions do not fix the biomass so a rapid increase in fishing or predation pressure will still reduce sole biomass.

One of the major problems with tuning the model is knowing what to change. Regarding sole, there is significant evidence that there is a correlation between temperature and recruitment, although the exact mechanism of this depends on the region (Rijnsdorp *et al.* 1992, Philippart *et al.* 1996, Henderson 1994). Hence it was justifiable to include this in the model. For other groups that had not been studied so extensively it was difficult to know which data to trust. Using Virtual Population Analysis will mean that past data becomes more accurate with each new assessment. Conversely, past estimates for a group may be based only on an expert's guesstimate or current techniques such as acoustic surveys may mean that the contemporary estimates are better. Consequently, although time-series data may suggest a change is necessary, high confidence in your contemporary model may mean that it is not changed.

It is at this stage of tuning that the pedigree screen in *Ecopath* is valuable. This gives an indication of what data can be trusted and provides a basis for tuning. For example, in the English Channel it is known that the abundance of sharks (blue sharks *Prionace glauca*, porbeagle *Lamna nasus* and tope *Galeorhinus galeus*) has decreased. This is attributable to both a reduction in prey species and increasing fishing mortality. The pedigree screen in *Ecopath* indicated that there was little confidence for most of the data for this group. Hence I could legitimately modify biomass, P/B and the vulnerability until the model predicted the decline in abundance that the literature seemed to indicate.

Tuning cannot overcome all of the inadequacies in a model. It can identify where functional groups may need to be sub-divided, particularly if temperature influences recruitment. Comparing the model's output to stock assessment data is a valuable exercise that can bring confidence to both the modeler and the policy maker that the results are realistic. In response to the original question in the Prince Rupert workshop concerning their value to decision-making (see Pitcher *et al.* 2002), we would affirm that yes these models are valuable when tuned to data and that enhancing the time-series data can only increase their predictive power.

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For discussion after the oral presentation of this paper, see page 150.

DEALING WITH MIGRATORY SPECIES IN ECOSYSTEM MODELS

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ABSTRACT

This paper sets out the logic for dealing with migratory species in the *Ecopath*-with-*Ecosim* whole ecosystem simulation modelling framework. Examples are provided from salmon and hake populations.

As a technical convenience, *Ecopath* models are bounded by arbitrary borders that allow the user to 'define' the system. This 'box' should be large enough so that interactions *within* the system add up to a larger flow than the interactions *between* the system and the ecosystems outside the box. In almost all cases, it is not possible to define such an area that includes the entire life history of all groups in the model. Furthermore, some groups only use a portion of the box, and never interact with other groups in the model (e.g., A in Figure 1), whereas another group's distribution may overlap with the defined ecosystem model (e.g., B in Figure 1). Neither example poses a significant problem when building an *Ecopath* model. In case A, simple accounting of trophic interactions determines which prey is consumed and which predators consume the group that has a limited distribution. In case B, the fraction of the stock that resides within the model area is used as the biomass input. But what potential problems arise as we move from static pictures of the ecosystem to dynamic changes over time?

Biomass dynamics can have profound effects on the distributions of species, *Ecosim* is a biomass dynamics model that uses the *Ecopath* inputs to calculate initial states. If the ellipse in example A, figure 1, represents the entire distribution of the species in question, then there are no real or potential problems in calculating biomass within the system over time. In example B, it is possible to assume that the fraction of the total stock remains constant over time, and there is little or no exchange across the boundary. At first this assumption may sound crazy, but consider species such as abalone that have limited mobility and small dispersal distances. A more realistic, and worrisome, case is represented in example C, where the distribution of a species changes over time. Here the area over which the stock is distributed is a function of stock size. When stock is reduced in abundance, though fishing activities, or perhaps increased predation, the range collapses to a smaller area of more favorable habitat. This phenomenon has been observed in many fish

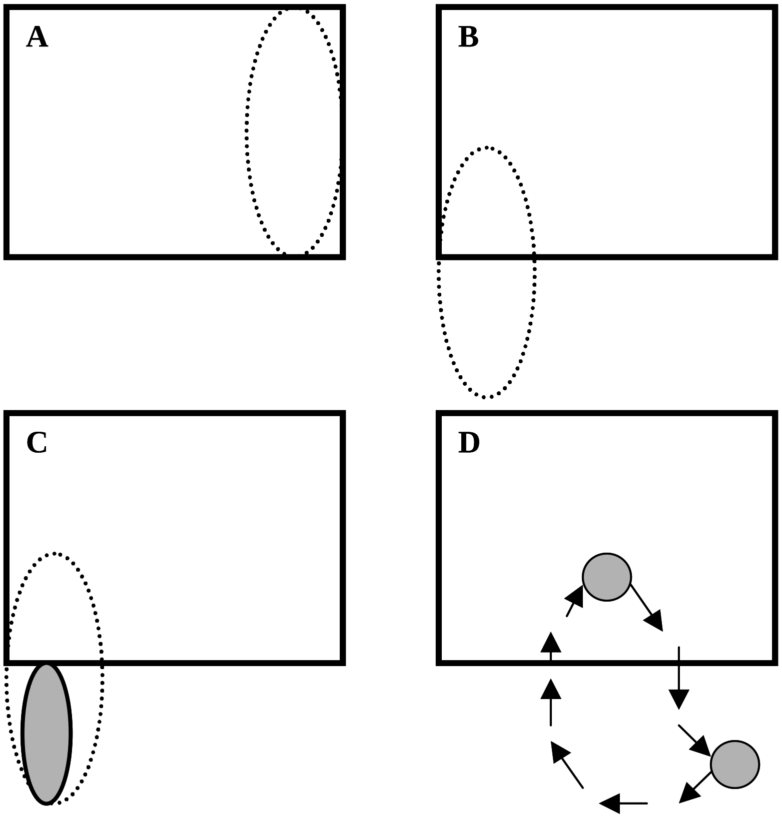

Figure 1. Four examples of ecosystem models, where model boundaries are represented by rectangles, and ovals represent distribution of a group in the model. A) Here the group is only partially distributed in the entire ecosystem, B) the distribution overlaps with ecosystem boundaries, C) distribution overlaps, but may collapse outside boundaries as stock is reduced, and D), arrows represent a complicated life history trajectory, where the gray circles might represent an area of importance such as spawning grounds, or where the fishery takes place.

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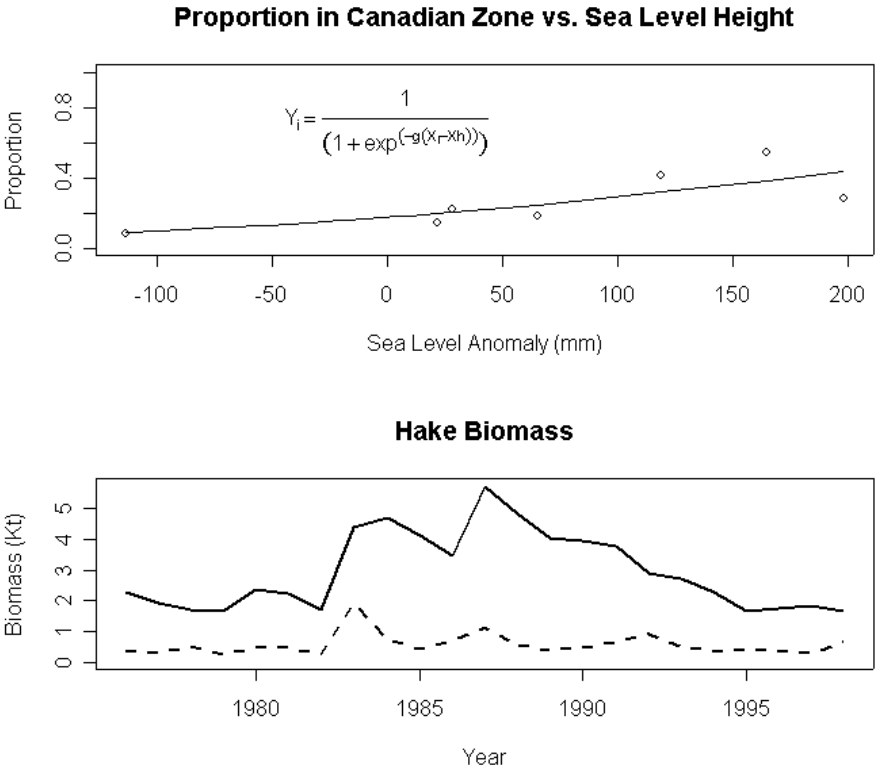

Figure 2. An example of estimating the fraction of a large migratory hake stock that enters the Canadian zone each year. The fraction of hake that enters Canadian waters is a function of sea level height, and we can use sea-level height to predict the fraction of the total stock (solid line) to generate a time series of biomass (dashed line) that enters the defined ecosystem.

stocks around the world (e.g., Atlantic cod off the east coast of Canada, Rose 1999). Such range collapses might involve the species leaving the defined ecosystem.

Example D in Figure 1 represents one of the more difficult issues to be represented in *Ecosim*. In this example, only a portion of the life history trajectory is within the defined ecosystem, and important events such as spawning or targeted fisheries occur both in and outside the define ecosystem. Pacific salmon are probably the best example of a species with a highly migratory life history, where fisheries occur in the oceanic, coastal and freshwater environments. Furthermore, there is another ecosystem that salmon play a functional role in, such as food for bears and eagles, and providing nitrogen through decaying bodies (Watkinson, 2001). Juvenile salmon spend one or more years rearing in freshwater and are subject to variable mortality rates due to competition with other stream inhabitants or anthropogenic impacts such as logging or urban development. Modellers of salmon using *Ecosim* need therefore to deal with this issue (Stanford 2002). Users of *Ecosim* should be aware that the stock recruitment relationship represented by split pool dynamics, assumes that everything that happens outside the box remains constant over time. There are however, some built in tools that can be used to represent variation in stock-recruitment production, or the effect of hatcheries. More on this later.

There are two phases to the 'Back to the Future' (BTF) approach. First is to reconstruct several ecosystem models, usually representing the present day, some time period in the past that might represent an unfished state, and one or more 'intermediate' states between pristine and present. The second phase is to simulate how one should optimally utilize the resources of an unfished ecosystem and compare results of such simulations to present day states. The reconstruction phase might include fitting *Ecosim* models to time series data to help parameterize the model. The second phase simply makes forward projections to explore alternative management policies. Each phase challenges the modeler with different problems for dealing with migratory species or populations that are only partially represented in the ecosystem model. In the reconstruction phase, time series information is required about changes in abundance within the model area. For example, if the distribution is changing over time, then what fraction of the total stock, or total catch, at each year were within the defined system? Often the data lack the spatial resolution that would allow total catch, or biomass to be partitioned among spatial areas. This should be taken into consideration when defining the boundaries of the system.

Species with complicated life history trajectories, where only part of the life history is represented in the model, are even more problematic. For example, dramatic changes in abundance may be a result of mortality that occurred outside the defined ecosystem, yet we search for mortality agents within the ecosystem to explain the observed declines. These problems carry forward into the simulation phase of BTF in addition to deciding how to represent life-history trajectories that occur outside the defined ecosystem. We cannot assume the freshwater phase of salmonid production remains constant over time! *Ecosim*, at its present stage of development, is not capable of explicitly modeling dynamic changes that might occur outside the system. Despite this limitation, there are alternative solutions for dealing with migratory species, or populations that share boundaries between neighboring ecosystem.

One of the most obvious options is to simply do nothing. That is, just assume that what happens outside the ecosystem remains constant over time, and assume that stocks that overlap the defined system are disconnected. Such assumptions may be valid for reasons such as limited dispersal, or because the biomass pool is simply too small to be of importance to modeling questions. An alternative option is to increase the scale of the model such that the entire distribution, or life history trajectory is included in the model (e.g. turn cases B and C in Figure 1 into case A). Exercising this option may be tricky for groups that have long distance migrations, as expanding to such large scales may introduce more problems with data. Having to add additional groups that live outside the previously defined ecosystem, may also require increased participation and substantial increase in the scale of the project.

There are a couple of alternative options for dealing with migratory species in reconstructed dynamic models using *Ecosim*. One such option is to impose a time series of biomasses on the ecosystem, where this time series is estimated independently of *Ecosim*. For example, biomass for a particular group could be estimated using single species models (incidentally, this should be done anyway to generate a fishing rate time series to drive fishing mortality in *Ecosim*), then read into *Ecosim*[1]. Also, the time series should be corrected for the fraction of the total stock that is within the defined ecosystem. As an example, Pacific hake populations off the west coast of Vancouver Island are part of a large migratory stock that winters in southern California and some fraction of the total stock migrates into Canadian waters in late spring-early summer. The stock is assessed every three years using information from fishery independent surveys, and the proportion present in the Canadian zone is a function of sea level height which is correlated with water temperature (Dorn 1995). Here the southern boundary of the ecosystem model is the Canada-US border, and the objective is to correct the assessment predictions to reflect hake biomass inside Canadian waters. Figure 2 presents a logistic relationship between sea level height and proportion in the Canadian zone. This logistic equation can be applied to the total stock to estimate a time series of hake biomass present inside the defined ecosystem (dashed line in bottom panel), this time series can then be read in as a forced biomass pool.

'Egg forcing' is an option for forcing split pool dynamics frequently used to represent enhancement programs such as salmon hatcheries. Salmon hatcheries, in some cases, have more than doubled smolt output into the marine environment, and in *Ecosim* this is just represented by a doubling of egg production in the forcing function. A time series of hatchery releases scaled to wild salmon production is read into *Ecosim* using the standard *.csv file and specified shape number. The shape number is then applied to egg production for the salmon group. A similar option could also be used to represent other disturbances that might have occurred in the freshwater phase of salmonids (e.g., set egg production to near zero to represent the catastrophic impacts of the Fraser canyon slide that nearly destroyed Fraser River sockeye stocks). This could apply to entrainment of fish larvae in cooling towers for nuclear power plants. The options are endless, but just require some time series data and a known scale of the effect of juvenile production. These time series effects on fish production could also be implemented in forward projections, where alternative hypotheses about the magnitude of the effect can be explored. For example how might the removal of hydroelectric dams affect eulachon populations in the Columbia River?

Another interesting simulation issue related to salmonids, anadromous fishes, and groups that move between two distinctly different ecosystems is how to connect the two systems. As an example, consider the life history of Pacific salmon, where the marine phase involves complicated migrations, consumption, predation, and variability in annual survival rates. These exact same processes also occur in the freshwater phase, where adult salmon are food resources for scavengers/predators such as bears and eagles (Watkinson 2001 and references therein). The remaining adults that survive the predator gauntlet are also responsible for egg production

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[1] Note that this time series should be scaled to *Ecopath* units (e.g. tonnes/km^{-2}), and use the '–1' option for the data type code in the *.csv file.

and recruitment, while juveniles remain in the freshwater environment for up to a year or more and face other challenges. If the ecosystem model only represents the marine phase of the salmonid life history, we might simply proceed with policy exploration using new high-tech gear that reduces by-catch and conclude it is safe to proceed with such developments. The new policy works great in the model and in practice, but grizzly bears are starving and going extinct in many watersheds. Oops! Clearly we should consider how our policy affects neighboring ecosystems, and the question is how do we do this?

