Abstract

The contributions in this report stem from a workshop held in April 2000 to review the methodology deployed by the research team of the *Sea Around Us Project.* This project, funded by The Pew Charitable Trusts, Philadelphia, USA, is designed to provide an integrated analysis of the impacts of fisheries on marine ecosystems, and to devise policies that can mitigate and reverse harmful trends whilst ensuring the social and economic benefits of sustainable fisheries. The data–rich North Atlantic was selected as the target area for case studies to be conducted in the first two years of the project, with other areas to follow in subsequent years. The methodology deployed by the project includes: (1) the development of a spatially explicit catch and effort information system that allows in-depth analysis of fisheries catches for various large marine ecosystems, i.e., reported landings, nominal catches, unreported catches, misreported catches, and discarded by-catch, sorted by species and sector; (2) the quantification of the biological and economic impacts of the present fishing trends or a change thereof on the ecosystems, with reference to past ecosystems reconstructed from time series of scientific data and the Ecopath with Ecosim software; (3) the quantitative evaluation of the status of fisheries by sector, gear type and location using a robust and simple system of rapid appraisal (Rapfish) that may be applied to past, present and alternative future fisheries; (4) approaches for scaling all results to a basin-wide scale; and (5) quantification of the economic and other benefits to be gained from re-establishing healthy ecosystems, relative to the losses expected from a continuation of the status quo. An important feature of the methodology assembled to meet these requirements is that it does not compete with the elaborate single-species methodology conventionally applied to the management of fisheries, and which generally pertain to geographic and temporal scales much smaller than the basin-wide scale considered by the *Sea Around Us Project.*

Daniel Pauly

Principal Investigator
Sea Around Us Project
Professor of Fisheries
UBC Fisheries Centre

Director's Foreword

The Fisheries Centre at the University of British Columbia supports research that first clarifies, and then finds ways to mitigate, the impacts of fisheries on aquatic ecosystems. Only with such insight of how whole aquatic ecosystems function can management policies aim to reconcile the extraction of living resources for food with the conservation of biodiversity, with the maintenance of ecosystem services, with amenity and with other multiple uses of aquatic ecosystems. Indeed, the present dire state of marine ecosystems and their fisheries around the globe signals a pressing need for what may be termed the "ecosystem imperative."

Although ecosystem agendas of this kind have recently become embodied in the legislative goals of many nations, and are an integral part of the FAO *Code of Conduct for Responsible Fisheries*, in practice there have been few attempts to work out how it might actually be done. In sponsoring the *Sea Around Us Project*, the Pew Charitable Trusts of Philadelphia, USA, have devoted a significant amount of funding to an ambitious pilot project that focusing on the North Atlantic that aims to address this question. A research team of senior scientists, postdoctoral research assistants, graduate students, consultants and support staff commenced work in late 1999.

Members of this team are excited and challenged by the unprecedented scope of the research work. Moreover, most of the methods used to tackle the problem magnitude are new. It seems that in concentrating on the perfection of quantitative methods that set catch quotas for large heavily-industrialized fisheries, traditional fisheries science has avoided trying to address ecosystem-based questions since the days of the pioneers in the early 20th century.

This report presents the edited output of a workshop held in May 2000 that examined the methodological bases of the research for the *Sea Around Us Project.* Each of the papers has been subjected to peer review by at least two referees, and has been scrutinized before and during the workshop by a visiting team of experts from FAO and major fisheries management agencies in Canada, the USA and Europe (See Appendix 2).

Summary comments about the project and its methods from these visiting experts are reported in Appendix 3. The *Sea Around Us* research team are especially pleased with the overall support that the project is receiving from FAO, DFO, ICES and others.

The report is the latest in a series of *Fisheries Centre Research Reports* published by the UBC Fisheries Centre. A full list is shown on our web site at www.fisheries.ubc.ca, and the series is fully abstracted in the *Aquatic Sciences* and *Fisheries Abstracts.* The research report aims to focus on broad multidisciplinary problems in fisheries management, to provide a synoptic overview of the foundations and themes of current research, to report on research work-in-progress, and to identify the next steps and ways that research may be improved.

Edited reports of workshops reported in *Fisheries Centre Research Reports* are distributed to all project or workshop participants. Further copies are available on request for a modest cost-recovery charge. Please contact the Fisheries Centre by mail, fax or email to office@fisheries.ubc.ca.

Tony J. Pitcher

Professor of Fisheries
Director, UBC Fisheries Centre

Preface and Acknowledgement

The contributions included in this report originate from a workshop held from April 1st to 5th, 2000 at Dunsmuir Lodge, Sydney, Vancouver Island, B.C., and devoted to reviewing the methodology to be deployed by the research team of the *Sea Around Us Project.*

This project, fully funded by the Pew Charitable Trusts, Philadelphia, USA, is designed to provide an integrated analysis of the impacts of fisheries on marine ecosystems, and to device policies that can mitigate and reverse harmful trends whiles ensuring the social and economic benefits of sustainable fisheries. The data-rich North Atlantic was selected as the target area for cast studies to be conducted in the first two years of the project, with other areas to follow in subsequent years.

The *Sea Around Us Project* aims to collate and analyze catch and ecosystem information using analytical tools being developed at the Fisheries Centre, in partnership with a global network of scientists providing data, evaluating and peer review. These elements are required in developing strategies and action plans to manage marine ecosystems.

Thus, the methodology deployed by the project includes:

1. The development of a catch and effort information system that allows in-depth analysis of fisheries catches for each ecosystem, i.e., reported landings, nominal catches, unreported catches, misreported catches, discarded by-catch, kill by ghost-fishing, sorted by species and sector;
2. The quantification of the biological and economic impacts of the present fishing trends or a change thereof on the ecosystems, with reference to past ecosystems reconstructed from time series of scientific data;
3. The quantitative evaluation of the status of fisheries by sector, gear type and location using a robust and simple system of rapid appraisal that may be applied to past, present and alternative future fisheries;
4. Approaches for scaling all results to a basin-wide scale;
5. Quantification of the benefits to be gained from re-establishing healthy ecosystems, relative to the losses expected from a continuation of the *status quo.*

An important feature of the methodology assembled to meet these requirements is that it does not compete with the elaborate single-species methodology conventionally applied to the management of fisheries, and which generally pertain to geographic and temporal scales much smaller than those considered by the *Sea Around Us Project.* Thus, we were able to build on the results of traditional approaches in fisheries sciences to derive our methodology, which we hope will be seen as complementary to traditional approaches.

In fact, the *Sea Around Us Project* has much progressed since the workshop documented here was held, and already, some of the methods in this report have been modified after they were applied to a wide range of concrete situations. Interested readers are advised therefore to consult the project web page (at www.fisheries.ubc.ca/projects/SAUP) for current versions, and sample results.

We conclude by thanking the Pew Charitable Trusts for their support of the *Sea Around Us Project.* Thanks are also due to the dedicated staff of the *Sea Around Us Project*, and to our panel of invited experts: Lee Alverson, Kevern Cochrane, Poul Degnbol, Paul Fanning, Richard Grainger and Jay Maclean.

We are most grateful to the following external referees for providing their comments in a timely and insightful manner: Ragnar Arnason, Trond Bjorndal, John Blaxter, Cutler Cleveland, Michael Fogarty, Kenneth Frank, Quentin Grafton, Normal Hall, Rognvaldur Hannesson, Paul Hart, Simon Levin, Pamela Mace, Paul Medley, Leif Nottestad, David Pimentel, David Ramm, Saul Saila and Michael Sinclair.

Daniel Pauly and Tony Pitcher

September 2000.

ASSESSMENT AND MITIGATION OF FISHERIES IMPACTS ON MARINE ECOSYSTEMS: A MULTIDISCIPLINARY APPROACH FOR BASIN-SCALE INFERENCES, APPLIED TO THE NORTH ATLANTIC[1]

D. Pauly and T.J. Pitcher

Fisheries Centre, University of British Columbia, Vancouver, Canada, V6T 1Z4

ABSTRACT

The aim of the *Sea Around Us Project* is to quantify, in ecological and economic terms, the impact of fisheries on the marine ecosystems of the North Atlantic, and to evaluate the costs and benefits of various scenarios of mitigation, such as *status quo*, rebuilding of depleted resources and implementation of closed areas. Dealing with these issues requires a methodological package related to, but different from, that typically used in fisheries management, notably because of its ecosystem focus and the much larger temporal and spatial scales, relative to standard fisheries assessments. This paper summarizes the methodology deployed by the project by introducing a suite of papers in which its rationale and operational details are provided.

First, we review the relationships between scale and methodology choices in marine science. Then, the principle modules of the *Sea Around Us Project* methodology are described as follows:

1) The North Atlantic as study area, where we report a new ecosystem classification scheme that is compatible hierarchically with previous work and with all statistical divisions;
2) North Atlantic fisheries catches in time and space, where we present the project's catch and effort database, discuss the problems in estimating total extractions, and outline methods used to overcome them;
3) Fish distribution transects, where the biology and migrations of key commercial North Atlantic species are used to link catches by shallow-water and offshore fisheries;
4) Bio-economic analyses of fisheries sectors, where the effect of competition between small and large –scale fisheries is quantified using a multi-species, multi-gear yield per recruit model and the combination of effort producing a Nash equilibrium is identified;
5) Ecosystem modeling, discussing the use of ECOPATH with ECOSIM and ECOSPACE to represent present and past North Atlantic ecosystems with their embedded fisheries, to evaluate ecosystem status, and to simulate likely response to change;
6) Evaluating alternative ecosystem-based management regimes to quantify the benefits of different ecosystem-based management scenarios;
7) Energy consumption and the ecological footprint of North Atlantic fisheries, to contrast the energy incorporated in landed fishes to that required to catch them;
8) Rapid interdisciplinary appraisal of fisheries status and compliance analyses using RAPFISH, to compare and characterize North Atlantic fisheries in terms of their sustainability (in ecological economic technological and social fields), analysis of their ethical status, and to score their compliance with the FAO Code of Conduct for Responsible Fisheries, together with the compliance of North Atlantic countries vis-à-vis their internationally agreed commitments.
9) Mapping the fate of fisheries landings from the North Atlantic, to identify possible pressure points for intervention by fish product consumers;

We present a diagram expressing the articulation of the various methodological components listed above. The synthesis to emerge from integrating the results of these modules may contain many surprises, both in terms of the ecological damage and economic waste presently generated by the North Atlantic fisheries, and in clarifying the foregone benefits that could be regained, were these economic and ecological issues to be addressed.

INTRODUCTION

The task of the *Sea Around Us Project*, funded by the Pew Charitable Trusts, Philadelphia, and executed at the Fisheries Centre, University of British Columbia, Vancouver, is to provide a synthesis of the impacts of fisheries on marine ecosystems of the North Atlantic. More precisely, the questions to be answered are:

1. What are the total fishery catches from the ecosystems? Total fishery catches includes both reported and unreported landings and discards at sea.
2. What are the biological impacts of these withdrawals of biomass for the remaining living components of the ecosystem?
3. What would be the likely biological and economic impacts of a continuation of current fishing trends (i.e., a maintenance of the status quo)?
4. What were former states of this ecosystem like before the expansion of large-scale commercial fisheries?
5. How does the present-day ecosystem evaluate on a scale from 'healthy' to 'unhealthy'?
6. What specific policy changes and management measures should be implemented:

   (a) to avoid continued worsening of the present situation?
   (b) to improve ecosystem 'health', as defined in (5)?

Each of these questions, though straightforward-looking at first, leads to further questions, many seemingly without answers. Nevertheless, the project staff has developed a 'methodology package' for providing the best possible answers to these questions. This package differs from that normally used to assess local fish populations and local fisheries in that our methods are scalable to the entire North Atlantic basin, and indeed, eventually, to the world ocean. This package therefore, emphasizes aspects of fisheries and other marine science that are usually given short thrift in local studies. Conversely, we do not attempt to assess the exploitations status of exploited single-species fish populations. As we shall attempt to demonstrate, methods concerned with local or single-species studies and those in our methodology package support and complement each other.

Before we present the various elements of this methodology package, we shall briefly contrast two views of (marine) sciences, and provide reason why, given the present, much depleted state of North Atlantic fish populations, and the ruinous state of the fisheries depending thereon, we have chosen to identify with one of these views.

Two views of (marine) sciences

Our reading of the history of science in general, and marine science in particular suggests two basic way that advances are made:

1. Through what, for lack of a better term, we shall call Smart New Tricks (SNT), or
2. Through assimilation of large sets of pre-existing data, and, based thereon, through the creation of New Mental Maps (NMM).

Examples of SNT in fisheries were the invention of Virtual Population Analysis (usually attributed to Gulland, 1965), or of Bayesian risk analysis (reviewed by Punt and Hilborn 1997). SNT usually resolve one problem (often one that was not even perceived as such), and do this in a new way that is often regarded as 'neat' or 'elegant'. On the negative side, we should add that SNT can also be seen as 'techno-fixes', resolving the technological aspect of a problem but usually leaving wide open the underlying process that generated the problem. In the case of the two examples above, the problems were how to estimate fishing mortality, and how to present management options to politicians, respectively. Their downside as techno-fixes was that the former quickly bred a misplaced confidence in its outputs (see Walters and Maguire 1996, Pitcher and Hart 1982), whilst the latter, even though labelling them as such, provided ultra-risky options to industry and politicians (Mace 2000), decision-takers who, by the nature of their professions, tend to prefer risky options to safer ones.

The alternative to the SNT, the NMM can sometimes build on one or several small SNT. The important feature of the NMM, however, is that it involves the assimilation (or meta-analysis) of large (sometimes enormous) data sets. Our best example is the realization by U.S. Navy Commander Mathew F. Maury, in the mid-1800s, that mariners collectively held in their head enough information on currents and winds (e.g., in the North Atlantic), to generate maps which would improve navigation, i.e., shorten the route between Europe and the Americas (Maury 1963).

Maury thus promised cooperating mariners copies of his planned maps, should they agree to contribute their individual knowledge on most favorable routes. These data (and depth soundings he also gathered) enabled him not only to produce, after lots of painstaking work, the best navigation maps then in existence ('applied' science), but also to be the first to perceive the existence of mid-Atlantic ridge ('basic' science). Moreover, single-handedly created the mode of interactions between mariners and naval offices that still prevails, and which has enabled the emergence of modern physical oceanography as a discipline wherein data are *shared*. Hence the existence and collaboration, even during the coldest years of the Cold War, of Data Center A (in Washington, D.C.) and B (in Moscow). This, incidentally, is also the reason why oceanographic data can be used to verify the occurrence of global changes: the data are available since the late 19th Century.

Which brings us to marine biology and fisheries. Here, like Maury since the end of the 19th Century, we inherit a mountain of data on the various organisms, from phyto-plankton (net samples, C_{14} measurements, satellite oceanography) and zooplankton (Hensen nets samples, Hardy samplers time series), trawl and benthic surveys, catch time series, landing and price data, etc. – an enormous, ever-growing data set. Yet we are very often told by fisheries scientists and others that there are "no data" upon which to make inferences about the state of North Atlantic ecosystems, and on remedial actions regarding their depletion, and on the future of the commercial species therein. We are told that what we need is 'new, better data', or indeed that we should hope for a SNT to somehow resolve the problem(s) that led to the mountain of data being accumulated in the first place.

Yet major NMM are based on assimilation of existing data, even in areas with which all are familiar. Thus, for example, it is relatively well known that the report which convinced the US authorities, and the US public, and later others in other parts of the world, that cigarettes are bad for smokers did not present a SNT. Rather, it was meta-analysis of a large number of small studies, each perhaps not very convincing by itself, but jointly providing incontrovertible evidence. Further, more recent meta-analyses added the effects of second-hand smoke, now leading to widespread restraints on smoking in enclosed spaces, both public and private. What changed here is the position of cigarettes in peoples' mental maps.

In a similar way, Rachel Carson's *Silent Spring* assimilated into a coherent whole a large number of previously unconnected observations, and this created a NMM wherein the location of DDT and other pesticides was radically different from it previous position (Lear 1997).

The Convention of Biological Diversity (CBD) requires that all the countries of the world make inventories of their biodiversity, and take measures to protect it. Where does small country X get a global reference list of the plants and animals that have so far been described (and of which the species in country X must be a subset)? Such a list still does not exist, despite the straightforward nature of the science that would be required (just as for Maury's maps).

In the late 1980s, work on a large database intended to provide a rigorous nomenclature and classification for all the fishes in the world, and key facts for each of these 25,000 species. Ten years later, the job is largely done (see www.fishbase.org): the countries of the developing world now thus have a tool that enables them to get started on meeting their obligations vis-à-vis the CBD, at least concerning the fishes (presently, the Internet version of FishBase gets over half a million visits per month, several orders of magnitude more than for any comparable product). Moreover, the database thus created, in a collaborative mode resembling Maury's, has many elements serving as model for Species 2000, which aims at producing a list of all organisms so far described (see www.species2000.org).

An excellent example of a meta-analysis is the series of contributions by R.A. Myers and collaborators on the stock-recruitment relationships of fishes, based on their vast compilation of time series of published stock and recruitment time series. This work recently culminated in Myers et al. (1999) and has the potential to produce massive changes in the mental maps of fisheries biologists. Myers' study shows conclusively that the common feature of stock-recruitment relationships across species (a narrow range of slopes near the origin, indicative of a narrow range of reproductive potentials of individual female fish) was not seen previously because nobody bothered to standardize, over a large number of cases, the scales of plots of recruitment versus parent stocks. Rather, earlier authors emphasized the 'uncertain', even 'chaotic' nature of stock-recruitment relationships, entirely missing what turns out to be highly predictable relationships. It is as if Maury had complained about the 'complex' nature of mariners' knowledge, rather than assemble his maps.

These items are examples of NMM, major pieces of work that make available to practitioners tools that assimilate much of the work previously done in a given area. In each case data was already available in principle, but was not assimilated within a rigorous framework. So we can ask: why are there not more of these collaborative exercises in marine biology and fisheries, given their potential impact?

One reason might be that, in the context of government-funded research, such work can be done only after a consensus has emerged about the research to be conducted, the idea being that such NMM should emerge *from the bottom up*. The problem here is the tendency for collective and committee-led research to reduce new sets of ideas, 'visions' as it were, to a least common denominator: voluminously documented research proposals favoring safe science over risky new approaches.

The methodology package we have assembled to answer the questions above, related to the impacts of fisheries on the ecosystems of the North Atlantic, thus reflect our vision, not yet widely shared, that such questions can be tackled at basin-wide scales. The methodology is devoted to assimilating, in rigorous, quantitative terms, a large amount of previous work and to involving multiple collaborative arrangements. However, we shall maintain standards such that coherent products emerge.

Such approach, from *the top down* is, we believe, the only way products can emerge which are useful at scales above that at which marine and fisheries biologists typically operate, usually that defined by the boat of a university research station, or by the commercial vessels used in a fishery under study.

The North Atlantic as Study Area

As defined by the *Sea Around Us Project*, the North Atlantic includes all marine waters North of Miami, Florida in the West and North of Cape Bojador, Morocco in the East. This area is identified in Fig. 1, which also identifies the Biogeochemical Provinces (BGCP), which are compatible with the Large Marine Ecosystems (Sherman and Duda 1999) of the North Atlantic (see below). These articulate, at different levels, the ecosystem classification adopted by the project (see Pauly et al. 2000). Note that this definition excludes the Mediterranean from the scope of the project. Moreover, for various pragmatic reasons, we also exclude the Baltic proper, though not its connections with the North Sea, the Kattegat and Skagerrak. Except for a Southern border a bit further south, and the omission of the Baltic, our definition of the North Atlantic thus overlaps with the area jointly covered by FAO areas 21 (Eastern North Atlantic) and 27 (Western North Atlantic), themselves largely overlapping with the area for which ICES, and NAFO, respectively, are responsible.
The questions posed of the *Sea Around Us Project*, referring to the ecosystem impact of fisheries,