For neighboring ecosystem models that share a couple of groups (consider a near-shore versus off-shore ecosystem, where one group forages and spawns near-shore during the summer months) it is a simple matter to combine two *Ecopath* models. In nature, species interactions can be direct (i.e. predation), or indirect (i.e. competition). In *Ecopath*, direct interactions are specified by setting a non-zero value in the diet matrix for predator j on prey i, and indirect interactions are specified when two groups share the same resource. It is possible to carry out the same mass balance exercise for two independent *Ecopath* models that are loaded into the same file. In other words, you can have two independent models of 10 groups each, or one model with 20 groups. When you balance these models, parameter estimates are the same if the two diet matrices are independent of each other. To connect the two ecosystems, to represent a group that moves between the two systems, simply recalculate the diet composition for that group, where some proportion P comes from one model, and 1-P comes from the adjacent model. Such an exercise has already been shown to work quite well for the Prince William Sound model (Okey and Pauly 1998), where the ecosystem was sub-divided into nearshore and offshore components. Since predator-prey interactions are specified in the *Ecopath* diet matrix, there is no problem moving into *Ecosim* and representing a group moving between the two systems. With a little programming experience, it is also possible to integrate *Ecosim* with other models that represent the dynamics of neighboring ecosystems.

For example, suppose we had a terrestrial model for salmon recruitment in the freshwater environment that includes predation and population dynamics of bears and eagles (Watkinson 2001). The input to this model is the number of adult salmon entering the river. Within each annual time step, we can pass predicted adult abundance of salmon from *Ecosim* to the terrestrial model, where bear and eagle dynamics are updated partly based on how much salmon was available. The terrestrial model then returns the number of juvenile salmon to *Ecosim* 1-2 years later, where *Ecosim* graduates the juveniles into adults and the process repeats for N years. Such a framework would provide insights about the affects of harvesting salmon in the marine environment on bears and eagles in the terrestrial environment.

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For discussion after the oral presentation of this paper, see page 148.

ESTIMATING THE EFFECTS OF PREDATOR-PREY VULNERABILITY SETTINGS ON ECOSIM'S DYNAMIC FUNCTION

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ABSTRACT

In the context of foraging arena theory, prey vulnerabilities are proportional to the flux of prey from safe refuges to feeding areas, where they are subject to predation. In the absence of empirical data, *Ecosim* modelers may use an approximation method to estimate prey vulnerabilities. Four such methods are evaluated in this report in their ability to permit *Ecosim* to generate predictions of abundance that resemble stock assessment time-series. The first method is to scale prey vulnerabilities proportionately to the trophic level of their predators ('predator control' hypothesis). The second is to scale vulnerabilities proportionately to the trophic level of the prey ('prey control'). The third is to apply a flat vulnerability to all groups. The fourth method customizes group vulnerabilities according to logical rules. Four *Ecosim* models are used to compare the assumptions. The results fall marginally in the favour of prey control. Three out of four models show improved dynamic functioning under this assumption and biomass trends are improved in 18 out of 32 functional groups (compared to 12 groups for predator control). Prey control was therefore adopted for all Back-to-the-Future applications. Ideally, each predator-prey combination should receive its own independent score. This will be addressed in later revisions of 'Lost Valley' policy search methodology.

Central to the dynamic function of *Ecosim* are the input prey vulnerabilities to predators. Vulnerability, a concept rooted in foraging arena theory, describes the flux of prey from safe refuges to feeding areas, where they are subject to predation (Walters *et al.*, 1997). The vulnerability parameter (v) is assumed by *Ecosim* to be proportional to the relative time spent feeding and hiding. Figure 1 shows a schematic representation of *Ecosim* vulnerabilities.

The vulnerability parameter is defined in *Ecosim* on a logarithmic scale from 0.01 to 1.0. Low prey vulnerability indicates bottom-up control; high vulnerability indicates top-down (Lotka-Volterra) control. Christensen *et al.* (2000) warn that strict bottom-up control in *Ecosim* tends to produce unrealistically smooth changes in prey and predator biomass that fail to propagate through the food web, while strict top-down control may cause rapid oscillations in biomass and unpredictable simulation behaviour (see also Mackinson 2002).

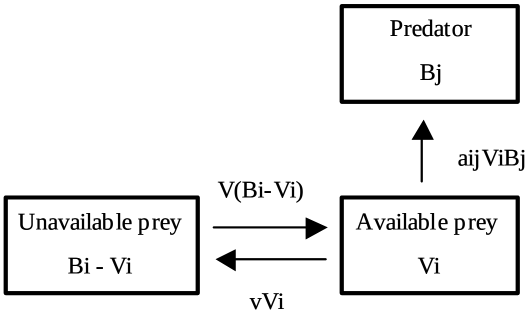

Figure 1. *Ecosim* vulnerabilities in the context of foraging arena theory. Vulnerability (v) describes the exchange rate between vulnerable prey biomass pool (Vi) and invulnerable pool (Bi-Vi). (a) describes predator (i) search rate for prey (j). Bj is predator biomass pool. Source: Walters *et al.* 1997.

Further, Cheung *et al.* (2002) suggest that using the blanket assumption (applied to all groups) of top-down control (>0.5 vulnerabilities) will generate a complex response surface with many optima; they found it difficult to find a global maximum when searching for optimal fisheries. Moreover, Martell *et al.* (2002) found that low blanket vulnerabilities impart on the system a high degree of resiliency to fishing effects. Models based on this assumption, they suggest, will return unreliably optimistic policy recommendations. The default setting in *Ecosim* describes a mixed condition (on the low end of the vulnerability spectrum as established by convention), where all prey vulnerabilities are set to 0.3. Cheung *et al.* (2002) report that a consensus emerged at the FAO/Fisheries Centre *Ecopath* workshop that scaling vulnerabilities in proportion to trophic level (TL) was more realistic than the blanket assumption. Here we test these methods as well as a more customized approach, which involves assigning group vulnerabilities according to logical rules.

Walters (*pers. comm.*) suggests that each predator-prey combination should ideally receive its own unique vulnerability since anti-predator defenses (e.g. behavioural, structural) may provide differential protection against various modes of predator attack. In lieu of vulnerability estimates derived from data, modelers may employ a shortcut - scaling vulnerabilities proportionately to either predator or prey trophic

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Ainsworth, C. (2004) Estimating the Effects of Prey-predator Vulnerability Settings on *Ecosim*'s Dynamic Function. Pages 45–47 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.

level (TL). While Cheung *et al.* (2002) were the first to try the latter method (repeated by Mackinson *et al.* (2002), Martell *et al.* (2002) and others), the former is tried here for the first time. These two techniques make different assumptions about trophic interactions. The former 'predator control' assumption contends that a prey species will be more vulnerable to high TL predators than low TL predators. The alternate hypothesis, 'prey control', implies that low TL prey is more vulnerable to predators than high TL prey.

This paper examines whether prey or predator control hypotheses enable *Ecosim* to predict a biomass trend that more closely resembles a time-series of biomass from stock assessment, and whether either technique improves on the default (all vs =0.3) assumption. Finally, we test the ability of a more customized vulnerability regime to recreate known biomass trends.

To test these issues, I have used four *Ecopath* models of past times from various authors along with time-series abundance estimates of their (commercial) functional groups: 1970 Bay of Biscay (Ainsworth *et al.*, 2001), 1950 Strait of Georgia (Dalsgaard *et al.*, 1998), 1973 English Channel (Stanford 2002) and 1950 northern British Columbia (Ainsworth *et al.*, 2002).

METHODS

Using default *Ecosim* settings for all four models, I first set the vulnerability of prey groups in proportion to their predators' trophic level (predator control), and then in proportion to their own (prey control). In the *Ecosim* interface (under the 'flow control' tab), vulnerabilities are entered vertically for predator control and horizontally for prey control. For both trials, the range of vulnerabilities was set from 0.8 for high TL groups to 0.2 for low TL groups.

A simulation was run for each model, under each hypothesis. The biomass trend, obtained from *Ecosim*'s output CSV file, was compared to stock assessment records with a non-parametric Spearman's correlation test.

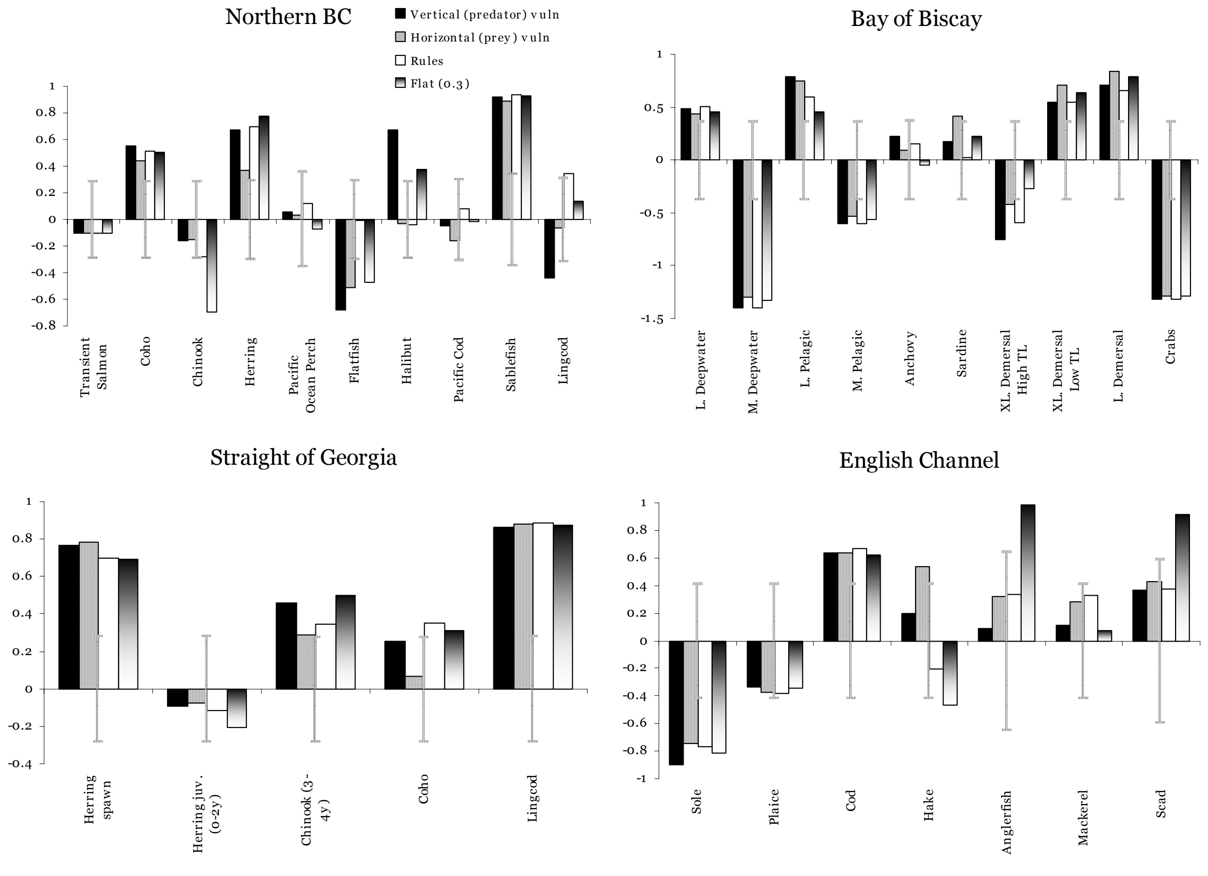

Figure 2. Correlation of biomass outputs from four *Ecosim* models with time-series stock assessment under four assumptions of prey vulnerability. Dark bars show predator control; stippled bars show prey control; white bars show logical rules; shaded bars show all vs = 0.3. Crossbars show correlation needed for significance at >= 0.05.

RESULTS

Figure 2 shows the correlation of stock assessment information with the biomass trend predicted by *Ecosim*. Dark bars show the correlation under predator control, light bars show correlation under prey control. Significance level at a=0.05 is indicated by crossbars for each functional group.

Prey control vulnerabilities allow *Ecosim* to generate a biomass trend that more closely resembles stock assessment information in 18 out of 32 functional groups studied; predator control vulnerabilities perform better for 12 groups and the two methods perform equally well for 2 groups. Prey control generates a closer overall correlation in all models except northern BC, where only 3 functional groups correlate better under prey control and 6 groups correlate better under predator control.

CONCLUSION

In most cases, prey control vulnerabilities allow *Ecosim* to predict an index of relative abundance that more closely conforms to stock assessment than the alternate hypothesis, predator control. Unfortunately, the BC model (which is the subject of CUS BTF applications) does not perform better under this assumption. However, we judge predator control to be less supportable, since it requires that prey know which predator is attacking them. Although evolution has probably equipped them with this ability to some extent, in the absence of supportive data it is safer to assume that a prey has adjusted its transfer rate to protect itself equally from all likely predators encountered.

Ideally, we would examine each predator-prey combination individually; this is the next step to improve the dynamic function of the BC model. Avdin and Friday (2001) found that vulnerabilities in the lower order prey groups were most critical to the simulation; this is where fine-tuning should begin.

Consequently, vulnerabilities for all *Ecosim* models used in the Lost Valley policy search (Ainsworth *et al.* 2004, this volume, Ainsworth, 2004a and 2004b) were set to prey control (vulnerabilities proportional to prey TL in the range 0.2-0.8 by convention).

ACKNOWLEDGEMENTS

I would like to thank Dr Carl Walters and Dr Villy Christensen for their input and discussion.

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POLICY SEARCH METHODS FOR BACK-TO-THE-FUTURE

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ABSTRACT

Using the policy search routine in *Ecosim* we identify the pattern of exploitation that would allow us to gain the most benefit from restored '*Lost Valley*' ecosystems of northern BC and Newfoundland. The policy search determines the fishing mortalities for each gear type that will maximize its objective function over a 50 year-simulation. Five objective functions are considered: ecological, economic and social, as well as a mixed objective and a conservative 'portfolio log utility function' that resists altering the ecosystem far from its baseline. The ecological function increases the abundance of slow-growing groups, the economic function maximizes rent from the system, and the social function maximizes fishery employment. A mixed objective function combines economic, social and ecological priorities. The portfolio log-utility function combines these priorities as well, but includes a risk aversion algorithm. Using the mandated rebuilding routine, constraints were included in ecological and mixed objective runs for northern BC models to prevent extinctions. Four time periods are evaluated as starting points for the optimization in each ecosystem (1750, 1900, 1950 and 2000 for northern BC and 1450, 1900, 1985 and 1995 for Newfoundland); the most valuable of these represents possible restoration goals. Three fleets are considered in their ability to harvest the restored system. The '*lost valley*' fleet includes twelve and sixteen fisheries in northern BC and Newfoundland, respectively. These allow a minimal level of bycatch and discards. The '*no recreational*' fleet omits the sport fishery and the '*no trawlers*' fleet omits groundfish trawl and shrimp trawl. We confirm that the search routine has identified the optimal policy by conducting additional trials using random fishing mortalities a starting point rather than *Ecopath* baseline values. The restored systems are subjected to 100 years of simulated fishing including a 50-year (dynamic) fishery development phase and a 50-year (steady-state) equilibrium phase. Seven valuation techniques examine the resulting harvest profile and ecosystem condition to measure the success of each restoration period, fleet and harvest objective. Economic valuation considers the conventional and intergenerational net present value of the harvest profile. Ecological valuation measures biodiversity of the restored system based on the Q90 statistic, the change in ascendancy throughout the system, and the occurrence of local extinctions. Social valuation determines the total number of jobs generated by each restoration/harvest scheme, and the employment diversity across fishing sectors.

INTRODUCTION

Harvesting the Lost Valley

The methods presented here describe an attempt to determine the combination of gear types and fishing mortalities will allow us to optimally exploit the restored *Lost Valley* ecosystems of northern BC and Newfoundland under a variety of policy objectives. The general principles and methodology of the *Lost Valley* approach is described in Pitcher (2004b, this volume) and in Pitcher *et al.* (2004).