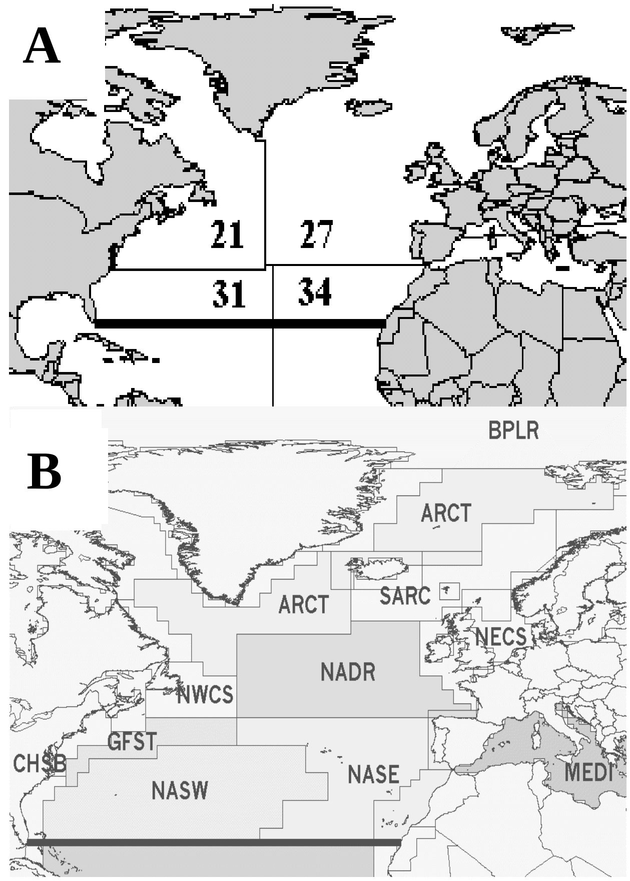


Figure 1. (A) (top) Map of North Atlantic showing that *Sea Around Us Project* area (southern boundary is thick horizontal line) overlaps four major FAO statistical areas. (B) (bottom) The nine major biogeochemical provinces in the *Sea Around Us Project* area. ARCT=Atlantic Arctic Province (in two regions); NECS = Northeast Atlantic Shelves Province; SARC = Atlantic Subarctic Province; NADR = North Atlantic Drift Province; NASE = North Atlantic Subtropical Gyral Province (East); NASW = North Atlantic Subtropical Gyral Province (West); GFST = Gulf Stream Province; CHSB = Chesapeake Bay Province; MEDI = Mediterranean. For further details of how these zones are conflated with Sherman's Large Marine Ecosystems, ICES and NAFO management areas, and USA and Canadian statistical zones, using a half-degree square *Sea Around Us* database, see Pauly et al. (2000).

require that we identify the ecosystems of the North Atlantic. In the spirit of the foregoing, which emphasizes the need to assimilate large amount of pre-existing data, we have adopted, for the *Sea Around Us Project*, the large Marine Ecosystem (LME) concept and definitions developed in the last 15 years by K. Sherman and co-workers, and recently summarized in Sherman and Duda (1999).

This decision was facilitated by the discovery that the LMEs so far defined can be easily mapped onto, and re-expressed as components of coastal Biogeochemical Provinces (BCGP), the larger ecosystem units proposed by Longhurst (1995, 1998) to provide a stratification of the world ocean.

Indeed, this redefinition of LME provides the lower rungs of a hierarchy ranging from 'biomes', i.e., large, circum-terrestrial entities with similar climate (Polar; Westerlies; Trades; and Coastal Boundary) to 56 BGCP and about 80 LME (see Pauly et al. 2000). Moreover, the LME themselves can be further subdivided, especially for modeling purposes (see Pauly et al. 2000 and Christensen and Walters 2000).

This structure for ecosystem classification, proposed as a consensus of several research groups working on this type of issue (Pauly et al. 2000), appears well suited for the stratification required for basin-level estimates of various states and rates and to address the issue of variability of scales emphasized by Levin (1990). Moreover, using this scheme, fisheries may be mapped onto the ecological entities, the ecosystems, that generate the fish caught, and not the artificial boundaries of countries, EEZs, and jurisdictions, our next topic.

North Atlantic Fisheries Catches in Time and Space

Accurate time series of fisheries catches, here understood as *all* animals killed by fishing gears, and not only those that are *landed*, are at the heart of the *Sea Around Us Project*. However, contrary to what may be believed, assembling such time series for the North Atlantic is not a matter of setting up a new program for sampling primary data in the countries bordering the North Atlantic. Rather, it is largely a matter of identifying, for each of these countries, those elements (if any) that prevent their official catch statistics from reflecting the true effects of fishing gears.

In many cases, even landings are incomplete because the data collecting entity is not mandated to collect data from certain types of gear (often small-scale gear, or sport fishers), notwithstanding the potential impacts of a large number of such gear.

In other cases, obvious sources of biases, notably massive discarding of by-catch are not considered in compiling catch statistics. This also applies to illegal catches, even when, as occur in some fisheries, all those involved – including government scientists - know of their existence, and even their magnitude.

Watson et al. (2000) review these and related issues, and thereby present the database structure and methodology we shall use to obtain, for the North Atlantic, figures that will better reflect true catches (i.e., all withdrawals) than those presently available, illustrated by an example of cooperation with a government agency. Moreover, Pitcher and Watson (2000) explore this issue further, by estimating percentage in each category of unreported catches, following in time the changes in legal instruments, including the Law of the Sea, that provide disincentives to accurate reporting. The analysis is presented such that it can be easily refined by further work.

However, even the first round of estimates resulting from these considerations should contribute to making our catch figures more realistic. This contrasts with the assumption of zero in those categories, the common default position of public agencies, and one that is neither useful nor acceptable to the public itself.

We are well aware that the data set thus assembled will remain fragmentary and incomplete, and that far better data sets will exist on local scales. At the LME and basin-wide scale, however, we expect that our data set will be the most accurate, in that all sources of fishing mortality will be accounted for.

Pauly et al. (2000) present the method by which the global FAO fisheries catch data set will be re-expressed on a global LME map. The key component of the method proposed therein is that it will proceed 'by subtraction', i.e., by first assigning fishes with clear affinities to depth ranges, habitat types and/or certain LME, e.g. the anchoveta *Engraulis ringens* to the inshore part of the Humboldt Current LME, or the neritic fishes reported for Bangladesh to the shelf component of the Bay of Bengal LME, etc., each time subtracting the assigned fish groups from the database. Several rounds of subtraction will lead to small amounts of unallocated landings, pertaining mainly to fish landed in countries with distant water fleets (or providing flags of convenience to such fleets). Assigning the residual landings to the LME where these fleets are known to occur (see Bonfil et al. 1999 and references therein), in proportion to the catches per half-degree square previously allocated, will be sufficient for a first-pass allocation, especially since misallocations should generate visible patterns in the maps thus generated.

For the North Atlantic, this crude approach can be replaced by one in which the catch reported by species, from distinct ICES or NAFO sub areas is assigned to the half-degree squares in each area as a function of the mean depth of each square, and the observed depth distributions in the species in question, as plotted on the 'depth transects' presented below (see also Zeller and Pauly 2000). Here again, misallocations should generate visible patterns in the maps thus generated, and thus lead to improvements of the allocation rules.

FISH DISTRIBUTION TRANSECTS

As mentioned above, the *Sea Around Us Project* will not attempt to perform assessments of single-species fisheries, and not generally question such assessments as performed by various colleagues.

However, we do require connecting our work with key aspect of the distribution of major commercial species, for two reasons:

1) These distributions can help assign catches to areas (see above); and
2) The depth and distance from the coast of major population components determines their relative vulnerability to coastal (often small-scale) and offshore (often large-scale) gear and hence the existence and intensity of interactions and (potential) conflicts between these different fisheries.

The format we have developed for these transect fulfils these requirements by integrating the key information on the distribution and migration of fish in a single graph (see Zeller and Pauly 2000). Using such a graph, catches of both small- and large- scale fisheries, both inshore and offshore, can be partitioned and their impacts evaluated.

Bio-economic analyses of fisheries: small vs. large

Few, if any studies have quantified the economic rent lost from competition between the large- and small-scale sectors of a fishery. Here we have chosen an approach with three important key features:

1) Easily scalable from local fisheries to the entire North Atlantic;
2) Provides management alternatives by emphasizing, where possible, the substitutability of large by small scale fisheries (and vice versa);
3) Should lead to a reliable estimate of economic losses (waste) due to excess capacity and non-cooperative behavior between different elements of the fisheries sector (see Nash 1951, 1953).

This approach uses a multispecies, multifleet yield-per-recruit analysis to estimate, based on the present, calculated recruitment (= influx of young fishes and invertebrates to the fishing grounds), the features of a 'small scale' and a 'large-scale' fleet which maximize the gross value of the catches of both fleets. These features are the level of effort relative to present, and the selection curves of each gear relative to each species. Then, under the assumptions that the present fisheries are at or near their bioeconomic equilibrium point (where total costs equal gross total returns; Gordon 1954), and that fishing mortality scales linearly to fishing cost, we identify the equilibrium point at which maximum net returns can be obtained if the fleets adjusted their fishing mortality such that their joint net benefit is maximized (Munro 1979; Sumaila 1997).

The difference from between these optional returns and the Nash Frontier to the present position of the fleet allows the loss (=economic waste) due to non-cooperation and mismanagement. Finally, we partition benefits by sector and identify the Nash bargaining solution (Nash (1953) associated with the equilibrium point (Binmore 1982). This procedure can be applied successively to a large sample of representative North Atlantic fisheries, thus yielding, by addition, an overall estimate of economic losses, and, more importantly of the economic gains that would result from improved management (Ruttan et al. 2000). We expect these numbers to be very large, especially when scaled up to our reference area, through the ratio of the sum of all catches in the sample fisheries to the total North Atlantic catches.

The Achilles' heel of this approach is, of course, the assumption that for each fishery, relative recruitment, as obtained by dividing yield per recruit into average catches, will remain constant while fishing mortality varies. We note, however, that the approach we propose will tend to associate the Nash equilibrium with levels of fishing mortality lower than those commonly presently occurring in real fisheries (which tend to suffer from growth overfishing). This implies that recruitment would be assumed to remain constant over a small range of F-values only.

Moreover, the proposed method will treat each fishery independently from the others, using distinct mixes of species, each with their own sets of relative recruitment, and growth and selection parameters. Thus, given the Central Limit Theorem, our global estimate of economic loss will tend to be accurate, even if the estimates for certain fisheries are not.

Another aspect of our comparative bioeconomic studies of small-scale vs. large-scale fisheries is that they should provide a framework for evaluating government policies which purport to benefit employment, or other social goods: small-scale and large-scale fisheries often sharply differ in the employment opportunities or other social benefits they provide (see section below on RAPFISH; and Alder et al. 2000).

ECOSYSTEM MODELLING

Embedding the fisheries that generate the catches and economic returns discussed above into ecosystems will be achieved by constructing at least one ECOPATH model for each of the LME in the North Atlantic. The rationale for ECOPATH as modeling tool is that it is the only approach so far demonstrated to be widely applicable for modeling marine ecosystems, notwithstanding a common misunderstanding as to the ready availability of alternative approaches. Christensen and Walters (2000) review ECOPATH as used in the context of the *Sea Around Us Project*, with emphasis on this and other misunderstandings regarding the capabilities and limits of the approach it embodies.

Presently, ECOPATH models exist for numerous parts of the world (see Pauly et al., 2000). However, only 20 of these represent ecosystems of the North Atlantic basin, hence precluding simple raising of biomass flows from ecosystem to basin scales. Thus, a stratification scheme is required, based on the geographic structure outlined above, which can be used to scale models from the sampling area of the field data used to parameterize the models to the wider area that is assumed represented by these same models.