Briefly, we have constructed *Ecopath* with *Ecosim* (EwE) ecosystem models of northern British Columbia and Newfoundland to represent the marine environment as it appeared in the distant past, the recent past and the present. These periods are here referred to as *Lost Valley* ecosystems. In order to evaluate these as possible restoration goals for the future, we determine the optimal pattern of fishing mortality per gear sector (using the policy search routine in *Ecosim*) that will generate the greatest benefit in a specified harvest objective. Five objectives are tested that, together, span the spectrum of human use versus conservation. In order to explore which combination of gear types (see Ainsworth 2002) should be used to harvest the restored system, we conceptualize an idealized *lost valley* fleet, which includes only responsible fisheries (Pitcher 2004b this volume, Pitcher 2004a, Pitcher *et al.* 2004). We also test two abbreviated versions: one that includes *no recreational* sector and one that includes *no trawlers*.

We apply the optimal fishing patterns to the restored system, simulating 100 years of harvest (including a 50-year dynamic period and a 50-year steady-state equilibrium). The resulting harvest profiles are then valued in economic, ecological and social terms.

By quantifying the benefits that each historic period has to offer as a restoration goal, we can judge what costs are justified in achieving restoration. Future work (Ainsworth, *in prep*) will look at strategies for achieving restoration in northern British Columbia, and determine how far into the past we should restore to maintain cost-effectiveness.

Ainsworth, C., Heymans, J.J. and Pitcher, T.J. (2004) Policy Search Methods for Back to the Future. Pages 48–63 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.

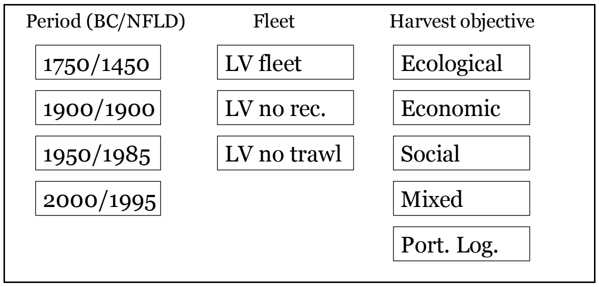

Figure 1. Policy searches. The optimal fishing policy is determined for each period, under each fleet and harvest objective. Sixty runs were conducted for each ecosystem. Re-trials with random F starting points validate the optima.

METHODS

The 1750, 1900, 1950 and 2000 *Ecopath* base models of northern BC used here are based on Ainsworth *et al.* (2002); the 1450, 1900, 1985 and 1995 models of Newfoundland are based on Pitcher *et al.* (2002). For both the west and east coasts, this exercise will test the ability of our three proposed responsible fleets to harvest each of our restoration goals (based on four historical periods), in order to maximize five harvest objectives (Figure 1). A total of sixty runs were conducted for each ecosystem.

For the northern BC models, we verify that the policy search has found the optimal combination of fishing mortalities under its objective by conducting additional searches using random fishing mortalities (F) as the starting point for the optimization procedure, rather than *Ecopath* base values. Ideally, the search algorithm will locate the global maximum on the response surface curve regardless of the starting values of F.

Ecosim parameterization

Most *Ecosim* parameters were left as default. Appendix A Tables A1, A2 and A3a and b detail parameters used to initialize *Ecosim* including specific run information, group information and juvenile/adult linking parameters. The juvenile/adult linking parameters for cod and American plaice in Newfoundland were different for 1985 and 1995; the values for 1985 were used for the 1900 and 1450 models.

Prey vulnerabilities to predators (Table A4a and b) were set in proportion to prey trophic level, where the lowest trophic level prey receives a vulnerability of 0.2 to its predators and the highest trophic level prey receives a vulnerability of 0.8. The prey vulnerabilities were varied for the different Newfoundland time period models (Table A4b). Scaling vulnerabilities proportional to prey trophic level rather than predator trophic level was chosen for reasons discussed in Ainsworth (2003).

Initializing the Policy Search

Fleets

The *Lost Valley* fleet chosen for the west coast includes groundfish trawl, shrimp trawl, shrimp trap, herring seine, halibut longline, salmon freezer troll, salmon wheel, live rockfish, crab trap, clam dredge, aboriginal and recreational fisheries. The *Lost Valley* fleet chosen for the east coast includes bottom and shrimp trawls, recreational and First nations fisheries, cod traps, capelin seine, longline, midwater trawl for redfish, traps for lumpfish, snow crab, inshore crabs and lobster, salmon wheels, pole and line, clam dredges and urchin diving. Retained bycatch on the west coast occurs in all fleets except salmon wheel, live rockfish, clam dredge and aboriginal fisheries, while cod longline, cod traps and redfish midwater trawls retain bycatch on the east coast. Discards were assumed to be minimal, only groundfish/shrimp trawls and clam dredge produce discards on both coasts.

Percentages listed in Appendix B Tables B1a and b (catch) and B2a and B2b (discards) refer to the proportion of total group biomass caught in the first year of the optimal policy exploration; these values will change throughout the simulation. Group biomass for each time period is listed in Ainsworth *et al.* (2002) for the west Canadian coast and in Pitcher *et al.* (2002) for the east coast.

Generally, the fisheries were set to initially catch 2.5% of the total biomass of their target groups annually, and 0.5% or 0.25% of retained bycatch groups. In northern BC, major discards were set at 1.25% of group biomass, while minor discards were set to 0.25% or 0.025% of group biomass. In Newfoundland discards were all set at 2.5% of group biomass for fish and 0.1% for birds. Catches and discards vary between time periods in proportion to system biomass. As the policy exploration progresses these values are free to change.

The baseline values of fishing mortality should have no impact on the final policy. However, we had to use initial Fs small enough to avoid having to rebalance the model for each of our trial fisheries (thereby affecting the search results), and large enough so that the routine's outputs (which are multipliers of the base F) remain small for convenience. Since the policy search routine was designed to accommodate much larger initial fishing mortalities (as would be seen when evaluating any real-world fishery for example), the output multipliers deliver an uninformative ">60" string, when optimal Fs are greater than sixty times the baseline value[1]. Careful choice of baseline fishing mortalities can circumvent this software limitation.

For both coasts there are three fleets tested in the present analysis. First, the *Lost Valley* fleet, secondly the *Lost Valley* fleet minus recreational gear and finally the *Lost Valley* fleet minus trawlers (shrimp trawl and groundfish trawl on both coasts). The aboriginal fishery was held constant, omitted from the policy search for all fleets and objectives. In 'no recreational fishery' trials, the recreational fishery was removed from the base model and omitted from the policy search. Similarly, shrimp trawl and groundfish trawl were removed from the model and omitted from the policy search for the 'no trawl' trials.

Policy objectives

Five *Lost Valley* policy objectives were considered: ecological, economic, social, mixed and portfolio log utility optimization. These are discussed in the following sections. Since the search routine does not normally attempt to preserve species biodiversity, we entered into the ecological and mixed objective runs of the northern BC ecosystem a constraint (using mandated rebuilding) that there should be no extinctions. The portfolio log utility optimization, in only a few cases, would recommend harvest policies that included extinction of vulnerable groups. A constraint was added to prevent this (see below). For the economic and social optimization runs, no such constraints were included – extinctions were allowed under these objectives. The optimization procedure was not constrained for any of the Newfoundland models.

Mandated rebuilding

The mandated rebuilding routine was designed to allow users to identify fishing policies that would facilitate the rebuilding of a depleted stock. In this exercise we do not try to increase stock size, but use the routine (in the BC models) to prevent extinctions by setting the biomass goal to one times the *Ecopath* base level. This novel procedure works well to maintain a steady abundance in protected groups. Although in ecological and mixed objective runs many functional groups tended towards extinction, it was possible in all cases to identify a key group, which when protected, allowed the run to proceed without any extinctions. The smallest mandated rebuilding weight that would stop extinctions was used, so as not to disturb the optimum policy any more than necessary.

Initially, with the BC trials, we tried to prevent extinctions ecological and mixed runs by increasing the biomass/production (B/P) ratio of key groups. As explained below, the ecological objective (present in the mixed objective run as well) increases the biomass of functional groups with high B/P ratios. Groups prone to extinction would then have an inflated importance in the policy search. However, this technique was rejected for mandated rebuilding since there was no single set of B/P values found that would stave off extinctions when commonly applied to all models.

Software difficulties

To prevent the policy search program from becoming unstable, it was sometimes also necessary to use mandated rebuilding to prevent groups from exploding or going extinct. The economic optimization runs were particularly prone to instability, 8 out of 12 economic runs in northern BC required restraint on problem functional groups to allow the program to operate. Two out of 12 social runs required manipulation. We gave mandated rebuilding a low priority in the policy search: enough to allow the search routine to function, but not enough to stop extinctions (since we did not wish to perturb the outcome any more than was necessary). In northern BC, the migratory group, *transient salmon*, in particular was prone to exploding in abundance under most policy objectives causing a computer crash. It was often necessary to restrict its growth to a factor of about eight times the baseline in order to avoid crashes. The problem in modeling migratory species has to do with the diet matrix. When groups feed primarily out of the study area, their food source is not subject to systemic fluctuations in productivity. In times of low system productivity, biomass of the migratory group is inappropriately bounded only by top-down control. For a complete discussion on the problems of migratory species in *Ecosim* modeling refer to Martell (2004, this volume). Less often than transient salmon, it was also

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[1] In preliminary work, the EwE code was modified to return numerical multipliers beyond sixty times. However, this version of the code was abandoned when a more fundamental bug was discovered in the policy search routine that limited the number of fleets that could be examined. Unfortunately, the next version of EwE, which corrected the more severe bug, did not include the maximum-multiplier fix.

necessary to manage skates and juvenile/adult turbot to allow the policy search to complete itself.

Mandated rebuilding was not used in the Newfoundland policy exploration; unstable runs are indicated in Appendix B Table B4b. Social runs proved the most problematic for the Newfoundland trials, with all the social runs in 1450 and 1985 becoming unstable. Unstable runs resulted in either huge oscillations of biomass, a collapse in biomass (especially salmon, shortfin squid and large and small crabs), or an explosion of biomass (adult Greenland halibut in the 1450 model).

Policy objectives

Ecosystem

Under the ecosystem policy objective, the search seeks to maximize the occurrence of long-lived species. Pristine and unfished ecosystems have been characterized as having many large slow-growing animals (Odum 1969). Therefore, using a high biomass/production (B/P) ratio across functional groups as a surrogate to describe this condition, the ecosystem policy objective suggests an exploitation profile that will increase the abundance of slow-growing functional groups. Cheung *et al.* (2002) were the first authors to use this technique. The B/P ratios used in the present exercise are listed in Appendix A Table A5a and b. However, the ecological objective does not necessarily preserve species diversity; it will sacrifice high turnover groups (e.g. predators, competitors) in favour of the long-lived animals. Therefore, (in the BC models) we used mandated rebuilding to protect against extinction of any functional group under this policy objective.

Economic

The economic objective seeks to maximize total rent from the system. Under this objective, high value fisheries will be favoured at the expense of low value fisheries, even to the extent of causing extinctions among detrimental groups (e.g. predators, competitors). We do not expect this run to preserve biodiversity. Economic valuation methodology is presented in Ainsworth and Sumaila (2004a).

Social

The social optimization will increase the number of jobs by eliminating fisheries with a low number of jobs per catch value in favour of more labor-intensive gears. Appendix B Table B3a and b give the jobs per catch value used for initialization. At 15 jobs per catch value unit (an estimate), the recreational fishery of northern BC employs three times as many people as the next highest fishery. Relative values were estimated by expert opinion (Pitcher, *pers. comm.*).

Mixed

The mixed objective combines ecological, economic and social elements. The search routine attempts to maximize the total objective function (the weighted sum of all components). Mackinson (2002) tried a similar mixed objective function on a model of the North Sea. He found that the relative improvement in ecosystem criteria consistently failed to match the relative improvement of social and economic criteria and it did not improve markedly as a higher relative weight was given. However, that author used much smaller relative weightings for ecology than the present paper (i.e. the largest relative weighting he applied was 10, 1 and 1 for ecological, social and economy). Zeller and Freire (2002) likewise found that the relative improvement over baseline of ecology was quite invariant to the weighting given to the ecological objective. Buchary *et al.* (2002) also found that a 1,1,1 mixed search for ecology, economics and social benefit results in an optimal policy that is very similar to their social optimization. These authors used a low relative weighting, with the ecological function receiving the same weight in the policy search as economy and social (i.e. 1, 1 and 1 for ecology, economy and social functions).

However, it is evident that entering equal weightings in the *EwE* software panel does not result in an equal improvement in criteria over *Ecopath* baseline. Since there is no intrinsic comparability between the three objective functions, then the relative weightings used to parameterize the search are meaningless and so a 1:1:1 ratio between the three objective functions does not imply that the policy search will increase all objectives evenly. Rather, only the relative improvement in each field over baseline is significant. We therefore adjust the weightings iteratively, based on the overall figure given by the completed search, so that each factor influences changes in the overall figure by an equal (or the desired) amount. This technique has been used in the LV work reported in Pitcher *et al.* (2004).

From the baseline condition, we find that in general, a much higher relative weighting must be given to ecology in order to achieve an equal improvement among mixed factors (see Pitcher

2004b, this volume). In this paper, the relative weighting of the three fields were determined in such a way as to minimize variance between the overall improvement values of the three functions. It turned out that a relative weighting of 1, 1 and 100 for economic, social and ecological priorities was found to consistently produce the most equal increase as measured from the final line of multipliers in the policy search. This ratio was therefore adopted for all BC runs. Variance of the relative ecological, economic and social improvements for BC runs are presented in Table B4a.

The Newfoundland models required a relative weighting of 0.1, 0.1 and 100 for economic, social and ecological weightings to obtain an equal increase in each priority. Thus, the ecological priorities had to be three orders of magnitude higher than the social and economic priorities to get similar outcomes for these three functions (as opposed to the two orders of magnitude required by the BC models). The only Newfoundland model that did not conform to the 0.1:0.1:100 ratio was 1450 (see Appendix B Table B4b).

The very high value required for ecological improvement in both ecosystems suggests that it is more difficult to manage an increase in the B/P surrogate than it is to increase rent, for example (i.e., *Ecosim* must structure virtually the entire strategy towards ecological gain in order to produce a minimal increase in average B/P of the system). Although the relative weightings required to levy an equal improvement across criteria will be model-specific. The relative insensitivity of the ecological function is also noted by Mackinson (2002), Zeller and Freire (2002), Buchary *et al.* (2002) and Pitcher *et al.* 2004.

Portfolio Log Utility

The recently devised portfolio log utility function attempts to account for the inherent uncertainty in changing the system far from its base state. Christensen *et al.* (2000) and Christensen and Walters (2004) provide a more detailed description of this *Ecosim* subroutine. Policies that promise the greatest benefit tend to carry with them the greatest risk, since the extreme combination of fisheries required to manipulate the ecosystem into a hyper-productive state will change the system far from its present condition. Such a policy may, for instance, involve destroying competitors and predators of the most valuable species, as is done in agriculture.

In portfolio log utility the user enters three parameters. Prediction variance describes the amount of uncertainty associated with changing the ecosystem far from its baseline. A high value will increase the discounting rate (reducing the net present value of future benefits), and make large returns unappealing when they require drastic manipulation of the ecosystem. Existence value defines the worth one assigns to the continued existence of functional groups: assigning a high value to this parameter will maintain a diverse biological ‘portfolio’ in economics terms. Finally, users enter a coefficient that modifies the net present value from the system (the sum of profits from all functional groups, discounted over time). A high value of this can make risky policies worthwhile.

For some runs with the BC models there was a precarious balance between receiving a policy recommendation that included extinctions, and receiving a flat line (zero change from base state). To fine-tune these runs we added a very small prediction variance, from 0.02 to 0.003. This fix helps prevent extinctions by devaluing daring portfolio choices. Only the lowest existence value that would still prevent extinctions was used. With the Newfoundland models this was not a problem. We only used existence values without having to use prediction variance. The existence values used for the Newfoundland models ranged from 0.01 to 0.1 (Appendix B Table B4b).

The portfolio log utility trials are very stable. Runs change slowly from the base state, and are not subject to the same wild fluctuations in biomass often seen when using the other policy objectives. This is the most conservative method. We do not expect high returns from the system compared to ecological, economic, social or mixed runs.

Verification of optimal policy

For the northern BC models, we next repeated each optimization 25 additional times, using random fishing mortalities as the starting point for the optimization, rather than *Ecopath* base values. Ideally, each replication should result in the same optimum fishing pattern (i.e., locate the global maximum on the response surface). However, prior investigations revealed that random F starting points do not necessarily allow the search routine to converge on the same maximum. Rather, the resultant ‘optimal’ fishing mortalities seem to cluster around common peaks, indicating that the search can stall on local maxima of the response surface.

In the CUS BTF results report for BC models

(Ainsworth *et al.* 2004), a two-way analysis of variance tests whether the 25 treatments have generated a statistically similar pattern of fishing mortalities. Results from the second factor, gear type, are discarded since we expect fishing mortality to vary between gear types. If the random F runs are shown to be dissimilar, this may indicate that the policy search routine has identified two or more local maxima for a given scenario, or alternatively, that the search routine has identified a single, broad peak (i.e. a plateau) where major variation in the harvest pattern yields an equivalent improvement over baseline.

Multidimensional scaling (MDS) offers a method to differentiate between these possibilities. Using SPSS v.10.0 statistical software, MDS is performed on a subset of runs (chosen to demonstrate the potential of this analysis in describing the shape of the response surface). MDS reduces all factors affecting scenario performance (i.e. fishing mortalities per gear type) to two dimensions, allowing us to sketch the shape of the optimal peak and/or detect the presence of local maxima. Such an approach may be used to judge the robustness of a harvest recommendation for management; however, more random F runs would be required to fully explore the shape of the response surface.

If a recommended harvest policy resides on a narrow peak, than any variation from the specified optimal fishing pattern may result in sub-optimal harvests. If however, the identified maximum resides on a broad peak, than deviations in fleet-effort structure may still result in a near-optimal manipulation of the ecosystem. The latter situation may represent a more robust goal for management than the former.

Valuation indices

Having determined the optimal combination of fishing mortalities per gear type that will maximize our five objective functions for each restored period, we then simulated a 100-year harvest regime (50 years dynamic and 50 years equilibrium) under each of our 3 idealized fleet structures. The resulting harvest profile was evaluated using two economic measures: conventional and intergenerational net present value (Sumaila and Walters 2003, 2004; Sumaila 2001, Sumaila and Bawumia, 2000). Economic valuation methodology is discussed in Ainsworth and Sumaila (2004, this volume).

The ecological success of the restoration/harvest scheme was determined using three valuation measures: the Q-90 statistic, system resilience and presence of local extinctions. Based on Kempton's Q index (Kempton and Taylor 1976), the Q-90 statistic is a measure of biodiversity that concerns species evenness. It looks at the slope of the cumulative species abundance curve between the 10 and 90 percentiles (see Ainsworth and Pitcher 2004, this volume, for methods). A second index involves measuring the resiliency of the system to fishing using ecosystem redundancy from network analysis (see Heymans 2004, this volume, for methodology and theory). The third measures the risk of local extinctions in composite functional groups (see Cheung and Pitcher 2004, this volume).

Social valuation measures include relative number of jobs created and employment diversity. Relative number of jobs created by an optimal plan is calculated as the product of total catch value (i.e. all simulation years summed) and the gear-specific jobs per unit catch value (Tables B3a and b). Employment diversity across fishing sectors is calculated after Atteran (1986). Ainsworth and Sumaila (2003b) describe how this index was applied to BTF methodology.

Using Kendall's coefficient of concordance (W; Kendall 1962), we finally determine the ability of each restoration period, fleet structure and harvest objective to maximize these economic, ecological and social valuation measures. Specific expectations are discussed below.

All valuation results will be presented in Ainsworth *et al.* (2004) for British Columbia and Heymans *et al.* (2004) for Newfoundland.

DISCUSSION

Ecosystem value will depend mainly on what period is restored. The pre-contact systems have in them the greatest biomass of valuable commercial groups; we therefore expect this period to permit the most valuable fisheries – scoring high in the economic analyses. On both coasts, models of the recent past represent a more depleted state than do models of the distant past; these will not be able to generate as much economic benefit.

Since the conventional model of discounting places most value on the immediate future, we expect also that the pre-contact and 1900 runs will do especially well under this valuation scheme. These simulations start at a high level of biomass and *Ecosim* can fish down the natural capital, generating immediate revenue and leaving the system in a depleted (but more productive) state. Intergenerational discounting, however, will not favour the immediate profit as strongly; it will be content to leave more natural capital in the sea and maintain high harvests farther into the future. Therefore, although the pristine states (pre-contact and 1900) should always produce greater revenues than the more depleted systems (1950 and 2000 in BC; 1985 and 1995 in Newfoundland), the difference will be more apparent under conventional discounting than under intergenerational discounting because of the relative shape of harvest profiles. The more recent time periods will require rebuilding in order to generate maximum monetary returns. Their harvest profiles will slope upwards (or slope downwards less sharply than distant past periods); therefore, they will score proportionately better under intergenerational discounting.

Of the three fleets tested (*Lost Valley*, *no recreational*, *no trawl*), we expect the *Lost Valley* fleet to generate the most valuable harvest of the restored system for two reasons. First, the additional gear types allow the search routine to probe for the best policy with improved dexterity. Since the policy search is at liberty to minimize any of its fleets, allowing more gear types can only enhance the search routine's ability to manipulate the ecosystem into its most commercially valuable condition. Secondly, the CUS BTF models (at this stage) do not consider the problems of trawl damage, ghost fishing, or any other deleterious gear effect. In the simulation, there is no ecological or economic benefit associated with preserving habitat, and nothing is to be gained by restricting damaging fisheries (except perhaps a coincidental reduction in discards). Similarly, ecologically responsible fleets that omit damaging gear types will not be credited with their full ecological benefit. Future efforts to model the system spatially will allow us to include these considerations.

We expect the mixed objective function to yield exploitation profiles similar to the ecological runs. Our preliminary efforts have confirmed the findings of other researchers that the ecological objective is the most difficult to maximize – the policy search must virtually disregard the other objective functions in order to increase the ecological criteria. For example, a typical ecological run will rarely exceed a 10% improvement in the B/P surrogate over 50 years, under even the most vigorous attempts to do so. Rent and jobs, on the other hand, regularly exceed a seven times improvement on the economic and social objective functions. Where improving the ecology involves a slow restructuring of the ecosystem (and a sacrifice in catch), the economic and social functions need only to redistribute fishing effort to increase rent or jobs. This is especially true since the economic and social functions were not constrained by the requirement to avoid extinctions. Further, the search routine will be hard pressed to improve the B/P ratio of the already under-exploited pre-contact and 1900 models. As an objective, it is easier to disassemble the ecosystem, particularly one that is under-exploited, than it is to build it.

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For discussion after the oral presentation of this paper, see page 153.

APPENDIX A
ECOSIM PARAMETERS

Table A1. Run Information for both ecosystems.

Duration of simulation (years)	50
Integration steps (per year)	100
Relaxation parameter [0,1]	0.5
Discount rate (% per year)	5
Equilibrium step size	0.003
Equilibrium max. fishing rate (relative)	3
Number of time steps for averaging results	5

Table A2. Group Information for both ecosystems

Maximum relative feeding time	2
Feeding time adjustment rate	0.5
Fraction of 'other' mortality sensitive to changes in feeding time	1
Predator effect on feeding time	0
Density dependant catchability	1
QBmax/Qbo	1000

Table A3a. Stage (Juvenile/adult linking parameters) for northern BC.

	Herring	Piscivorous rockfish	Turbot	Flatfish	Halibut	Sablefish	Lingcod	Pollock	Pacific Ocean Perch	Pacific Cod
Min. time as juv. (rel. to orig. setting)	1	1	1	1	1	1	1	1	1	1
Max. time as juv. (rel. to orig. setting)	1.0001	1.0001	1.0001	1.0001	1.0001	1.0001	1.0001	1.0001	1.0001	1.0001
Recruitment power parameter	1	1	1	1	1	1	1	1	1	1
Weight (g) at transition to adult group	1	1	1	1	1	1	1	1	1	1
Age (year) at transition to adult group (tk)	2.1	16	4.5	4.5	10	4.5	4	2.3	16	2.3
Wavg / Wk (Av. adult weight / weight at transition)	2	2.7	2	2	1.357	1.88	3.684	3.597	2.7	1.725
K of the VBGF (/year)	0.47	0.05	0.243	0.243	0.08	0.3	0.263	0.373	0.88	0.27
Base fraction of food intake used for reproduction	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3
Fraction of increase in food intake used for growth	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8

Table A3b. Stage (Juvenile/adult linking parameters) for Newfoundland

	Cod		American plaice		Greenland halibut
	1985	**1995**	**1985**	**1995**	
Min. time as juv. (rel. to orig. setting)	1	1	1	1	1
Max. time as juv. (rel. to orig. setting)	1.0001	1.0001	1.0001	1.0001	1.0001
Recruitment power parameter	1	1	1	1	1
Weight (g) at transition to adult group	1	1	1	1	1
Age (year) at transition to adult group (tk)	7	7	5	5	9
Wavg / Wk (Av. adult weight / weight at transition)	1.247	1.051	3.427	2.299	2.000
K of the VBGF (/year)	0.07	0.07	0.099	0.099	0.025
Base fraction of food intake used for reproduction	0.3	0.3	0.3	0.3	0.3
Fraction of increase in food intake used for growth	0.8	0.8	0.8	0.8	0.8
Weight at transition	1.927	2.353	0.104	0.095	
Adult weight	2.403	2.474	0.358	0.218	

Table A4a. Flow control in northern BC.

Functional Group	1750	1900	1950	2000
Seals, sea lions	0.80	0.80	0.80	0.80
Transient salmon	0.50	0.50	0.50	0.50
Coho salmon	0.76	0.76	0.76	0.76
Chinook salmon	0.71	0.71	0.71	0.71
Small squid	0.57	0.57	0.57	0.57
Squid	0.71	0.71	0.71	0.71
Ratfish	0.57	0.57	0.57	0.57
Dogfish	0.64	0.64	0.64	0.64
Juvenile pollock	0.53	0.53	0.53	0.53
Pollock	0.60	0.60	0.60	0.60
Forage fish	0.46	0.46	0.46	0.46
Eulachon	0.48	0.48	0.48	0.48
Juvenile herring	0.47	0.47	0.47	0.47
Adult herring	0.51	0.51	0.51	0.51
Juvenile POP	0.48	0.48	0.48	0.48
Adult POP	0.52	0.52	0.52	0.52
Inshore rockfish	0.72	0.72	0.72	0.72
Juvenile picivorous rockfish	0.53	0.53	0.53	0.53
Adult picivorous rockfish	0.59	0.59	0.59	0.59
Juvenile planktivorous rockfish	0.49	0.49	0.49	0.49
Adult planktivorous rockfish	0.62	0.62	0.62	0.62
Juvenile turbot	0.73	0.73	0.73	0.73
Adult turbot	0.77	0.77	0.77	0.77
Juvenile flatfish	0.51	0.51	0.51	0.51
Adult flatfish	0.53	0.53	0.53	0.53
Juvenile halibut	0.70	0.70	0.70	0.70
Juvenile Pacific cod	0.44	0.44	0.44	0.44
Adult Pacific cod	0.79	0.79	0.79	0.79
Juvenile sablefish	0.54	0.54	0.54	0.54
Adult sablefish	0.66	0.66	0.66	0.66
Juvenile lingcod	0.70	0.70	0.70	0.70
Adult lingcod	0.77	0.77	-	-
Shallowwater benthic fish	0.62	0.62	0.62	0.62
Skates	0.65	0.65	0.65	0.65
Large crabs	0.45	0.45	0.45	0.45
Small crabs	0.47	0.47	0.47	0.47
Commercial shrimp	0.35	0.35	0.35	0.35
Epifaunal invertebrates	0.20	0.20	0.20	0.20
Infaunal carnivorous invertebrates	0.23	0.23	0.23	0.23
Infaunal invertebrate detritivores	0.20	0.20	0.20	0.20
Carnivorous jellyfish	0.23	0.23	0.23	0.23
Euphausiids	0.25	0.25	0.25	0.25
Copepods	0.20	0.20	0.20	0.20
Macrophytes	0.20	0.25	0.25	0.25
Phytoplankton	0.20	0.23	0.23	0.23

Table A4b. Flow control in Newfoundland.

Functional Group	1500	1900	1985	1995
Walrus	0.53	0.53	0.48	0.53
Cetaceans	0.71	0.72	0.66	0.67
Grey seals	0.79	0.80	0.73	0.79
Harp Seals	0.73	0.73	0.68	0.74
Hooded Seals	0.80	0.80	0.80	0.80
Ducks	0.45	0.45	0.42	0.45
Piscivorous Birds	0.78	0.76	0.69	0.75
Planktivorous Birds	0.58	0.58	0.53	0.53
Cod (> 40 cm)	0.68	0.68	0.67	0.71
Cod (≤ 40 cm)	0.60	0.60	0.60	0.63
American plaice (> 35 cm)	0.56	0.56	0.56	0.55
American plaice (≤ 35 cm)	0.54	0.54	0.56	0.59
Greenland Halibut (> 65 cm)	0.79	0.79	0.75	0.77
Greenland Halibut (≤ 65 cm)	0.75	0.75	0.68	0.73
Yellowtail Flounders	0.48	0.48	0.44	0.48
Witch flounder	0.45	0.45	0.42	0.46
Winter flounder	0.47	0.47	0.43	0.45
Skates	0.75	0.75	0.68	0.73
Dogfish	0.70	0.70	0.63	0.67
Redfish	0.62	0.62	0.56	0.58
Transient Mackerel	0.66	0.66	0.60	0.64
Dem. BP Pisc. (>40 cm)	0.77	0.77	0.71	0.75
Dem. BP Pisc. (≤ 40 cm)	0.67	0.68	0.63	0.61
Demersal Feeders (> 30 cm)	0.54	0.54	0.49	0.51
Demersal Feeders (≤ 30 cm)	0.52	0.52	0.48	0.48
Small Demersals	0.47	0.47	0.44	0.47
Lumpfish	0.59	0.59	0.54	0.55
Greenland cod	0.67	0.71	0.64	0.69
Salmon	0.76	0.76	0.69	0.74
Capelin	0.51	0.51	0.47	0.49
Sandlance	0.50	0.50	0.46	0.48
Arctic cod	0.54	0.54	0.50	0.51
Herring	0.52	0.52	0.48	0.49
Transient Pelagics	0.70	0.71	0.65	0.68
Small Pelagics	0.55	0.55	0.51	0.50
Small Mesopelagics	0.54	0.54	0.50	0.50
Shortfin squid	0.69	0.69	0.64	0.69
Arctic Squid	0.52	0.52	0.48	0.48
Large Crabs (> 95 cm)	0.43	0.43	0.40	0.43
Small Crabs (≤ 95 cm)	0.47	0.47	0.43	0.46
Lobster	0.43	0.43	0.40	0.43
Shrimp	0.31	0.31	0.30	0.31
Echinoderms	0.20	0.20	0.20	0.20
Polychaetes	0.20	0.20	0.20	0.20
Bivalves	0.20	0.20	0.20	0.20
Other Benthic Invertebrates	0.20	0.20	0.20	0.20
Large Zooplankton	0.34	0.34	0.32	0.28
Small Zooplankton	0.20	0.20	0.20	0.20
Phytoplankton	0.30	0.30	0.30	0.30

Table A5a. Biomass/production ratios for BC *.