LMEs are seen here as providing the key level for ecosystem model construction. For each LME, an ECOPATH model will be constructed to describe the ecosystem resources and their utilization, and to ensure that the total fisheries catch of each LME is used as output constraint (just as their primary production will be used as input constraint). In addition, the stratification scheme used must be such that it can straightforwardly accommodate any number of additional ECOPATH models for each LME. This can be done so as to simultaneously address the issue of parameter uncertainty, as described in Pauly et al. (2000).

The LME ECOPATH models require information on abundance, production and consumption rates and diets for all ecosystem groupings. Such information can be obtained from the following sources:

- Abundance, production and consumption rates, and diets of marine mammals are available from the *Sea Around Us Project* database for all (117) species of marine mammals and on a seasonal basis;
- Fishery catches: available from the spatially structured catch database generated as described above (see also Watson et al. 2000), and covering all species groups;
- Occurrence, biology and ecology of marine fishes: available from FishBase (www.fishbase.org) at LME level for the North Atlantic, as a result of cooperation between the *Sea Around Us Project* and FishBase projects.
- For marine invertebrates: only limited information (beyond the catches in the FAO database) is available from electronic databases, but a variety of publications provide extensive information. Production rates can be estimated from the well-founded empirical relationships of Brey (1999), now included in ECOPATH;
- Primary production estimates: establishment of a global database aimed at supplying fine grid level satellite based estimates of primary production is presently underway through a cooperation between the Space Applications Institute, EC Joint Research Centre, Ispra, Italy, and several members of the *Sea Around Us Project*.

The LME-level ECOPATH models will serve as the backbone for addressing issues related to fisheries impacts, to derive indices related to ecosystem health (Rapport et al. 1998a; 1998b; Costanza and Mageau. 1999), to evaluate, using ECOSIM and ECOSPACE (Walters et al. 1997, 1999; Walters and Christensen 2000), the likely effects of changes in fishing patterns, including setting up of marine protected areas, and to estimate the expected economic benefits of such interventions.

Moreover, these LME-level ECOPATH models, representing the present states of the systems in question, will also serve as templates for models of selected areas (notably the Gulf of Maine, Newfoundland and the North Sea) for reconstructions representing these systems prior to the onset of large scale mechanized fisheries, and the ensuing resource depletion. Thus, LME-level ECOPATH models of past ecosystems will provide the basis for estimating the benefits that would obtain from rebuilding strategies, as required to address Question 6 in the Introduction (see also Figure 5). The next section provides more details on this issue.

Evaluating alternative ecosystem-based management regimes

To complement the analysis of small- and large-scale fisheries as outlined above, leading to an estimate of potential economic gains from improved management, we will simulate the results of various management regimes, and evaluate their results in the framework of fisheries economics, extended to make it applicable to ecosystem analysis.

The extended theory is then applied to explore a number of questions including (i) to what extent is it worth society's while to restore current ecosystems to their past states? (ii) What is the optimal approach path to the past ecosystem? Is it optimal to invest (disinvest) rapidly in restoring the ecosystem, or should investment (disinvestment) proceed more slowly?

ECOPATH and ECOSIM models will form the ecological basis for our analysis, while ecological economics valuation techniques will help determine the economically feasible restoration plans and paths (see Munro and Sumaila 2000).

Mapping the fate of fisheries landings from the North Atlantic

The validity of the analyses described above depends on the markets presently existing for fish products, and their likely evolution. We propose therefore, that a spreadsheet-based framework can help track the flow of fish landings within the North Atlantic region (details in Sumaila et al. 2000).

Starting with the total fish landings from the waters of each major fishing nation within the North Atlantic region, a map can be developed showing how these landings flow into the major product forms under which they are marketed, i.e., fresh, frozen, salted and smoked. In addition, the portion of the product forms are consumed in the domestic versus the export market can be determined. Finally, the results derived can be used to identify the sectors or product forms which capture most of the economic benefits from the fishes of the North Atlantic.

Energy consumption and ecological footprint of the North Atlantic fisheries

One way to express the overcapitalization of North Atlantic fisheries (i.e., the excess of catching capacity) is to relate the energy dissipated in generating present landings to the energy contained in the landings.

This appears more straightforward than estimating fleet 'capacity', which is not only hard to measure, but even hard to define. Energy expenditures, on the other hand are easily defined, and can be estimated reasonably well from the size of the vessels, which relates strongly to that of their engines, and hence to their fuel consumption.

Hence our choice of Horsepower·days as measure of effort, a choice having the further advantage of allowing comparisons between otherwise widely different boat/fishing gear combination (see Watson et al. 2000).

The estimation of energy consumption by the fishing fleets of the North Atlantic, and the related estimation of their ecological footprint (Wackernagel and Rees 1996), are presented by Tydmer (2000), who provides details, as well, on the required distinction between variable energy costs (associated with running vessels) and fixed costs, associated with the construction and eventual retirements of the vessels comprising a fleet.
We anticipate that the aggregate energy costs of fishing, in the North Atlantic, will be very high, relative to the energy (and commercial value) of the landings, the difference being met by various subsides.

RAPFISH and compliance analyses

Evaluations in the *Sea Around Us Project* employ a new multi-disciplinary, rapid appraisal technique, called RAPFISH, that focuses on the comparative sustainability of fisheries (Pitcher and Preikshot 1998; Pitcher et al. 1998a; Pitcher et al. 1998b; Preikshot and Pauly 1998; Preikshot et al. 1998; Pitcher and Preikshot, in press). RAPFISH can be performed even when the rigorous survey data that enables conventional stock assessment are not available, as is the case for many North Atlantic fisheries.

As such, RAPFISH is a typical SNT, a smart new trick as defined above. It is however, suitable for the *Sea Around Us Project* because it allows us to quantify aspects of fisheries thought before to be unquantifiable, and thus allows for comparisons. Moreover, the method can be applied at all scales relevant to the *Sea Around Us Project*, from the fisheries of a small bay of gulf, to those of countries, or of the entire North Atlantic. As well, RAPFISH can be used to compare gears, and thus to contribute its unique perspective to the comparisons between small- and large-scale fisheries mentioned above.

In RAPFISH analyses, sets of attributes, chosen to reflect sustainability within each discipline, are scored on a ranked or binary scale. Where data are sparse or uncertain, scores may be refined when better information becomes available. Ordinations of sets of attributes are performed using multi-dimensional scaling followed by scaling and rotation. The leverage of each attribute on the results can be estimated with a step-wise procedure. The ordinations are anchored by fixed reference points that simulate the best (= 'good') and worst ('bad') possible fisheries using extremes of the attribute scores, while other anchors secure the ordination in a second axis normal to the first. Significant differences are defined by Monte Carlo simulation of errors attached to the original scores. Raw plots of the results show fisheries status in relation to 'bad' and 'good'.

Separate RAPFISH ordinations are performed in evaluation fields (disciplines) that express status in terms of ecological, economic, social, technological and ethical (Pitcher and Power 2000) sustainability: a further field evaluates compliance with the FAO Code of Conduct for Responsible Fisheries (Pitcher 1999). Status results may be combined in a hierarchical way in 'kite diagrams' (see Figure 2) to facilitate comparison of fisheries by gear type, country, ecosystem or size category, and data may be constructed to represent the outcomes of alternative policies (Alder et al. 2000).

At this stage in the SAU project, we present a paper reporting preliminary RAPFISH analyses of fisheries in two major North Atlantic areas, the Gulf of Maine and the North Sea (Alder et al. 2000). By the end of the SAU project all major fisheries will be covered by RAPFISH evaluations. This will allow examination, for each country, of fisheries compliance with the FAO Code of Conduct for Responsible Fisheries. Compliance scored in this way will be also be evaluated using a matrix expressing international fisheries conventions to which each country in the North Atlantic is signatory.

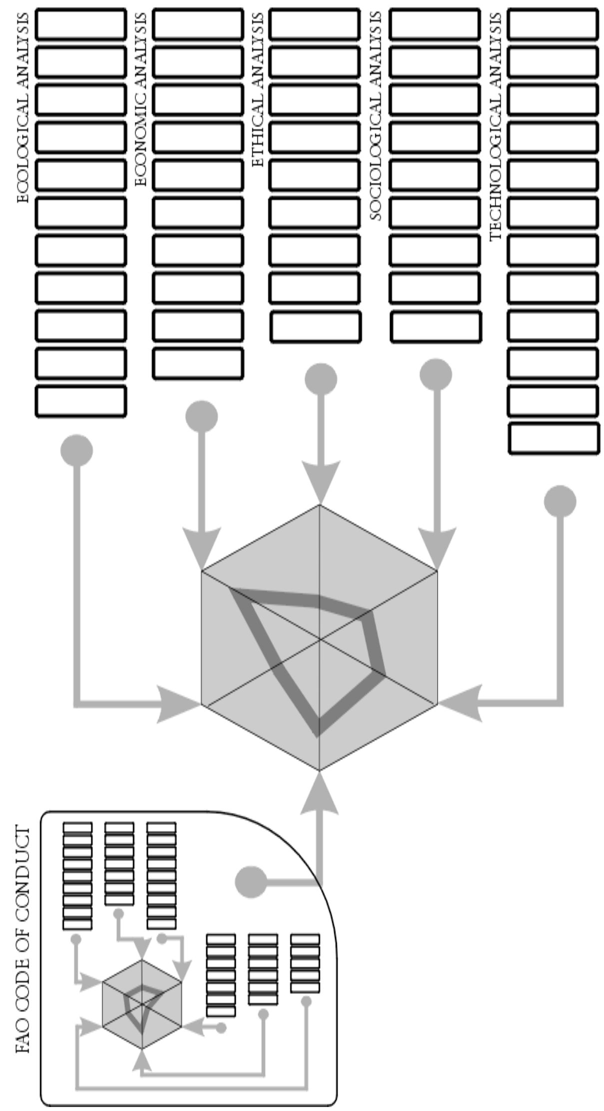


Figure 2. Diagram illustrating how RAPFISH evaluation fields for different modalities of sustainability can be considered together as scores on the axes of a kite diagram. Boxes represent the attributes used to ordinate fisheries within each evaluation field. Connections, arrows and kite apices represent a score between 0% and 10% from each field. The outer rim of the kite is equivalent to 100% scores (= 'good') in each field, while the centre of the kite represents scores of 0% (= 'bad'). Six evaluation fields are illustrated here, one of which, for the Code of Conduct, is comprised hierarchically of a five-field RAPFISH.