	2000	1950	1900	1750
Sea Otters	7.69	7.69	7.69	7.69
Mysticetae	50.00	50.00	50.00	50.00
Odontocetae	25.00	50.00	25.00	25.00
Seals, sea lions	16.67	16.67	16.67	16.67
Seabirds	10.00	10.00	10.00	10.00
Transient salmon	0.40	0.40	1.61	1.93
Coho salmon	0.36	0.36	0.94	0.86
Chinook salmon	0.46	0.46	2.75	2.73
Small squid	0.17	0.17	0.17	0.17
Squid	0.17	0.17	0.17	0.17
Ratfish	10.10	10.10	5.03	5.03
Dogfish	10.10	10.10	7.14	9.09
Juvenile pollock	0.94	0.94	4.35	4.35
Pollock	3.80	3.80	6.49	6.54
Forage fish	0.70	0.70	1.70	1.68
Eulachon	0.70	0.70	1.67	1.67
Juvenile herring	0.46	0.46	0.85	0.85
Adult herring	1.46	1.46	1.25	1.26
Juvenile POP	1.49	1.49	2.96	2.96
Adult POP	6.94	6.94	4.41	4.41
Inshore rockfish	5.26	5.26	5.49	5.49
Juvenile picivorous rockfish	3.83	3.83	3.83	3.83
Adult picivorous rockfish	27.03	27.03	27.03	27.03
Juvenile planktivorous rockfish	3.83	3.83	3.83	3.83
Adult planktivorous rockfish	14.71	14.71	14.71	14.71
Juvenile turbot	3.03	3.03	3.03	3.03
Adult turbot	4.55	4.55	4.55	4.55
Juvenile flatfish	0.52	0.52	2.62	2.62
Adult flatfish	1.05	1.05	3.89	3.89
Juvenile halibut	1.67	1.67	8.62	10.10
Adult halibut	2.50	2.50	11.90	14.93
Juvenile Pacific cod	0.51	0.51	3.88	3.88
Adult Pacific cod	0.76	0.76	5.75	5.75
Juvenile sablefish	1.67	1.67	3.66	3.66
Adult sablefish	3.62	3.62	5.43	5.46
Juvenile lingcod	0.83	0.83	2.57	2.57
Adult lingcod	1.25	1.25	3.33	3.82
Shallowwater benthic fish	0.67	0.67	3.76	3.76
Skates	3.23	3.23	6.67	6.67
Large crabs	0.67	0.67	0.67	0.67
Small crabs	0.29	0.29	0.29	0.29
Commercial shrimp	0.09	0.09	0.18	0.18
Epifaunal invertebrates	0.69	0.69	0.69	0.69
Infaunal carnivorous invertebrates	0.50	0.50	0.50	0.50
Infaunal invertebrate detritivores	0.74	0.74	0.77	0.77
Carnivorous jellyfish	0.06	0.06	0.06	0.06
Euphausiids	0.16	0.17	0.17	0.17
Copepods	0.04	0.04	0.04	0.04
Corals and sponges	100.00	100.00	100.00	100.00
Macrophytes	0.19	0.19	0.19	0.19
Phytoplankton	0.01	0.01	0.01	0.01

**Ecological objective maximizes B/P surrogate*

Table A5b. Biomass/production ratios for NFLD.

	1450	1900	1985	1995
Walrus	16.6	16.6	16.6	16.6
Cetaceans	20	10	10	10
Grey seals	16.6	16.6	16.6	16.6
Harp Seals	9.8	9.8	9.8	9.8
Hooded Seals	9.2	9.2	9.2	9.2
Ducks	4	4	4	4
Piscivorous Birds	4	4	4	4
Planktivorous Birds	4	4	4	4
Cod (> 40 cm)	4.6	10.4	2.4	3.4
Cod (≤ 40 cm)	4.8	4.2	0.6	0.6
American plaice (> 35 cm)	12	12	4.4	11.4
American plaice (≤ 35 cm)	8	8	1.6	2.4
Greenland Halibut (> 65 cm)	17	29.8	3.4	10.2
Greenland Halibut (≤ 65 cm)	13.2	39.8	1.2	2.6
Yellowtail Flounders	3.2	3.2	1.8	3.2
Witch flounder	4.2	4.2	1.8	2.8
Winter flounder	3.8	3.8	3.8	3.8
Skates	4.2	9	2.8	3.2
Dogfish	6.2	6.2	5.2	5.2
Redfish	8.8	8.8	2	6.8
Transient Mackerel	1.8	1.8	3.4	3.4
Demersal BP Piscivores (>40 cm)	10.2	10.2	1.6	4.8
Demersal BP Piscivores (≤ 40 cm)	6.8	6.8	6.8	6.8
Demersal Feeders (> 30 cm)	6.4	6.4	3.6	4.4
Demersal Feeders (≤ 30 cm)	4.4	4.4	4.4	4.4
Small Demersals	1.8	1.8	1.8	1.8
Lumpfish	8.8	8.8	8.8	8.6
Greenland cod	9.8	9.8	6	1.6
Salmon	3.6	3.6	1.6	1.6
Capelin	1.4	2	0.8	0.8
Sandlance	1	1	0.8	0.8
Arctic cod	1.8	1.8	2.4	1.8
Herring	2	2	1.8	1.8
Transient Pelagics	5.4	5.4	2.4	2.4
Small Pelagics	1.6	1.6	1.6	1.6
Small Mesopelagics	0.8	0.8	0.8	0.8
Shortfin squid	1.6	1.6	1.6	1.6
Arctic Squid	2	2	2	2
Large Crabs (> 95 cm)	2.6	2.6	2.6	2.6
Small Crabs (≤ 95 cm)	2.6	2.6	2.6	1.6
Lobster	2.6	5.2	2.6	2.6
Shrimp	0.6	0.6	0.6	0.6
Echinoderms	1.6	1.6	1.6	1.6
Polychaetes	0.4	0.4	0.4	0.4
Bivalves	1.8	1.8	1.8	1.8
Other Benthic Invertebrates	0.4	0.4	0.4	0.4
Large Zooplankton	0.2	0.2	0.2	0.2
Small Zooplankton	0.2	0.2	0.2	0.2

Appendix B
Policy Search Parameters

Table B1a. Lost Valley catch for BC *

	Groundfish Trawl	Shrimp Trawl	Shrimp Trap	Herring Seine	Halibut Longline	Salmon Freezer Troll	Salmon Wheel	Rockfish Live	Crab Trap	Clam Dredge	Aboriginal	Recreational
Transient salmon						2.5	2.5				2.5	
Coho salmon						2.5					2.5	2.5
Chinook salmon						2.5					2.5	2.5
Ratfish	0.25	0.25										
Dogfish	0.25	0.25				0.25						
Pollock	0.25											
Eulachon		2.5									2.5	
Juvenile herring				2.5								
Adult herring				2.5								
Adult POP	2.5											
Inshore rockfish	2.5				0.25	0.25		2.5				0.25
Adult picivorous rockfish	2.5					0.25						0.25
Adult planktivorous rockfish	2.5					0.25						
Juvenile turbot					0.25							
Adult turbot	0.25	0.25			2.5							
Juvenile flatfish					0.25							
Adult flatfish	2.5	0.5			0.25							
Juvenile halibut					2.5							0.25
Adult halibut					2.5						2.5	0.25
Adult Pacific cod	2.5				0.25							
Adult sablefish	0.25				0.25							
Adult lingcod	0.25				0.25			2.5				2.5
Shallow water benthic fish		0.25	0.25	0.25								
Skates	0.25	0.25			2.5							
Large crabs	0.25								2.5			
Small crabs									0.25			
Commercial shrimp		2.5	2.5									
Epifaunal invertebrates										2.5		

*Percentages indicate the fraction of the total group biomass caught in the first year of the policy exploration. The *Ecopath* description is available in Ainsworth *et al.* (2002). 2.5% of total biomass is caught for target species, 0.25% or 0.5% of total biomass is caught in retained bycatch.

Table B1b. Newfoundland Lost Valley Catch as a percentage of the biomass of each group *

Group Name	Bottom trawl	Shrimp trawl	Recreational	First Nations	Cod trap	Capelin	Longline	Redfish	Lumpfish trap	Snow crab traps	Inshore crab traps	Lobster traps	Salmon	Pole and line	Bivalves	Urchins
Walrus				0.25												
Cetaceans				0.01												
Grey Seals				0.25												
Harp Seals				0.25												
Hooded Seals				0.25												
Cod > 35cm	2.5		0.25		2.5		2.5	0.25								
Cod < 35 cm	0.25	2.5			0.25			0.25								
American plaice > 35cm	2.5						2.5	0.25								
American plaice < 35cm	0.25	2.5														
Greenland halibut > 40cm	2.5						2.5	0.25								
Greenland halibut < 40cm	0.25	2.5					0.25									
Yellowtail Flounder	2.5						2.5	0.25								
Witch flounder	2.5						2.5	0.25								
Skates	2.5	0.25					2.5									
Dogfish	2.5	0.25					2.5									
Redfish	2.5							2.5								
Transient mackerel			0.25													
L. D. Bentho-pelagic Pisc.	2.5						2.5	0.25								
S. D. Bentho-pelagic Pisc.	0.25	0.25					0.25									
L.Dem.Feeders	2.5						2.5									
S.Dem.Feeders	0.25	0.25					0.25									
O.S.Demersals	0.25	0.25														
Lumpfish	0.25	0.25							2.5							
Greenland cod					2.5											
Salmon			2.5										2.5			
Capelin						2.5										
Herring	0.25															
Transient Pelagics														2.5		
Small Pelagics	0.25	0.25	0.25													
Shortfin squid	0.25															
Large Crabs										2.5						
Small Crabs											2.5					
Lobster												2.5				
Shrimp	0.25	2.5														
Echinoderms																0.25
Bivalves															2.5	

*Percentages indicate the fraction of the total group biomass caught in the first year of the policy exploration. *Ecopath* description is available in Pitcher *et al.* (2002). 2.5% of total biomass is caught for target species, 0.25% or 0.25% of total biomass is caught in retained bycatch.

Table B2a. West coast discards. Percentages indicate the fraction of total biomass caught in the first year of the policy exploration. Major sources of bycatch are set at 1.25% of group biomass, minor bycatch is 0.25% or 0.025%.

Group Name	Groundfish Trawl	Shrimp Trawl	Salmon Freezer Troll	Clam Dredge
Seabirds			0.025	
Small crabs	1.25	1.25		0.25
Epifaunal invertebrates	1.25	1.25		0.25
Infaunal carnivorous invertebrates	1.25	1.25		0.25
Infaunal invertebrate detritivores	1.25	1.25		0.25
Corals and sponges	1.25	1.25		0.25

Table B3a. Jobs per catch value for northern BC.

Fleet	Jobs/catch value
Groundfish Trawl	0.4
Shrimp Trawl	0.6
Shrimp Trap	5
Herring Seine	4
Halibut Longline	1.3
Salmon Freezer Troll	2
Salmon Wheel	0.2
Rockfish Live	5
Crab Trap	5
Clam Dredge	5
Aboriginal*	-
Recreational	15

*Policy search did not include aboriginal fleet.

Table B2b. East coast discards. Percentages indicate the fraction of total biomass caught in the first year of the policy exploration. Major sources of bycatch are set at 1.25% of group biomass, minor bycatch is 0.25% or 0.025%.

Group Name	Bottom trawl	Shrimp trawl	Bivalves
Echinoderms	2.5	2.5	2.5
Polychaetes	2.5	2.5	2.5
Bivalves	2.5	2.5	
Other Benthic Invertebrates	2.5	2.5	2.5

Table B3b. Jobs per catch value for Newfoundland.

Gear	Jobs/catch value
Bottom trawl	0.4
Shrimp trawl	0.6
Recreational	15
First Nations	0.1
Cod trap	2
Capelin	0.4
Cod long-line	1.3
Redfish	0.6
Lumpfish trap	5
Offshore crab traps	1
Inshore crab traps	5
Lobster traps	5
Salmon	0.2
Pole and line	1
Bivalves (clams etc.)	10
Sea urchins	10

Table B4a. Value weight settings for fleets, years and policy objectives in northern BC. *Bold values indicate that mandated rebuilding was required to prevent computer crashes. ** Numbers in parentheses indicate the biomass goal of the policy search relative to the *Ecopath* baseline.

Fleet	Period	#	Objective	Policy Search Parameters					
				Ecological	Economic	Social	Mandated Rebuilding*	Variance (□^{2}) of mixed	MR protected groups**
Lost Valley	1750	1	Ecological	1	0	0	0		
		2	Economic	0	1	0	0		
		3	Social	0	0	1	0		
		4	Mixed objective	100	1	1	0.1	0.309	Juv/ad turbot (1)
		5	Portfolio Log Utility	Existence value = 0.1					
	1900	6	Ecological	1	0	0	5		Juv/ad turbot (1)
		7	Economic	0	1	0	0		
		8	Social	0	0	1	0		
		9	Mixed objective	100	1	1	5	0.192	Juv/ad turbot (1)
		10	Portfolio Log Utility	Existence value = 0.1					
	1950	11	Ecological	1	0	0	10		Skates (1)
		12	Economic	0	1	0	5		Juv/ad turbot (1)
		13	Social	0	0	1	0		
		14	Mixed objective	100	1	1	10	0.316	Skates (1)
		15	Portfolio Log Utility	Existence value = 1			Prediction variance = 0.005		
	2000	16	Ecological	1	0	0	7		
		17	Economic	0	1	0	5		Transient Salmon (1)
		18	Social	0	0	1	0		
		19	Mixed objective	100	1	1	10	0.679	Transient Salmon (1)
		20	Portfolio Log Utility	Existence value = 10			Prediction variance = 0.02		
No Recreat.	1750	21	Ecological	1	0	0	0		
		22	Economic	0	1	0	10		Transient Salmon (0.5)
		23	Social	0	0	1	0		
		24	Mixed objective	100	1	1	0.1	0.194	Juv/ad turbot (1)
		25	Portfolio Log Utility	Existence value = 1					
	1900	26	Ecological	1	0	0	5		Juv/ad turbot (1)
		27	Economic	0	1	0	10		Transient Salmon (0.5)
		28	Social	0	0	1	0		
		29	Mixed objective	100	1	1	50	0.304	Juv/ad turbot (1), Skates (1.5)
		30	Portfolio Log Utility	Existence value = 1					
	1950	31	Ecological	1	0	0	0		
		32	Economic	0	1	0	10		Juv/ad turbot (1)
		33	Social	0	1	0	10		Skates (1)
		34	Mixed objective	100	1	1	5	0.268	Skates (1)
		35	Portfolio Log Utility	Existence value = 1					
	2000	36	Ecological	1	0	0	2		Skates (1)
		37	Economic	0	1	0	10		Transient Salmon (1)
		38	Social	0	0	1	1		Skates (1)
		39	Mixed objective	100	1	1	20	0.247	Skates (1)
		40	Portfolio Log Utility	Existence value = 1			Prediction variance = 0.003		
No Trawlers	1750	41	Ecological	1	0	0	0		
		42	Economic	0	1	0	5		Transient Salmon (1)
		43	Social	0	0	1	0		
		44	Mixed objective	100	1	1	0	0.218	
		45	Portfolio Log Utility	Existence value = 0.1					
	1900	46	Ecological	1	0	0	1		Transient Salmon (1)
		47	Economic	0	1	0	5		Skates (1)
		48	Social	0	0	1	0		
		49	Mixed objective	100	1	1	2	0.097	Juv/ad turbot (1)
		50	Portfolio Log Utility	Existence value = 0.1					
	1950	51	Ecological	1	0	0	0		
		52	Economic	0	1	0	0		
		53	Social	0	0	1	0		
		54	Mixed objective	100	1	1	10	0.258	
		55	Portfolio Log Utility	Existence value = 0.1					
	2000	56	Ecological	1	0	0	0		
		57	Economic	0	1	0	0		
		58	Social	0	0	1	0		
		59	Mixed objective	100	1	1	0	0.278	
		60	Portfolio Log Utility	Existence value = 0.1					

Table B4b. Value weight settings for fleets, years and policy objectives in Newfoundland. *Mandated rebuilding was not used with the Newfoundland models; some species went extinct. **Group biomass increased or decreased more than twice. Increased indicated by + and decreased indicated by - ***Unstable indicates that ecosystem never stabilized over the 50 year time span.

Fleet	Period	#	Objective	Policy Search Parameters					
				Ecological	Economic	Social	Mandated Rebuilding*	Variance (σ^2) of mixed	Large change in group biomass**
Lost Valley	1450	1	Ecological	1	0	0	0		Salmon (-) G. halibut (+)
		2	Economic	0	1	0	0		Many (+), many (-)
		3	Social	0	0	1	0		Unstable***
		4	Mixed objective	100	1	0.5	0	0.646	Skate, sf squid (-) halibut (+)
		5	Portfolio Log Utility	Existence value = 0.05					
	1900	6	Ecological	1	0	0	0		Salmon (+) short fin squid (-)
		7	Economic	0	1	0	0		Large and small crabs (-)
		8	Social	0	0	1	0		Crabs, transient pelagics (-)
		9	Mixed objective	100	0.1	0.1	0	0.195	Salmon (+) short fin squid (-)
		10	Portfolio Log Utility	Existence value = 0.05					
	1986	11	Ecological	1	0	0	0		
		12	Economic	0	1	0	0		Salmon (-) short fin squid (+)
		13	Social	0	0	1	0		Unstable***
		14	Mixed objective	100	0.1	0.1	0	0.181	
		15	Portfolio Log Utility	Existence value = 0.1					
	1996	16	Ecological	1	0	0	0		
		17	Economic	0	1	0	0		Salmon (-)
		18	Social	0	0	1	0		Salmon (-)
		19	Mixed objective	100	0.1	0.1	0	0.304	
		20	Portfolio Log Utility	Existence value = 0.1					
No Recreational	1450	21	Ecological	1	0	0	0		Skate, sf squid (-) halibut (+)
		22	Economic	0	1	0	0		Many (+), many (-)
		23	Social	0	0	1	0		Unstable***
		24	Mixed objective	100	1	0.1	0	0.095	Skate, sf squid (-) halibut (+)
		25	Portfolio Log Utility	Existence value = 0.1					
	1900	26	Ecological	1	0	0	0		Salmon (+) short fin squid (-)
		27	Economic	0	1	0	0		Large and small crabs (-)
		28	Social	0	0	1	0		Large and small crabs (-)
		29	Mixed objective	100	0.1	0.1	0	0.196	Salmon (+) short fin squid (-)
		30	Portfolio Log Utility	Existence value = 0.01					
	1986	31	Ecological	1	0	0	0		
		32	Economic	0	1	0	0		Short fin squid (+), many (-)
		33	Social	0	1	0	0		Unstable***
		34	Mixed objective	100	0.1	0.1	0	0.177	
		35	Portfolio Log Utility	Existence value = 0.05					
	1996	36	Ecological	1	0	0	0		Salmon (-)
		37	Economic	0	1	0	0		Many (+), many (-)
		38	Social	0	0	1	0		Many (+), many (-)
		39	Mixed objective	100	0.1	0.1	0	0.191	Salmon (-)
		40	Portfolio Log Utility	Existence value = 0.05					
No Trawlers	1450	41	Ecological	1	0	0	0		Salmon (-) G. halibut (+)
		42	Economic	0	1	0	0		Many (+), many (-)
		43	Social	0	0	1	0		Unstable***
		44	Mixed objective	100	1	0.1	0	0.335	Salmon (-), G. halibut (+)
		45	Portfolio Log Utility	Existence value = 0.1					
	1900	46	Ecological	1	0	0	0		Salmon (+) short fin squid (-)
		47	Economic	0	1	0	0		Large and small crabs (-)
		48	Social	0	0	1	0		Large and small crabs (-)
		49	Mixed objective	100	0.1	0.1	0	0.145	Salmon (+) short fin squid (-)
		50	Portfolio Log Utility	Existence value = 0.05					
	1986	51	Ecological	1	0	0	0		
		52	Economic	0	1	0	0		Short fin squid (+), many (-)
		53	Social	0	0	1	0		Unstable***
		54	Mixed objective	100	0.1	0.1	0	0.039	
		55	Portfolio Log Utility	Existence value = 0.1					
	1996	56	Ecological	1	0	0	0		
		57	Economic	0	1	0	0		Many (+), many (-)
		58	Social	0	0	1	0		Salmon (-)
		59	Mixed objective	100	0.1	0.1	0	0.251	
		60	Portfolio Log Utility	Existence value = 0.1					

ENVIRONMENTAL ARCHAEOLOGY: PRINCIPLES AND CASE STUDIES

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ABSTRACT

Archaeological data are most commonly applied towards understanding past human activities. However, these data can include environmental information such as animal and plant remains which offer insight into past environmental history. This paper outlines general introductory principles of environmental applications in archaeology, including the character of archaeological data, the preservation of environmental remains, and problems of interpretation arising from the "cultural filter" through which these remains necessarily have passed. We conclude by noting problems and prospects in environmental archaeology, leading to two case studies which demonstrate the value and potential of archaeological analyses to the reconstruction of past ecosystems. The first case study explores the period of European contact in Gwaii Haanas (Queen Charlotte Islands, British Columbia), a time characterized by rapid and substantial environmental changes. In particular, archaeological evidence is described that relates to the extirpation of the sea otter during the maritime fur trade and the resulting impact on ecologically related species such as abalone, sea urchin, and kelp-dependent fish. The second case study examines prehistoric fish use in the Aleutian Islands. Specifically, size reconstruction of Pacific cod specimens recovered from Aleut archaeological sites shows the harvesting of fish that exceed the size of those commonly encountered by modern commercial fisheries. Together, these case studies demonstrate that archaeological analysis can provide a picture of the past environment that is not readily available through other sources of data.

INTRODUCTION TO ENVIRONMENTAL ARCHAEOLOGY

Archaeological faunal remains provide a useful, if imperfect, record of the past environment. The majority of archaeological faunal remains enter a site's deposits through direct human action, though a portion of such remains may result from the activities of scavengers and other animals, or may enter a site as a secondary by-product of the targeted resources (Erlandson and Moss 2001, Lyman 2002, Moss and Erlandson 2002, Orchard 2001b). As humans tend to harvest resources from a wide variety of niches, these deposits often provide a broad view of the environments available to a site's inhabitants. The anthropogenic nature of archaeological deposits, however, means that faunal remains from archaeological sites can be seen as a culturally filtered sample of the environment from which the site residents obtained their resources. Despite this bias, however, the abundance, accessibility, visibility and broad scope of archaeological faunal deposits make them a particularly useful environmental record, especially when compared to typically rare and limited natural faunal deposits. This is particularly true for marine mammals and fish, which have vanishingly small probabilities of ending up in accessible paleontological deposits. The value of archaeological sites as sources of environmental history has been recognized in a number of recent projects and texts (Amorosi *et al.* 1997; Cannon 1995; Grayson 1984, 2001; Orchard 2001b; Reitz *et al.* 1996; Reitz and Wing 1999; Sandweiss 1996). Of particular interest and relevance to the case studies outlined below, are papers that discuss and exemplify the role that zooarchaeological analysis can play in wildlife management (Amorosi *et al.* 1996; Lyman 1996). The following are some simple analytical techniques or domains which have promise for answering questions about paleo-ecology.

Addressing bio-diversity is most straightforward through the creation of a species list from identified remains. Such lists from shell-bearing archaeological sites – which typically offer the best preservation of bone – can run into hundreds of taxa. From such lists, local ecological niches can be identified and past biodiversity compared to the present. Of particular interest are indicator or keystone species with very narrow niches or specific environmental tolerances, or whose presence or absence is a strong predictor of other species. Sea Otter probably fills such a role in near-coastal marine ecologies, as discussed below (and see Pitcher 2004, this volume).

Another area of inquiry includes changes in faunal 'demographics'. Some species will have undergone historic change in population structure or growth and development as a result of changing human or animal predation patterns or intensity. For example, species which are under heavy predation may exhibit a flattened population structure, with fewer mature individuals and more immature individuals than

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Orchard, T.J. and Mackie, Q. (2004) Environmental Archaeology: Principles and Case Studies. Pages 64–73 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.

might be expected. Potentially, population information can be derived from:

- Shellfish, especially bivalves, through the study of annual growth rings, size, and growth rates. Such studies have been commonly undertaken in archaeology, sometimes showing a decline in average size with apparent increased predation (Ham and Irvine 1975; Wessen 1988, Claassen 1998). Shellfish incorporate seasonal and annual growth rings, and relatively complete shells, especially bivalve shells, can be thin-sectioned and these rings examined. This has the potential to illuminate both the cultural and natural history of an area by tracking changing predation pressures, water temperatures, and so forth.

- Fish, through the study of size and age structure of the population. The main sources of data would be *otoliths* (ear bones) whose rings track age and growth rates, and *scales*, which can preserve surprisingly well in archaeological sites. Rockfish otoliths are the largest and most robust amongst likely fish remains to be found (Wigen pers. comm.). Fish vertebrae can also be aged using x-ray densitometry. Key indicator fish skeletal elements can be correlated via regression equations to length and body mass. For example, rockfish size can be accurately estimated using the diameter of the atlas (Wigen pers. comm.); Pacific cod using dimensions of the quadrate and other mouth elements (see below).

- Mammals and birds: as they have different reproductive strategies than the above, may be predated upon differently, and as their remains may be rarer in absolute terms, it is more difficult to be confident in one's ability to draw conclusions. An interesting exemplary species is the sea otter, whose population is known to have declined to extirpation by ca. 1830. Knowing the temporal parameters and outcome of this increased predation it would be of use to see if this was archaeological visible through changing age structure of recovered remains (see below). Also, juvenile mammal remains can be aged, teeth can reveal information about dietary stress, and stable-isotope analysis can show changes in long-term diet. For example, preliminary, unpublished results from Haida Gwaii suggest that prior to ca. 10,000 years ago, black bears consumed little or no marine protein, in stark contrast to their present day habits.

Archaeological data can provide a very alluring source of Palaeonvironmental data for other historical sciences, but the use of these data should be well-informed. The following discusses some interpretive constraints in environmental archaeology, with emphasis on BC coastal processes.

Cultural choice: the faunal record at an archaeological site is a product of culturally-mediated choice. It is not a microcosm of the natural ecology, but a reflection of the human niche in that ecology. In spite of this, it is important to remember that not all the taxa were taken directly by humans: some came in as incidentals, stomach contents, etc. Furthermore, humans can only select from what is actually available, although trade and exchange can widen their catchments considerably. This “cultural filter” must always be accounted for. Hence, for example, a finding that 70% of the fish bones in a particular site are from herring tells us more about human taste in food than the absolute or relative abundance of herring in the environment.

Differential preservation of environmental remains: some classes of evidence, such as large land mammal bones and shells preserve relatively well, while other remains, such as delicate fish elements, crustaceans, small land mammal and bird bones preserve less well. Some environmental information, such as terrestrial plants, marine plants, and fungi only preserve in special situations. Further, preservation may be spatially heterogeneous across the site (Stein 1996). Therefore, the diversity and proportions of environmental remains in the present do not necessarily bear a 1:1 correspondence with the material when deposited. Furthermore, most of these taphonomic processes unfold over time, meaning that the actual remains found in an archaeological site is a complex function of time, inherent durability, soil chemistry, and site sampling strategies. All of these factors need to be accounted for when attempting interpretation of environmental remains, whether for cultural or natural historical ends.

A major interpretive consideration is the amount of material that must be excavated to produce a reliable sample size and representative taxonomic diversity. At Crescent Beach, a shell midden near Vancouver, the relationship between diversity of fish taxa and size of sample (expressed as number of identified skeletal elements, NISP) is clear. After ca. 750 to 1,000 elements of all fish taxa are recovered, there is virtually no increase in taxonomic diversity (Driver 1993: 93). Achieving these sample sizes of archaeological fish remains is fairly common. However, fish tend to be among the more numerous faunal categories, and if similar numerical relationships hold for birds and mammals, then it may become an issue whether taxonomic diversity is fully represented at any given site.

The case studies outlined below exemplify some of the methods which can be applied to the gathering and analysis of environmental data from archaeological sites, as well as the types of results that may be obtained. There are many more methods that could be, or have been, applied to these cases (see, for example, Dincauze 2000), and the results presented are in some cases preliminary. Together, the case studies demonstrate some of the problems and prospects of an archaeological contribution to marine environmental reconstruction and management.

CASE STUDY 1: GWAII HAANAS

The period of European Contact in Gwaii Haanas National Park Reserve/Haida Heritage Site, Haida Gwaii (Queen Charlotte Islands, British Columbia), was one of rapid and dramatic change for the Haida (Acheson 1998; Duff and Kew 1958). Similarly, the current ecology and environment of Gwaii Haanas has been profoundly influenced by historic-period environmental changes, beginning with the first European contact in 1774 (Blackman 1990). In particular, activities related to the maritime fur trade such as the rapid extirpation of sea otter populations, had a dramatic impact on the local environment. The removal of sea otters, for example, is known to have allowed the spread of sea urchins, which in turn limits the growth of kelp forests and their associated ecosystems (Bodkin 1988; Breen *et al.* 1982; Duggins 1981; Estes and Palmisano 1974; Pace 1981). Similar changes are known to have resulted from the introduction of non-indigenous species, such as deer (Vourc'h *et al.* 2001), rats (Bertram and Nagorsen 1995; Drever 1997; Taylor *et al.* 2000), and raccoons (Hartman and Eastman 1999); and from modern industrial harvesting of timber and other resources (Forest 2001; Grzybowski and Slocombe 1988). Furthermore, European contact introduced diseases and changed settlement patterns which lead to mass human depopulation of Gwaii Haanas and the sequential amalgamation of small villages of 2 to 3 houses into larger villages (Acheson 1998). By 1890, all the surviving Haida had settled in the villages of Skidegate and Masset on Graham Island to the north of Gwaii Haanas (Blackman 1990), and thus the Gwaii Haanas human ecology had also been greatly altered.

Despite the importance of this period in Haida culture history, relatively little work has attempted to document or address these issues. Rather, most archaeological work in Gwaii Haanas has focussed on early Holocene occupations or on general site inventory (Fedje *et al.* 1996a,b, 2001; Fedje and Christensen 1999; Hobler 1978), with contact-period archaeology limited to excavations at only a very few sites (Abbott and Keen 1993; Acheson 1998; Duff and Kew 1958; MacDonald and Cybulski 1973). Of greatest relevance to the current case study is a project carried out by Acheson (1998), which revealed the wealth of environmental data available from sites dating to the last 2,000 years, recovering remains of 165 separate faunal taxa, representing a wide range ecological niches, from small scale excavation at 18 archaeological sites. Acheson's work, however, was not intended to address issues of environmental reconstruction, and only three of his excavated sites included historic period deposits (Acheson 1998). Although scholars in other disciplines have examined the Gwaii Haanas ecology from a current perspective while acknowledging historic changes (eg. Forest 2001; Grzybowski and Slocombe 1988), no one has specifically used archaeological data to examine the pre-contact to early contact period environment of the region.