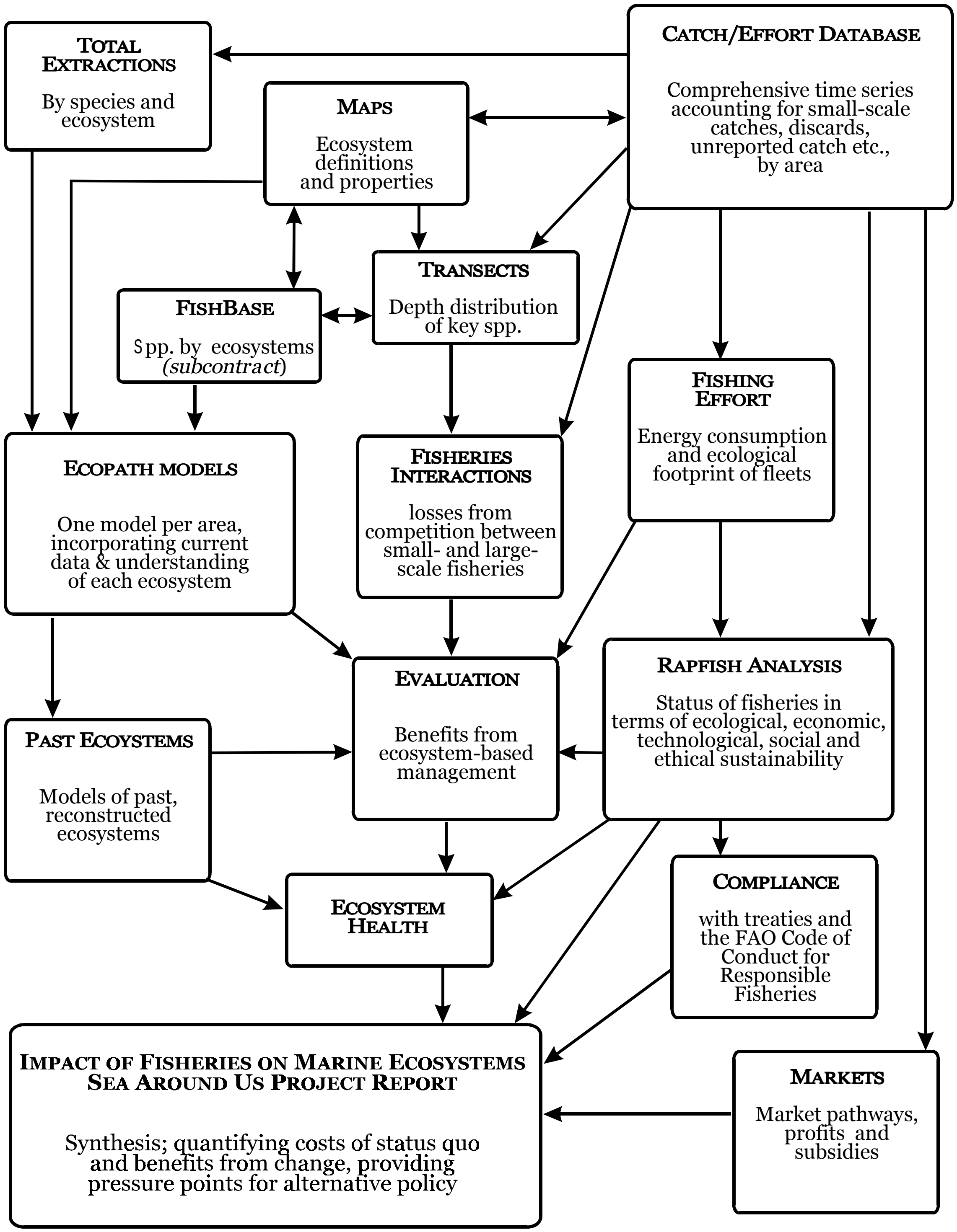


Figure 3. Conceptual diagram illustrating the relationships of the various methodological elements of the *Sea Around Us Project.*

CONCLUSIONS

The relationships among the various elements of the *Sea Around Us Project* are summarized in Figure 5. We anticipate that the synthesis to emerge from integrating the results of these modules will contain many surprises, both in terms of the ecological damage and economic waste presently generated by the North Atlantic fisheries, and the benefits that could be gained, were these economic and ecological issues addressed.

ACKNOWLEDGMENTS

The *Sea Around Us Project* is supported by the Environment Program of The Pew Charitable Trusts. We would particularly like to thank Dr. Joshua Reichert, for his support of the ideas which led to the development of this project. Also, we thank Drs Reg Watson, Rashid Sumaila, and Dirk Zeller for detailed comments on the draft of this contribution.

REFERENCES

Alder, J. Pitcher, T.J., Preikshot, D., Kaschner, K. and Ferriss, B. (2000) Rapfish estimates - how good is good? In: Pauly, D. and Pitcher T.J. (eds) Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. *Fisheries Centre Research Reports* 8(2), 136-182.

Binmore, K. (1982) *Fun and games: a text on game theory*. Chancellor Press, London.

Brey, T. (1999) A collection of empirical relations for use in ecological modelling. *Naga, the ICLARM Quarterly*, 23(3), 24-28.

Christensen, V. and Walters, C. (2000) Ecopath with Ecosim: Methods, capabilities and limitations. In: Pauly, D. and Pitcher T.J. (eds) Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. *Fisheries Centre Research Reports* 8(2), 79-105.

Costanza, R. and M. Mageau. (1999) What is ecosystem health? *Aquatic Ecology* 33, 105-115.

Gordon, H.S. (1954) Economic theory of a common property resource: the fishery. *Journal of Political Economy* 62, 124-142.

Gulland, J.A. (1965) Estimation of mortality rates. Annex to the Report of the Arctic Fisheries Working Group, *ICES C.M.* 1965/3, 9 pp.

Lear, Linda. (1997) *Rachel Carson: witness for nature. Henry Holt and Company*, New York. 634 pp.

Levin, S.A. (1990) Physical and biological scales and the modelling of predator-prey interactions in Large Marine Ecosystems. In: K. Sherman, L.M. Alexander, and B.D. Gold (eds.). *Large Marine Ecosystems: patterns, processes and yields.* Amer. Ass. Adv. Sci. Washington, D.C., 179-187.

Longhurst, A.R. (1998) *Ecological Geography of the Sea.* Academic Press, San Diego 398 pp.

Longhurst, A.R. 1995. Seasonal cycles of pelagic production and consumption. *Progress in Oceanography* 36, 77-167.

Mace, P.M. (2000) MSY reborn, but with a new identity: *Fish and Fisheries*. (in press.)

Maury, M.F. (1963) *The physical geography of the sea and its meteorology*. The Belknap Press of Harvard University Press, Cambridge, 432 pp.

Munro, G.R. (1979) The optimal management of transboundary renewable resources. *Canadian Journal of Economics*, 12(8), 355-376.

Munro, G. and Sumaila, U.R. (2000) Investment in capture Fishery resources: An ecosystem approach. In: Pauly, D. and Pitcher, T.J. (eds) Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. *Fisheries Centre Research Reports* 8(2), 106-113.

Myers, R.A., K. G. Bowen, and N.J. Barrowman. (1999) Maximum reproductive rate of fish at low population sizes. *Can. J. Fish. Aquat. Sci.* 56, 2404-2419.

Nash, J.F. (1951): Non-cooperative games. *Annals of Mathematics* 54, 286-295

Nash, J.F. (1953): Two person cooperative games. *Econometrica* 21, 128-295.

Pauly, D., Watson, R., Longhurst, A., Sherman, K. and Platt, T. (2000) Mapping Fisheries onto Marine Ecosystems. In: Pauly, D. and Pitcher T.J. (eds) Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. *Fisheries Centre Research Reports* 8(2), 13-22.

Pitcher, T.J. (1999) Rapfish, A Rapid Appraisal Technique For Fisheries, And Its Application To The Code Of Conduct For Responsible Fisheries. *FAO Fisheries Circular* 947, 47 pp.

Pitcher, T.J. and Preikshot, D. (1998) Rapid Appraisal of the Status of West African Distant Water Fleet Fisheries Relative to Home Fleets using the Rapfish Technique. In: Bonfil, R., Munro, G., Sumaila, U.R., Valtysson, H., Wright, M., Pitcher, T., Preikshot, D., Haggan, N. and Pauly, D. (eds.) Distant water fleets: an ecological, economic and social assessment. *Fisheries Centre Research Reports* 6 (6), 90 – 93.

Pitcher, T.J. and Preikshot, D.B. (2000) Rapfish: A Rapid Appraisal Technique to Evaluate the Sustainability Status of Fisheries. *Fisheries Centre Research Reports*. (in press).

Pitcher, T.J. and Power, M.P. (2000) Fish Figures: Quantifying the Ethical Status of Canadian Fisheries, East and West. In: Coward, H., Ommer, R. and Pitcher, T.J. (eds). *Just Fish: the ethics of Canadian fisheries*. Institute of Social and Economic Research Press, St John's, Newfoundland, 225-253

Pitcher, T.J. and Hart, P.J.B. (1982) *Fisheries Ecology*. Chapman and Hall, London. 414 pp.

Pitcher, T. and Watson, R. (2000) The basis for change Part 2: Estimating the total extractions from marine ecosystems. In: Pauly, D. and Pitcher, T.J. (eds) Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. *Fisheries Centre Research Reports* 8(2), 40-53.

Pitcher, T.J., Bundy, A., Preikshot, D., Hutton, T. and Pauly, D. (1998a) Measuring the unmeasurable: a multivariate interdisciplinary method for rapid appraisal of health of fisheries. In: Pitcher, T.J., Hart, P.J.B. and Pauly, D. (eds.) *Reinventing Fisheries Management*. Kluwer, Dordrecht, Holland, 31-54.

Pitcher, T.J., Mackinson, S., Vasconcellos, M., Nøttestad, L. and Preikshot, D. (1998b) Rapid appraisal of the status of fisheries for small pelagics using multivariate, multidisciplinary ordination. In*:* Quinn, T.J. II, Funk, F., Heifetz, J., Ianelli, J.N., Powers, J.E., Schweigert, J.F., Sullivan, P.J. and Zhang, C.I. (eds) *Fishery Stock Assessment Models.* Alaska Sea Grant, Fairbanks, USA, 759-782

Preikshot, D.B. and Pauly, D. (1998) A multivariate interdisciplinary assessment of small-scale tropical fisheries. In*:* Quinn, T.J. II, Funk, F., Heifetz, J., Ianelli, J.N., Powers, J.E., Schweigert, J.F., Sullivan, P.J. and Zhang, C.I. (eds) *Fishery Stock Assessment Models.* Alaska Sea Grant, Fairbanks, USA, 803-814

Preikshot, D.B., Nsiku. E., Pitcher, T.J. and Pauly, D. (1998) An interdisciplinary evaluation of the status and health of African lake fisheries using a rapid appraisal technique. *J. Fish Biol.* 53 (Suppl A), 382-393.