Thus, though it is possible to speculate about many of the factors that are likely to have caused environmental changes in Gwaii Haanas since the time of first European contact, the pre-contact environment itself is largely unknown. Examination of environmental data from archaeological sites dating to the late pre-contact/early contact periods provides a unique window into this period of environmental change. Aside from providing a better understanding of the context in which the Haida people lived prior to European contact, knowledge of the "natural" pre-contact environment is a useful tool for the management of the relatively recently established Gwaii Haanas National Park Reserve/Haida Heritage Site. Although Parks Canada's mandate includes the environmental management of the region, the question remains as to which environment to manage, the pre-contact environment prior to the impact of European activities, the current environment, or that of some intervening period. Greater knowledge of a pre-contact environmental via archaeological environmental data would contribute to such management issues. In addition, demonstrating the inherent role of Haida food harvesting in the long-term ecological structure of the Gwaii Haanas region may provide evidence for aboriginal use-rights within the park reserve.

In order to investigate the potential for environmental archaeological work, a pilot project was conducted in June of 2000. As

Table 1. Number of Taxa Recovered Per Site (from Mackie *et al.* 2001). Totals are for all three sites, therefore columns do not add up.

Site	Vertebrate Taxa	Invertebrate taxa	Total[1]
1134T (Protected)	14	21	35
923T (Semi-Protected)	10	10	20
740T (Exposed)	23	24	47
Totals[1]	31	36	67

indicated above, much of the recent archaeological work in Gwaii Haanas has consisted of an extensive program of site survey, the results of which have been compiled in a Parks Canada database. This database contains information on the locations of all the known sites in Gwaii Haanas, the types of deposits found at each site, the dates of the sites when known, and the artifacts found or recovered at each site, providing a basis for the identification of sites with high potential for containing the information that we wished to recover. Specifically, we were interested in examining sites that: were occupied during the late pre-contact to early contact transition, and thus had dates or artifacts that indicated this period; contained shell midden deposits and thus had a high potential for the preservation of environmental remains; each represented a different set of environmental conditions in the form of exposed, protected and intermediate locations. Thus, the study sites (Figure 1) were selected from the database prior to the beginning of our field season.

Prior to excavation, each site was examined and tested via surface exposures, deposits in windfalls, cutbanks and other natural exposures, and through probe and auger testing. Such testing served primarily to verify the presence of preserved environmental remains in the form of shell midden deposits. Based on this testing one site, 1221T on the East coast of Lyell Island, was eliminated from our sample due to inadequate shell midden deposits. This site was replaced with site 740T on East Copper Island, another exposed site. Soil probes and augers were also used to aid in the placement of excavation units. Such subsurface sampling techniques have been shown to provide a reasonably good picture of the distribution of subsurface deposits (Stein 1986; Casteel 1970). Auger samples were also collected in some cases, and may be used as a supplemental source of environmental data (see Cannon 2000; Casteel 1970). Excavation units were placed judgmentally based on the results of soil probing and augering, with 1m by 1m units excavated in 10 cm arbitrary levels. To facilitate the recovery of environmental data, all material was water-screened through 1/8 inch mesh, with all bone, a representative sample of shell, and any other environmental remains, such as floral remains and fish scales, collected. In addition, column samples were collected from one wall of each unit following excavation, as column samples have been shown to provide a representative sample of environmental remains from an excavation unit (Casteel 1970, 1976a). All artifactual material was also collected, as were carbon samples for dating purposes when available, and each site was mapped with a total station.

The final analysis of materials from this pilot project is incomplete, and will form part of the ongoing Gwaii Haanas Environmental Archaeological Project being conducted as a component of the doctoral research program of the senior author. Nevertheless, preliminary results suggest that faunal remains from small-scale investigations can provide a picture of the past environmental characteristics of a site's local region, and can map regional environmental differences between sites in different ecological niches (see Table 1) (Mackie *et al.* 2001). This is

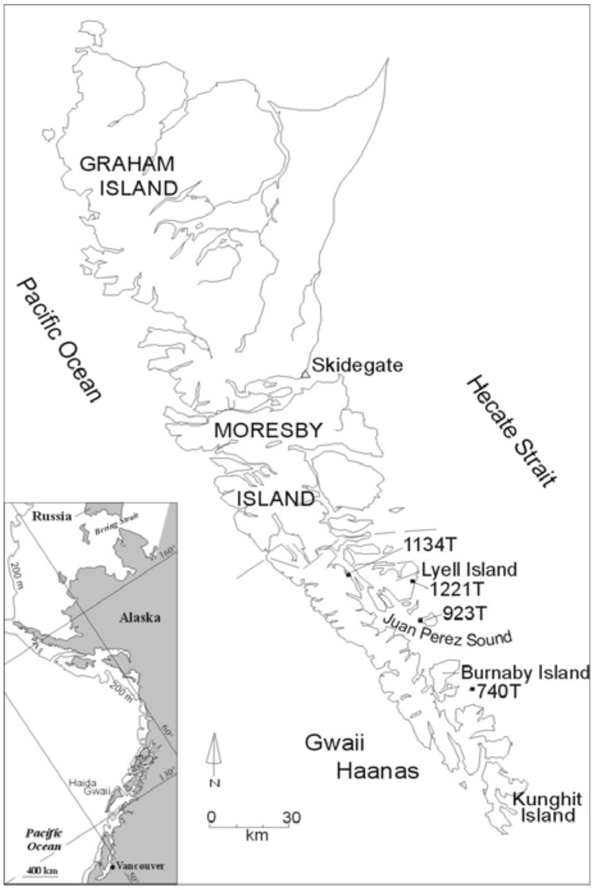

Figure 1. Map of Haida Gwaii showing location of study sites (Adapted from Fedje et al. 1996a).

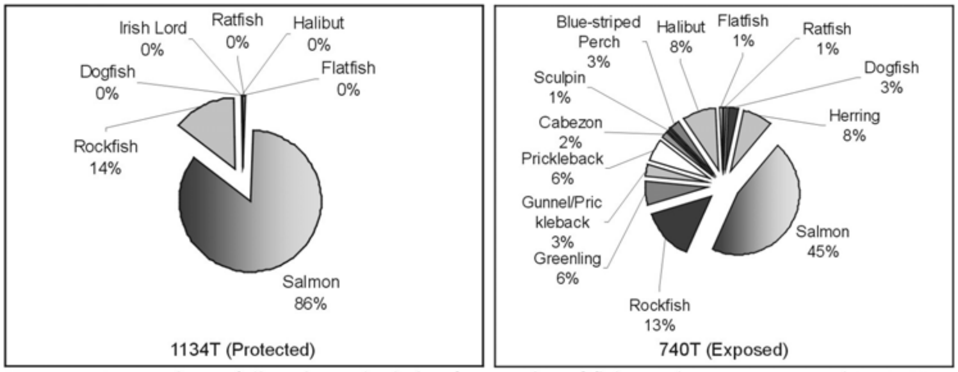

Figure 2. Comparison of diversity and relative frequencies of fish taxa between protected (1134T) and exposed (740T) locations.

particularly evident in the differences in the diversity of fish taxa between the protected site (1134T) and the exposed site (740T) as illustrated in Figure 2. Unsurprisingly, the protected site is dominated by salmon remains, contains the majority of the terrestrial-based avifauna, and was the only site to contain terrestrial mammal (Black bear). In contrast, the exposed site contained the greatest diversity of taxa, including a wide variety of fish (13 taxa), numerous remains of marine birds, and the greatest quantity of sea otter remains. Similarly, California mussel comprises the majority of invertebrate remains from the exposed and semi-exposed sites, whereas the protected site contains primarily Butter clam, Littleneck clam and small mussel (probably edible Mussel: *Mytilus trossulus*) (Mackie *et al.* 2001). Also of considerable interest is the small but intriguing correlation between the presence of sea otter in the assemblages and the presence of related taxa such as sea urchins, abalone, and kelp-dependent fish. As seen in Table 2, a strong presence of sea otter remains is loosely correlated with a near absence of abalone and sea urchin and an abundance of kelp dependent fish at the exposed (740T) and semi-exposed (923T) sites, while the opposite pattern is evident at the protected site (1134T). The well documented relationship between kelp and sea urchin grazing (Duggins 1981; Pace 1981) provides an ecological link between sea otter predation on sea urchins and the presence of nearshore, kelp dependent communities of fish. It is important to note, as well, that the low density and resulting low weight of sea urchin shell yields low proportions for sea urchin when compared to other invertebrate remains from each assemblage. However, the difference in proportion between 1134T (0.82) and the other two sites, 740T (0.03) and 923T (0.01), is relatively quite significant. Slightly differing dates at these sites (Mackie *et al.* 2001; Orchard 2001a) suggests that this pattern may map the shift from a pre-fur trade to a post-fur trade environment.

In addition to these interesting faunal results, radiocarbon dates from the sites and the recovery of contact-period artifacts (Mackie *et al.* 2001; Orchard 2001a) supports the occupation of the selected sites during the targeted time period, thus providing support for the utilized methodology. This is further evidenced by the absence, in the recovered faunal assemblages, of any introduced species, confirming that the recovered assemblages date prior to the major environmental changes discussed above. Though patterns in the data are clearly present, the small sample size and the potentially conflicting effects of varying exposure and varying temporal period may bias these results. An increased sample size resulting from ongoing work should clarify this issue. Generally, then, the pilot project demonstrated the potential of small-scale archaeological excavation to contribute to

Table 2. Sea Otter, Sea Urchin, and Ecologically Related Taxa (Derived from Mackie *et al.* 2001).

Taxon	740T (Exposed) 490±40 to 390±50[1]	923T (Semi-Protected) 150±50	1134T (Protected) 430±70 to 60±60
Sea Otter (% mammal by NISP)	57.1	57.1	0
Sea Urchin[2] (% invert. by weight)	0.03	0.01	0.82
Abalone[3] (% invert. by weight)	0	0.59	0
Nearshore/Kelp Forest Fish[4] (% fish by NISP)	21.2	35.7	14.1
Nearshore/Kelp Forest Fish (# identified taxa)	3	1	1

[1] Radiocarbon age ranges include marine reservoir corrected shell dates.

[2] Sea urchin is one of the primary food sources of sea otters (Estes and Palmisano 1974; Estes *et al.* 1978; Breen *et al.* 1982).

[3] Abalone density has also been inversely correlated with sea otters (Cooper *et al.* 1977).

4A variety of fish taxa are dependent upon or ecologically related to kelp forests, and are thus tied into the sea otter ecological web. In the Gwaii Haanas assemblages such fish include greenling (Estes and Palmisano 1974), rockfish (Bodkin 1988), and cabezon (Bodkin 1988).

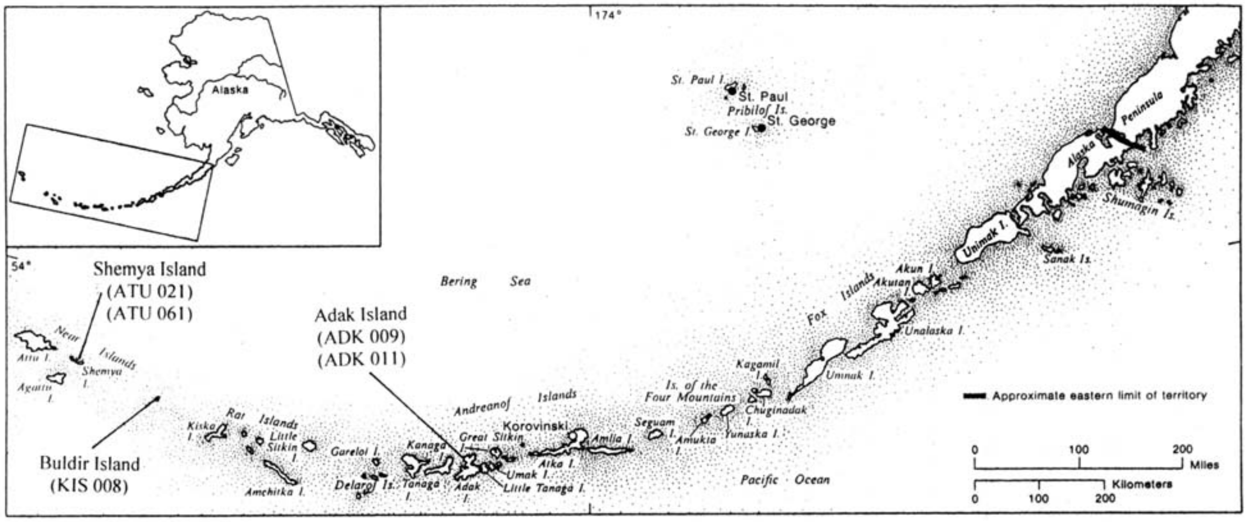

Figure 3. Map of Aleutian Islands study area with site locations (modified from Lantis 1984).

environmental reconstruction during the targeted late pre-contact to early contact time period. The "cultural filter" through which these data have passed is important, but the underlying ecological relationships show through. This confirms the availability of a wealth of environmental data in sites that are known, through the presence of early European trade goods, ethnohistoric records, and radiocarbon dates, to have been occupied through the early contact period (Mackie *et al.* 2001; Orchard 2001a).

CASE STUDY 2: ALEUTIAN ISLANDS PACIFIC COD

The reconstruction of the live size of animals represented by archaeological remains can provide useful information for both culture historical and environmental reconstructions. Our second case study examines the potential of such an approach in the context of environmental reconstruction through the synthesis of a project which was aimed at examining fish size in prehistoric Aleut sites as related to Aleut subsistence and to ecological change (Orchard 2001b). The Aleutian islands of southwest Alaska form a particularly interesting illustration of the potential of environmental archaeology, as they represent a relatively unique environmental context. It is this unique setting and the isolation of the archipelago that makes it particularly useful as a "cultural laboratory" (McCartney 1975: 288; cf. Black 1981; Corbett *et al.* 1997a; McCartney and Veltre 1999; Yesner and Aigner 1976). The project outlined here, completed as the M.A. thesis of the senior author (Orchard 2001b), involved the analysis of faunal assemblages from 5 sites in the central and western Aleutian archipelago (see Figure 3). This includes two sites on Shemya Island (ATU-021 and ATU-061), one site on Buldir Island (KIS-008), and two sites on Adak Island (ADK-009 and ADK-011). For the most part, the results of the excavations at these sites, all conducted by members of the Western Aleutian Archaeological and Paleobiological Project, remain unpublished. The exception is site KIS-008 on Buldir Island, which has generated several publications (Corbett *et al.* 1997b; Lefèvre *et al.* 1997; Bouchet *et al.* 1999), as well as a single publication from site ADK-011 on Adak Island (Bouchet *et al.* 2001).

Regression analysis provides a technique for the statistical comparison of the live size of fish, either length or weight, to the size of skeletal elements. This technique has been widely applied to fish taxa and has demonstrated the strong correlation that exists between fish size and skeletal element size (Casteel 1974, 1976b; Crockford 1997; Desse and Desse-Berset 1996; Enghoff 1983; Leach *et al.* 1996; Owen and Merrick 1994; Rojo 1986; Smith 1995). The case study involved the use of regression analysis to estimate the size (length and weight) of fish specimens from six of the most prevalent taxa encountered in the archaeological samples under consideration. The analysed taxa included Atka mackerel (*Pleurogrammus monopterygius*), greenling (*Hexagrammos* sp.), Irish Lord (*Hemilepidotus* sp.), Pacific cod (*Gadus macrocephalus*), rockfish (*Sebastes* sp.), and walleye pollock (*Theragra chalcogramma*). For each taxon comparative specimens of known live length and weight were used to generate regression formulae that compare these size measurements to measurements of a selection of skeletal elements (Orchard 2001b). These formulae, which produced strong correlations

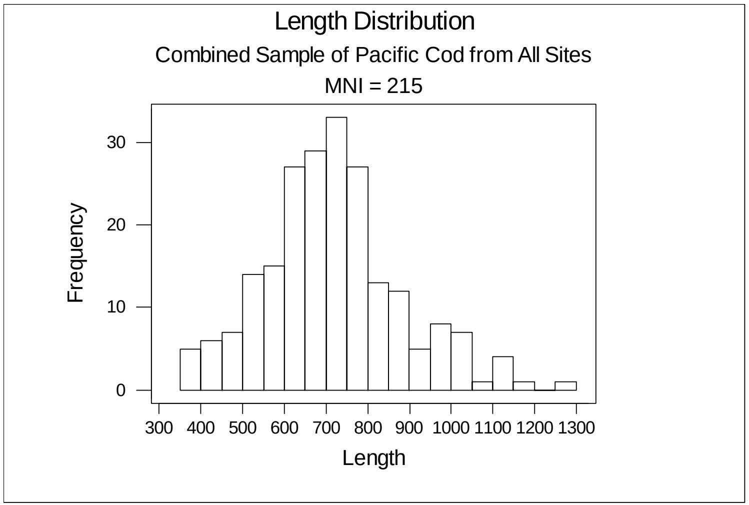

Figure 4. Size distribution of archaeological Pacific cod individuals from all five sites in the Aleutian Islands study area.

with r^2 values generally greater than 0.90 (Orchard 2001b), were then used to generate size estimates from the same measurements of archaeological skeletal specimens.

The estimated sizes of Pacific cod are particularly noteworthy in the context of discussions of environmental reconstruction and fisheries management. Archaeological Pacific cod specimens ranged up to and beyond the size ranges commonly encountered by modern commercial fisheries (see Figure 4). Of the total MNI[1] of 215 Pacific cod, 27 exceed 90 cm in length and 14 exceed 100cm in length. In comparison, published maximum sizes of Pacific cod range from 1 meter (Hart 1973) to 118 centimeters (Vinnikov 1996). Reported size ranges of commercial catches include 7 to 110 cm from the eastern Bering sea (Bakkala 1984), and 27 to 97 cm from Canadian catches (Foucher 1987). It is also telling that the largest specimen in the University of Victoria comparative collection, which was derived largely from modern commercial fisheries specimens, is only 88cm in length. In addition to the general size of Pacific cod specimens, there is some indication from the archaeological remains of a decrease in the size of Pacific cod over time (Table 3). Though the mean lengths show no consistent temporal trend across assemblages, the maximum lengths show a fairly consistent decrease over time (also see Orchard 1998). However, when the mean lengths and the proportion of individuals larger than 100cm in length are considered, site KIS 008 appears to stand out from the general trend (Table 3). Both the generally large size of Pacific cod from Aleutian sites and the apparent temporal trend provide insight into the structure of past populations of Pacific cod in the region. Generally, archaeological fish size profiles, such as those for Pacific cod presented in figures 4 and 5, may provide insight into ancient fish population structures, and when combined with established dates for the archaeological deposits, can reveal long term trends and variation in commercially important stocks. In a consideration of similar archaeological data for Atlantic cod, Amorosi and colleagues suggest that "zooarchaeology . . . would appear to have an important role in lengthening the observational series of environmental managers, perhaps warning of critical threshold discontinuities before the resource crash (rather than after, as in the case of the Atlantic cod)" (1996: 151). Thus, the cod length data presented here may have some utility in the management of the Pacific cod fishery. This is further evidenced by the utilisation of aspects of this methodology in the assessment of Steller sea lion prey consumption as it relates to North Pacific commercial fisheries (Zeppelin *et al.* 2001).

CONCLUSIONS

The two case studies presented above are unified in their use of archaeological faunal assemblages to help answer questions about past environmental conditions and changes. The first case study demonstrates that small-scale regional archaeological testing can provide faunal samples that reflect local ecological variation, and thus can be helpful in the reconstruction of local environmental histories. In addition, this case demonstrates that predicted changes in local ecology as a result of sea otter extirpation are visible in archaeological faunal samples. The

Table 3. Temporal patterns in Aleutian Islands Pacific cod (from Orchard 2001b).

Site	Radiocarbon Dates	Mean Length (mm)	Max. Length (mm)	Proportion > 100cm (%)
ATU 061	2570 ± 140 to 3096 ± 155	687	1250	10.00
ATU 021	1700 ± 70 to 1980 ± 60	746	1198	9.38
ADK 009	1040 ± 70 to 1240 ± 90	726	1122	4.62
ADK 011	180 ± 60 to 440 ± 40 (<2490 ± 50)	704	1048	1.96
KIS 008	220 ± 60 to 390 ± 80	807	1073	14.29

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[1] Note that MNI values were determined using a combination of the traditional MNI approach (White 1953) and the additional data available from regression-estimated lengths (see Orchard n.d.).

second case study demonstrates that the detailed reconstruction of fish size from archaeological faunal assemblages can provide data relevant to reconstructing the history of commercially important fish species, data which may play a role in current management plans for those species. Generally, these case studies exemplify the ability of archaeological data to make a useful contribution to the reconstruction of past environments and to the documentation of environmental changes.

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HOW TRADITIONAL KNOWLEDGE CAN CONTRIBUTE TO ENVIRONMENTAL RESEARCH AND RESOURCE MANAGEMENT

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Over the last three years I, along with my colleague Dr James Kari, have worked with First Nations in Alaska documenting their traditional knowledge of salmon. The objectives of this research are to provide fisheries biologists with information that could be useful in resource management and improve communications between First Nations and biologists. One of the problems is that within the scientific and management communities there is considerable uncertainty as to how traditional knowledge can contribute to scientific research. In this paper I outline four ways that traditional knowledge can contribute to environmental research and resource management. These are: 1) Traditional knowledge has a chronological depth which far surpasses written historical sources; 2) Traditional knowledge includes observations of the environment that are usually far more detailed than those collected by scientists; 3) Traditional management systems are community based; and 4) Traditional knowledge stems from a belief system that is ecological in nature.

Traditional knowledge can be divided into three analytical components: knowledge, practice and belief (Berkes 1998:13-14). The knowledge base includes such basic information as species identification, taxonomies, species behavior and distribution, and life histories. This knowledge has two significant attributes: it has considerable time depth and it is often very detailed. Collected over generations, traditional knowledge provides information that is not available anywhere else. The earliest written records relating to western Canada and Alaska go back to the 18th century and are often limited in time and space. As a result scientists today have short chronologies on which to build predictions or management plans. In contrast, the historical narratives and oral traditions of First Nations extend well past the earliest arrival of Europeans and often contain precise information about the environment and environmental change. For example, oral traditions often contain information about catastrophic environmental events such as floods, earthquakes, volcanic eruptions, and unusual weather, as well as descriptions of extinct flora and fauna (cf. Cruikshank 1981).

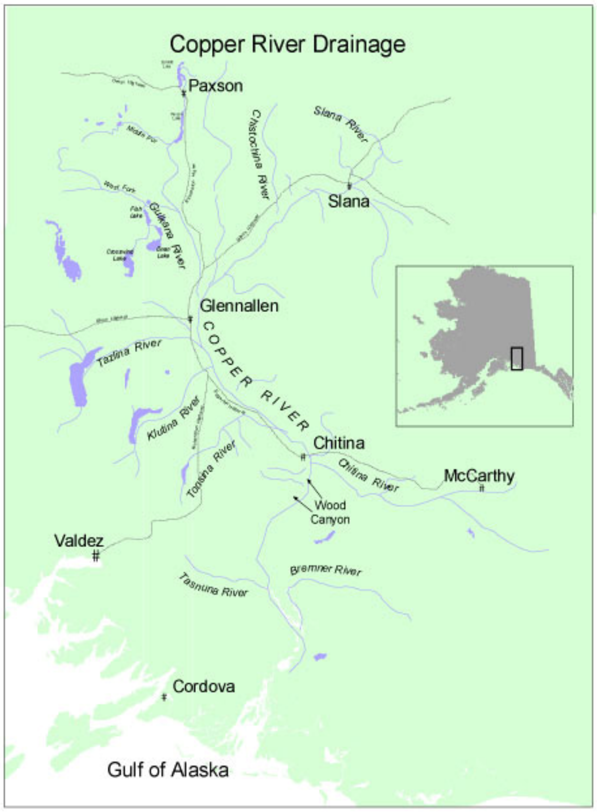

Figure 1. Map of the Copper River area, Alaska, USA.

Traditional knowledge includes considerable detail. Hunters and fishers acquire extensive knowledge of the environment because of the variety of activities they undertake in all seasons of the year. Their dependence on animals and plants for food, clothing, and tools requires a detailed knowledge of when and where resources are available and the environmental processes that affect their availability. This breadth of knowledge is reflected in traditional classification systems that are often much more extensive than those provided by science. Learning how First Nations classify natural systems provides us with a more detailed and nuanced view of the environment.

Ahtna Athabaskans, a First Nation living in Alaska, have developed an accurate and complete taxonomy of all fish species found in the Copper River Basin and gained knowledge of salmon distribution, salmon life histories, and behaviour.

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Figure 2. Andy Tyone of Gulkana pulling a chinook salmon out of his fish wheel on the Copper River, Alaska. Fish wheels were introduced to the Ahtna at the beginning of the 20th century and are now the preferred method for catching salmon on the Copper River. Each fishwheel is registered and receives a number from the Alaska Department of Fish and Game. All fish wheels are home made, usually out of logs and lumber. The Ahtna have a tradition that no metal is to be used in the construction of a fish wheel because the salmon are believed not to like metal.

The Ahtna lexical inventory for fish is a good example of local people’s ability to accurately describe local fauna. In the Ahtna language there are terms for 19 species of fish, including all 14 species found in the Copper River Basin, and inventoried by the Alaska Department of Fish and Game (ADF&G). The additional five species exist outside the basin and are known to Ahtna through trade. The Ahtna taxonomy for fish is divided into two empirical categories, ***tsabay***, which are fish other than salmon, and the more general term used for the class *Pisces*, and ***łuk’ae***, a term referring both to salmon in general and sockeye in particular.

For the term ***łuk’ae*** there is considerable lexical embellishment revealing extensive and specific knowledge of salmon ecology. For example, the Ahtna language includes terms covering almost every phase in the life cycle of salmon. Salmon alevin, are ***łuk’ae yiige*** (salmon’s spirit); salmon fingerling are referred to as ***łuk’ae ggaay*** (little salmon); little salmon fry headed down stream are called ***’uł’uli*** (those that are swimming past); spawning fish are ***tazdlaexi*** (those that are swimming in water), and dead salmon are called ***tułtaeni*** (the one that is dead in water). Female salmon are referred to as ***K’unn’i*** (the roe one), and male fish are ***tl’ets’i*** (the milt one). Seasonal variations of fish are also noted. Full sized, prime early

Figure 3. Processing salmon on the Copper River. Today some Ahtna keep fish camps but others bring their fish home to process them. On the left side of the photograph is a smoke house made from logs and chicken wire. ***Ba’*** or drying salmon can be seen hanging. Using the traditional method, the salmon are first covered with dust and placed in pits for one or two days and then soaked in the river. This removes some of the grease and makes them easy to handle. The heads are then removed and left to soak while the carcass is split and the backbone removed from the meat. The fish are hung for a week or more until they are dried and then bundled up and stored in a cache.

running sockeye are called ***nulaeggi*** (island swimmer), and late running sockeye are named ***dak'aay*** (that which is ridged, humped). Late running sockeye in Tonsina Lake, located in the lower Copper River drainage, are called ***tsiis luugge'*** (ocher salmon), and whitefish caught in late fall at freeze up are ***nen'ten luugge'*** (frozen ground fish). The comprehensiveness of these terms indicate that Ahtna have long been aware of the various phases in the life cycle of the salmon.

Figure 4. A processed sockeye salmon ready for hanging in the smoke house. A stick is used to hold the meat open so that is will not curl up and leave a raw space where flies can lay their eggs. The meat and backbone are left attached until they are completely dried, then the backbone is removed and stored separately.

Ahtna have recognized and named 21 distinct salmon populations that emanate from particular home streams. The best known of these, recognized by biologists and Ahtna alike, are ***natael luugu'*** 'roasted salmon fish,' the large sockeye bound for Tanada Lake, located in the Wrangell Mountains at the head of the Copper River. These populations are similar to the salmon stocks identified by biologists of the Alaska Department of Fish and Game, but whereas biologists differentiate between stocks that spawn at different locations within the same system, Ahtna do not. Biologists, for example, consider sockeye bound for Tanada Lake as two separate stocks, one that spawns at the outlet of the lake and one that spawns in the lake, but Ahtna classify all sockeye from Tanada Lake as ***natael luugu'.***

First Nations have put their knowledge of the environment into practice by developing successful management strategies. Traditional management systems are community based. Management is in the hands of the resource users who adhere to the rules in response to social pressure, cultural mores, and/or ideological conviction rather than government or administrative authority (Feit 1988: 74). The advantage of such systems is that they are designed around a common set of values that everyone understands and accepts. Decisions are not made at a distance or from the top down but locally. One key to implementing successful management strategies is to have the users understand and accept the goals and objectives of the resource managers. For this to happen the users have to have a stake in management.

The 'self management' systems developed by First Nations involve both an understanding of ecological processes and a code of ethics that govern human-environmental relationships. These ethical standards stem from a belief system, or worldview, that is ecological in nature. From this perspective everything in the environment is linked, there is no separation of society from nature. The individual is considered part of a complex web of relationships that includes both human society and the natural environment. Behaviour in all relationships, whether with humans or animals, is guided by a set of principles that stress cooperation, restraint, and balance. Animals are considered powerful actors who freely give themselves to humans, if humans treat them appropriately. Proper treatment involves the sustainable use of animals, maintaining a clean habitat, and taking only what you need without waste.

To avoid waste Ahtna carefully gauge their harvest against the capacity to process the fish. Once this capacity is reached the harvest is suspended, so that fish are not unnecessarily caught and spawning fish can escape. Ahtna are also concerned with catching the right kinds of salmon. To make ***ba'*** or dried fish**,** Ahtna select salmon based on their sex and reproductive condition, preferring male salmon to females because the former are larger and fatter. As one Ahtna elder remarked:

> *That what he used to do, he [we] keep more males…just throw em back in river. Sometime he [we] take em all, sometime he let the female go. That's why he used to have a lot of fish long time ago.* ***Kata'ile'i****, (spawning salmon) they let them go.*

In the past when female salmon were caught in a dip net or trap they were released, but modern fishing technology has altered this practice. Fishwheels run during the night when no one is around, so people are obliged to keep all of the fish they catch. As Ahtna elders note, old fishing practices were in place to "save everything," that is to ensure a sustained yield.

In summary, traditional knowledge can contribute to basic scientific research and to resource management. First Nations have detailed knowledge of their environment and an understanding of long-term ecological processes. Their knowledge provides a time depth that is unsurpassed in the North in its continuity and can help explain ambiguities found in other kinds of evidence that can be incorporated into research. While traditional management systems are rooted in an understanding of the human-nature relationship different from science they can provide us with insights that could spark alternative explanations about the natural world.

To gain understanding means sharing information, which requires creating venues where all parties can feel comfortable sharing information (cf. Pinkerton 1990: 335). Effective communication requires acknowledging that local people do have valuable information or insights, and that scientists and managers have legitimate views and concerns. The objective is to build relationships with local people so that managers and locals can develop common goals.

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