Punt, A.E. and Hilborn, R. (1997) Fisheries stock assessment and decision analysis: the Bayesian approach. *Rev. Fish. Biol. Fish.* 7, 35-63.

Rapport, D. J., Costanza, R. and McMichael, A.J. (1998) Assessing ecosystem health: challenges at the interface of social, natural and health sciences. *Trends in Ecology and Evolution* 13,397-402.

Rapport, D., Costanza, R., Epstein, P., Gaudet, C. and Levins, R. (1998) *Ecosystem Health.* Blackwell Scientific, New York, 372 pp.

Ruttan, L.M., Gayanilo, F.C. Jr. and Pauly, D. (2000) Small versus large scale fisheries: a multispecies, multifleet model for evaluating their potential to conserve fishes and fisheries. In: Pauly, D. and Pitcher, T.J. (eds) Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. *Fisheries Centre Research Reports* 8(2), 64-78.

Sherman, K. and Duda, A.M. (1999) An ecosystem approach to global assessment and management of coastal waters. *Mar. Ecol. Progr. Ser.* 190, 271-287.

Sumaila, U.R. (1997) Cooperative and non-cooperative exploitation of the Arcto-Norwegian cod stock in the Barents Sea. *Environmental and Resource Economics* 10, 147-165.

Sumaila, U.R., Chuenpagdee, R. and Munro, G. (2000) Tracking fisheries landings in the North Atlantic. In: Pauly, D. and Pitcher, T.J. (eds) Methods for assessing the impact of fisheries on marine ecosystems of the North Atlantic. *Fisheries Centre Research Reports* 8(2), 114-122.

Tyedmers, P. (2000) Quantifying the Energy Consumed By North Atlantic Fisheries. In: Pauly, D. and Pitcher, T.J. (eds) Methods for assessing the impact of fisheries on marine ecosystems of the North Atlantic. *Fisheries Centre Research Reports* 8(2), 123-135.

Wackernagel, M. and Rees, W.E. (1996) *Our Ecological Footprint: Reducing Human Impact on the Earth.* New Society Publishers, Gabriola Island, British Columbia. 160 pp.

Walters, C. Pauly, D. and Christensen, V. (1999) Ecospace: prediction of mesoscale spatial patterns in trophic relationships of exploited ecosystems, with emphasis on the impacts of marine protected areas. *Ecosystems* 2, 539-554.

Walters, C. Christensen, V. and Pauly, D. (1997) Structuring dynamic models of exploited ecosystems from trophic mass-balance assessments. *Rev. Fish Biol. Fish.* 7(2), 139-172.

Walters, C.J. and Maguire, J. J. (1996) Lessons for stock assessment from the northern cod collapse. *Rev. Fish Biol. Fish.* 6, 125-137.

Watson, R., Guenette, S., Fanning, P. and Pitcher, T.J.(2000) The basis for change: Part 1 - Reconstructing fisheries catch and effort data. In: Pauly, D. and Pitcher, T.J. (eds) Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. *Fisheries Centre Research Reports* 8(2), 23-39.

Zeller, D. and Pauly, D. (2000) Life history patterns and depth zone analysis. In: Pauly, D. and Pitcher, T.J. (eds) Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. *Fisheries Centre Research Reports* 8(2), 54-63.

MAPPING FISHERIES ONTO MARINE ECOSYSTEMS: A PROPOSAL FOR A CONSENSUS APPROACH FOR REGIONAL, OCEANIC AND GLOBAL INTEGRATIONS[a]

Daniel Pauly, Villy Christensen, Rainer Froese, Alan Longhurst, Trevor Platt, Shubha Sathyendranath, Kenneth Sherman and Reg Watson

ABSTRACT

Research on ecosystem-based fisheries management, marine biodiversity conservation, and other fields requires appropriate maps of the major natural regions of the oceans, and their ecosystems.

It is proposed here that a classification system proposed by T. Platt and S. Sathyendranath and implemented by A.R. Longhurst, defined largely by physical parameters, and which subdivides the oceans into four 'biomes' and 57 'biogeochemical provinces' (BGCPs), could be merged with the system of 50 Large Marine Ecosystems (LMEs) identified by K. Sherman and colleagues, which would represent subunits of the provinces. This arrangement enhances each of the systems, and renders them mutually compatible. For the LMEs, subprovinces are pragmatically defined to serve as a framework for the management of coastal fisheries, and other purposes, while the BGCPs have rigorous physical definitions, including borders defined by natural features. Moreover, incorporating the 50 defined LMEs into the framework of BGCPs will allow straightforward scaling-up of LME-specific flow estimates (including fisheries catches) up to basin and ocean scales. The combined mapping will allow the computation of GIS-derived properties such as temperature, primary production, etc., and their analysis in relation to fishery catch data for any study area.

A further useful aspect of the proposed scheme is that it will enable us to quantify the EEZ of various countries in terms of the distribution of marine features (e.g., primary production, coral reef areas) which has yet to be straightforwardly associated with coastal states. Applications to shelf, coral reef and oceanic fisheries, and to the mapping of marine biodiversity are briefly discussed.

INTRODUCTION

There is a broad consensus in the scientific community that fisheries management should be ecosystem-based, but very little agreement as to what this means (NRC 1999). Also, there is a need to analyze biodiversity data at larger scales than generally done so far, as demonstrated by, e.g., Sala et al. (2000) for terrestrial and freshwater biomes.

Clearly, when dealing with such complex issues, the first task, as in all science-based approaches to a problem, is to define the object(s) of concern, and to develop a consistent method to show how these objects are interrelated. Here, the objects are the marine ecosystems within which fisheries and biodiversity are to be analyzed, and marine life in general, is embedded.

Fortunately, establishing a consensus on the classification of marine ecosystems may be relatively easy, given the compatibility, so far never elaborated upon, of two classification schemes proposed in recent years. Both of these integrate enormous amount of empirical data, and are sensitive to previous analyses of marine ecology. These two schemes are (1) the global system of 57 'biogeochemical provinces' (BGCPs) developed by Platt and Sathyendranath (1988, 1993), Platt et al. (1991, 1992), Sathyendranath et al. (1989), Sathyendranath and Platt (1993), implemented by Longhurst (1995, 1998), and defined at scales appropriate for understanding physical forcing of ocean primary production and related processes; and (2) the 50 coastal Large Marine Ecosystems (LMEs) gradually defined by Sherman and co-workers (see e.g. Sherman et al. 1990, 1993), whose size and on-shelf location makes them particularly suitable for addressing management issues, notably those pertaining to fisheries on continental shelves, and coastal area management (Sherman and Duda 1999).

After reviewing selected features of these two schemes, we suggest how the partition of ocean regions that they imply can be made mutually compatible. The joint classification which then emerges is presented in form of a spatial hierarchy, and as maps, each emphasizing a key feature of the classification. Overall, the integrated scheme we propose allows explicit consideration of different scales, as discussed e.g. by Levin (1990).

[a] presented as C.M. 2000/T:14 at the Annual Science Conference of the International Council for the Exploration of the Sea.

Biogeochemical Provinces

This partition of the ocean is derived from a suggestion of Platt and Sathyendranath (1988) for the recognition of natural regions of the ocean, having characteristic physical forcing to which there is a characteristic response of the pelagic ecosystem. These regions were to be dynamic biogeochemical provinces ('dynamic' because their boundaries would respond to annual and seasonal changes in physical forcing, and 'biogeochemical' because within each the biota would respond to those characteristic geochemical processes which determine nutrient delivery to the euphotic zone). This concept has been used to partition both global and basin-scale analyses of primary productivity, though the 'dynamic' boundary aspect of the system remains to be exploited: so far, most applications of the partition have assumed that boundaries between provinces were fixed at locations representing average conditions, though dynamic boundaries have been used for analysis of Arabian Sea productivity.

The central principle in locating boundaries between provinces is that of the critical depth model of Sverdrup (1953), which remains the most useful formulation relating phytoplankton growth to surface illumination, and to the vertical density structure of the water column. It successfully predicts, for example, the timing of the North Atlantic spring bloom. A proposed partition of the North Atlantic into 18 BGCPs (Platt, et al. 1995) was followed by a partition of all oceans and adjacent seas into 57 provinces (Longhurst et al. 1995 and Longhurst, 1998).

The global partition was arrived at by examination of 26,000 archived chlorophyll profiles to determine Gaussian parameters describing the regional/seasonal characteristic profiles, surface chlorophyll from 43,000 grid-points from monthly Coastal Zone Colour Scanner images, and about 23,000 monthly mean mixed layer depths, together with other oceanographic variables. This analysis suggested that a two-level partition would be required adequately to represent regional differences in the expression of the Sverdrup model. The first partition is into a small number of biomes, following the usage of this term by terrestrial ecologists to mean a region of relatively uniform dominant vegetation type, with its associated flora and fauna: grassland, tundra, steppe, humid forest and so on (Golley 1993). Secondly, these biomes are each be partitioned into a number of regional entities, the biogeochemical provinces.

The four biomes (Figure 1) are defined by the dominant oceanographic process that determine the vertical density structure of the water column, which itself is what principally constrains the vertical flux of nutrients from the interior of the ocean.

In the *Polar biome*, vertical density structure is very largely determined by the flux of fresh or low-salinity water derived from ice-melt each spring and which forms a prominent halocline in polar and sub-polar oceans. In oceanographic terms, this occurs in each hemisphere polewards of the Oceanic Polar Front, whose location in each ocean is determined by the characteristic circulation of each. Though looming large on Mercator maps, the Polar biome occupies only about 6% of the ocean's surface.

Between the polar Fronts and the subtropical convergence in each ocean lies the *Westerlies biome*. Here, large seasonal differences in mixed-layer depth are forced by seasonality in surface irradiance and wind stress. Biological processes consequently may have sufficiently strong seasonality so that a spring bloom characterizes the plankton calendar.

Across the equatorial regions, between the boreal and austral subtropical convergences lies the *Trade-wind biome*. Here, the conjunction between low values for the Coriolis parameter, a strong density gradient across the permanent pycnocline and weak seasonality in both wind stress and surface irradiance result in relatively uniform levels of primary production throughout the year.

Upper continental slopes, continental shelves and marginal seas comprise the *Coastal Boundary biome*. This is constrained between the coastline itself and (usually) the oceanographic front characteristically found at the shelf-edge. The single generalization that characterizes this biome is that nutrient flux in the water column is forced by a great variety of processes: coastal upwelling, tidal friction, fresh-water outflow from river mouths, etc. In the partitions discussed above, subdivision of this biome into provinces was not carried as far as might be useful for some purposes. One of the objectives of the present study is to do just that, through the introduction of subprovinces and their identifications with LMEs.

The boundaries between the biomes thus defined certainly vary seasonally and between years, as can readily be inferred from satellite images, and dynamic boundaries that respond to this

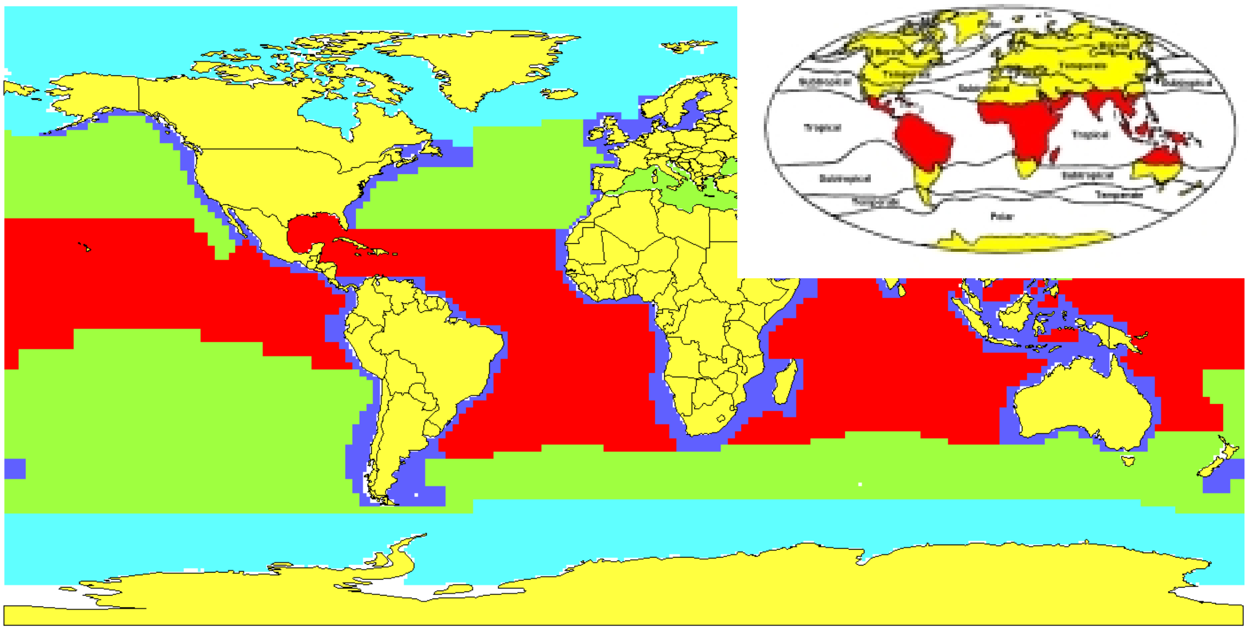


Figure 1. Map of the world ocean's biomes, the highest category in the proposed classification of the world oceans. Note its overall similarity to a conventional map of the atmospheric climate (inset, adapted from Anon. 1991). (Polar is lightest, Coastal is next darkest, followed by Westerlies and Trades)

variability are discussed for primary production and related studies by Platt and Sathyendranath, (1999). However, such dynamic schemes are neither practical nor necessarily useful for biodiversity and fisheries studies. For example, one of the tasks facing biodiversity studies are the creation of global maps documenting the distribution of hundred of thousands of marine species. Requiring that these distributions are assigned to habitats with variable boundaries would make even simple, first-order assignments of species extremely difficult and postpone the delivery of products whose need is already keenly felt by students of biodiversity.

Thus, in the case of fishes, of which about 15,000 species are marine, the assignment within FishBase (see www.fishbase.org) of species to climate type (as defined in the insert of Figure 1), required us to distinguish tropical from non-tropical species (see Pauly 1998), and this task alone required several person-months worth of work to complete.

Moreover, there are numerous types of floral or faunal assemblages whose location does not vary, though their habitat is part of, or affected by a surrounding or overlying pelagic ecosystem. Thus, the reef fishes of the Galapagos do not change their location when an El Niño event strikes the archipelago. Rather, it is their abundance which is affected (Grove 1985, Grove and Lavenberg 1997). A similar argument applies to benthic communities, whose boundaries will tend to reflect the long term average location of the boundaries of the overlying pelagic systems, rather than tracking their changing location (Ekman 1967).

The ecosystem classification scheme proposed here is thus deliberately fixed in space. On the other hand, we anticipate that its use by various authors will quickly lead to the identification and quantification of changes in species compositions, thus reintroducing the dynamic element required at various spatial and temporal scales (Levin 1990).

Oceanographic conditions within the four biomes are obviously not uniform, and each can be subdivided further using the same set of principles as determined the biomes themselves. For example, in both the westerlies and trades biomes there are definable ocean regions where heavy tropical rainfall or excessive continental fresh water runoff lead to the existence of a quasi-permanent low salinity 'barrier-layer' occupying the upper portion of the thermally-stratified surface layer. This has important biological consequences and suggests that these regions should be recognized as individual partitions.

Using such methods, based on close examination of regional physical oceanography, the four primary biomes can be further partitioned into 57 provinces, the BGCPs discussed above. Figure 2 illustrates these provinces as defined by Longhurst et al. (1995). This schema has been used to stratify the world ocean in two studies, pertaining to the global distribution of primary production (Longhurst et al. 1995) and tuna catches (Fonteneau 1998), with more forthcoming (Platt and Sathyendranath 1999, Pauly 1999).

Also, as part of the collaboration between the *Sea Around Us* project (details at www.fisheries.ubc.ca) and the FishBase project (Froese and Pauly, 2000), the world's marine fishes are presently being assigned to BGCPs, if somewhat tentatively in a few cases. We note that this work, which relies on a large number of local ichthyo-fauna lists, will require about 12 person-months to complete. However, it would require much longer were it necessary to compile first a global list of fish species, and to assign them directly to the BGCP, without prior assignment to FAO areas, countries, and oceanic islands, as is provided by FishBase.

This point is important with regards to invertebrate groups, whose global distribution will have to be mapped, in the long term, in a manner compatible to that used for fishes. This should, for example, be an important component of an Ocean Biogeographic Information System currently under consideration (Grassle and Stocks, 1999).

LARGE MARINE ECOSYSTEMS

In recent years, the formerly generic term 'Large Marine Ecosystem' (LME) has become specific, and is now mainly used for regions of ocean space encompassing coastal areas out to the seaward boundary of continental shelves and the outer margins of coastal current systems. As such, LMEs are regions of the order of 200,000 km² or greater, characterized by distinct bathymetry, hydrography and productivity patterns (Sherman 1994; Sherman and Duda 1999).

The 50 LMEs identified by Sherman and Duda (1999) are the source of about 95 % of the world's annual marine fisheries yields. Also, most of the global ocean pollution, overexploitation, and coastal habitat alteration occur within these 50 LMEs. They provide, therefore, a convenient framework for addressing issues of natural resources management. Moreover, given that most of them border developing countries, LMEs also provide a framework for addressing issues related to issues of economic development.

Various development agencies, notably the Global Environment Facility (GEF), the United Nations Development Programme, the UN Environment Programme, and the World Bank have endorsed the LME concept as framework for several of their

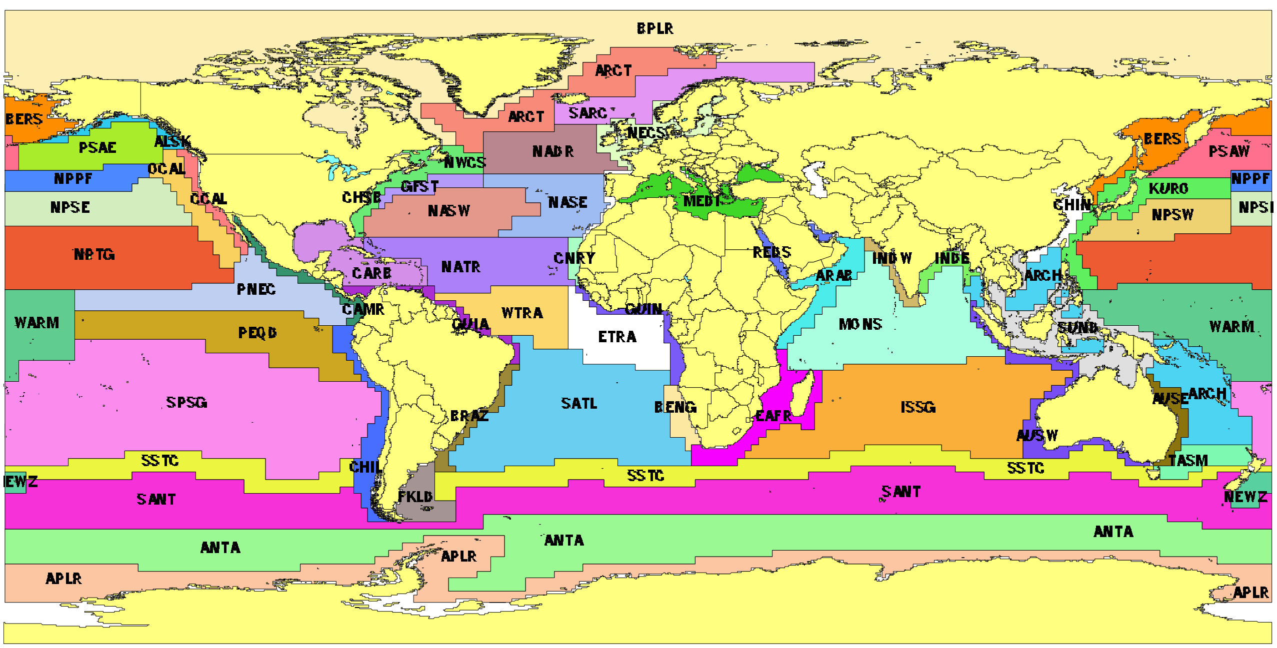


Figure. 2. Map of the world ocean's 57 biogeochemical provinces, the second level in our proposed classification of the world oceans. (The borders of a few disjunct provinces, notably ARCH, will be simplified; detailed file available from www.fisheries.ubc.ca

international development projects, for example in the Gulf of Guinea, with more such projects forthcoming (Sherman and Duda 1999). Given this considerable amount of interest, it is fortunate that a number of BGCP, i.e., those in the coastal domain, can easily be divided into 'sub-provinces' congruent with the 50 LMEs in the list of Sherman and Duda (1999). Thus, Figure 3 illustrates, for the North Atlantic, how the 15 LMEs occurring therein (including the Baltic Sea) can be mapped onto BGCPs of Figure 2, with some LMEs identified by two components (e.g. 'Southern' and 'Northern') when they straddle two provinces, and new subprovinces named where appropriate, i.e., for the parts of provinces not included in a defined LME. A similar map for the entire ocean, including all 50 LMEs in Sherman and Duda (1999) is currently in preparation (details on www.fisheries.ubc.ca).

This mapping provides, we believe, the elements that had been lacking within each of the systems thus rendered compatible. For BGCPs, we identify sub-provinces that are pragmatically defined to serve as framework for fisheries, coastal area and other applied research. As for the LMEs, they obtain, via their incorporation into the scheme of biomes and BCGPs discussed above, explicit physical definitions, including borders (here implemented in steps of half-degree squares), that allow GIS-based computation of system properties, such as mean depth, temperature, primary production, etc.

Another consideration is that our scheme for embedding LME and other subprovinces into BCGPs can be used as an ecological complement to the coarse stratification scheme used by the Food and Agriculture of the United Nations (FAO) to present global marine fisheries data, and which relies on 18 FAO statistical areas (7 for the Atlantic ocean, 3 for the Indian Ocean and 8 for the Pacific Ocean).

To facilitate comparisons between catch data stratified by these two schemes, we split the five circumpolar BCGP into ocean-specific subprovinces. This procedure enables 'closure' of the Atlantic, Indian and Pacific oceans and thus allows direct comparisons, at least at ocean-level scale, between catch data stratified within the scheme proposed here, and that used by FAO for its global catch database. Note that our next task, in this context, is to assign the catches in the

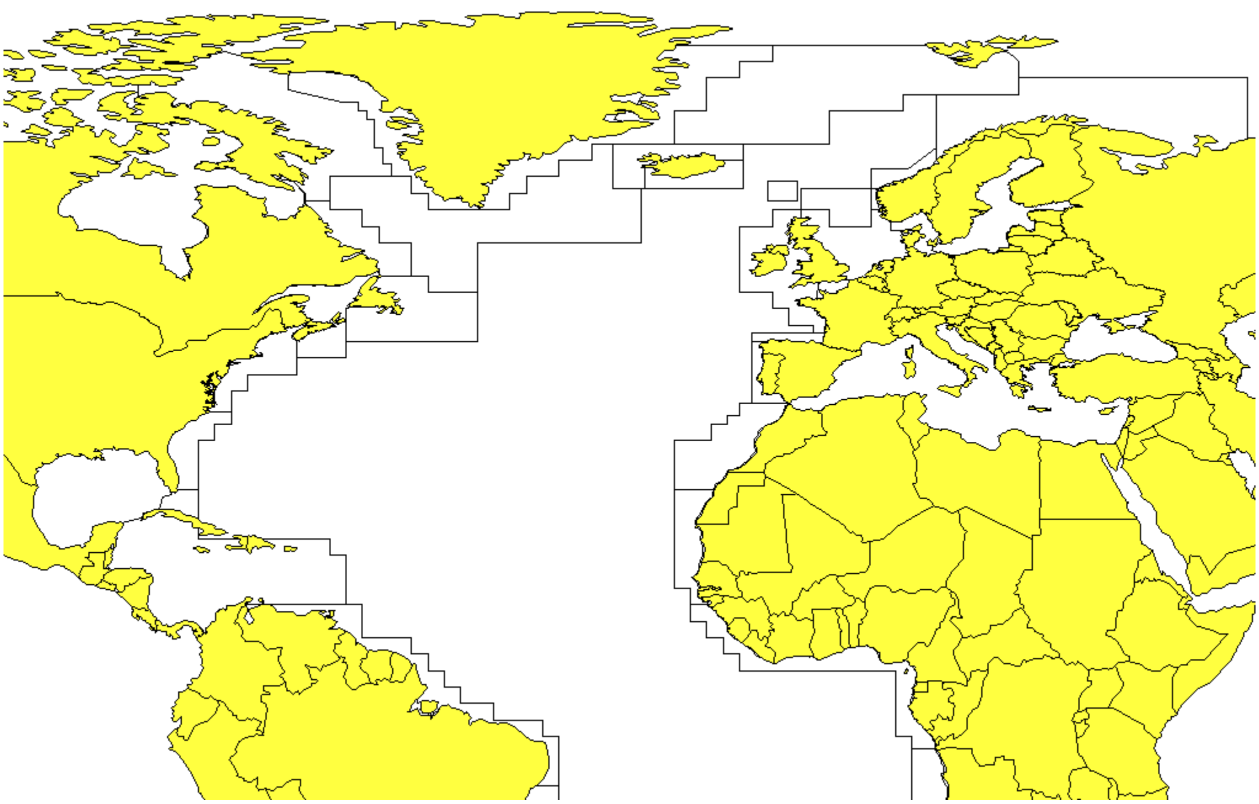


Figure 3. Map of the North Atlantic, illustrating how the LMEs identified for this basin (in Sherman and Duda 1999) can be identified with parts of biogeochemical provinces (Figure 2). Note that some LMEs may be subdivided, and their subcomponents assigned to different provinces. Also note definition of various sub-provinces for areas not currently covered by LMEs).

global FAO data set to BGCP and sub-provinces (and/or LMEs), pending its gradual replacement, starting with the North Atlantic, by locally-derived data sets. Among other things, this will allow for rapidly arraying fisheries catches and related data for comparative analyses, i.e., data now usually assembled on an *ad hoc* basis (see e.g. Caddy et al. 1998, or Pauly et al. 1998a), at scales that are often inappropriate for the intended results.

EXCLUSIVE ECONOMIC ZONES

Allocating freshwater species and their catches to countries is straightforward, as the international borders of countries are usually well defined. This is more difficult in the marine realm, where the fishes and invertebrates caught off the coast of a given country may be caught outside its territorial waters. The International Law of the Sea provides, at least in principle, a solution to this, in form of Exclusive Economic Zones, usually reaching 200 miles into the open ocean, and linking countries with much of the productive areas, i.e., the shelves adjacent to their coasts.

However, not all countries have EEZ accepted by their neighbors, and in certain areas, such as the South China Sea, the same rocky outcrops are claimed by up to half a dozen countries (McManus 1992). It cannot be expected that this and similar situations in other parts of the world will be resolved soon, nor peacefully for that matter, and we cannot expect therefore, that official maps of the EEZ will appear that could be used for assigning fisheries catches to the countries of the world.

Nevertheless, various scholars, and institutions have published EEZ maps of various parts of the world (see e.g. Mahon 1987, for the Caribbean), based on the rules for definition of EEZ established by the Law of the Sea Convention (Charney and Alexander 1993). We propose that such maps can be used to derive a coherent single map for the EEZ of the world, especially if care is taken to incorporate into such map the delimitations so far agreed though bilateral or multilateral treaties (as compiled, e.g., in Charney and Alexander 1993).

The advantage of such map is that, unlike like the map of LME and provinces mentioned above, it will enable the assignment of fish and other species, and of fisheries catch statistics to *countries*. This will enable comparisons of various features of the use and productivity of various countries' EEZ, with enough degrees of freedom for multivariate analyses, as now routinely performed for the land-based resources of various countries. It is clear, of course, that such a designation is unofficial and for scientific purposes only, and that it has no bearing, implicit or explicit, on the status of any disputes between sovereign nations about EEZ.

GLOBAL DISTRIBUTION OF CORAL REEF SYSTEMS

Coral reefs, though presently under threat throughout much of their range (Buddemeier and Smith 1999), support important fisheries wherever they occur (Munro 1996). However, quantifying these catches in reliable fashion has proven particularly difficult. One reason is that most countries with coral reefs are developing, with administrative infrastructures that preclude detailed monitoring of their fisheries.

As suggested by the pioneering work of Smith (1978), who performed the first analysis of this type, global assessment of present and potential fisheries yields from coral reefs would be much improved by comparative studies wherein the coral reef fish and invertebrate catches from various countries EEZ would be matched against the surface area of coral reefs within these same EEZ.

However, while it is possible to assign to coral reefs, at least roughly, a fraction of the catches of each country with reefs in the global FAO fisheries catch database, a matching set of coral reef area per country is not available, despite various global reviews of coral reefs (see e.g. Wells 1988; Polunin and Roberts 1996).

The model of Kleypas et al. (1999) can be used, however, to estimate expected coral reef area for any part of the world ocean with a well defined depth, temperature and light regime, and thus can be used to predict coral reef areas within each of the EEZ defined above. We anticipate, once this model becomes widely available, that plots of coral reef fish and/or invertebrate catches vs. reef area will allow us to identify countries with problematic catch data, and/or estimated reef areas, and thus to gradually improve the underlying databases and models.

SPATIAL EXPRESSION OF FISHERIES CATCH DATA

Fisheries catches are usually not reported on per-area basis (e.g. as $t \cdot km^{-2} \cdot year^{-1}$), though the areas from which they are derived are often specified. Maps of catch per area are rare, and indeed exist only for local studies, often pertaining to single-species fisheries.

Thus, one additional reason for the hierarchical system proposed above is that would allow, and make worthwhile, consistent, basin-scale and ocean-wide mapping of catches onto the ecosystems from which they originate.

We anticipate the emergence of such maps, at the global level, from two successive steps:

1) Mapping the global FAO statistics onto their (presumed) ecosystem of origin, for each of the 18 FAO statistical areas, by half-degree square;
2) Improving the map in (1) through successive replacement, by LME, of the FAO data by local data sets.

As (1) is only to provide a 'default' map, i.e., the background for locally-enriched, presumably more accurate data sets, there seem no need to allocate massive resources to this step.

Our proposed approach therefore, is to proceed by successive 'subtractions', i.e., by first assigning fishes with clear affinities to certain LME, e.g. the anchoveta *Engraulis ringens* to the Humboldt Current LME, or the neritic fishes reported for Bangladesh to the Bay of Bengal LME, etc., each time subtracting the assigned fish groups from the database.

Several rounds of subtraction should quickly lead to small amounts of unallocated landings, pertaining mainly to fish landed in countries with distant water fleets (or providing flags of convenience to such fleets). Here, we assume that assigning the residual landings to the LME where these fleets are known to occur (see Bonfil et al. 1999 and references therein), in proportion to the catches per half-degree square previously allocated, would be sufficient for a first-pass allocation, especially since misallocations should generate visible patterns in the maps thus generated.

Note that this procedure, whose application to tunas would be very problematic, does not in fact need to be applied to this group, as Fontenau (1998), based on detailed catch data from ICCAT, IATTC, and IPTP, has already allocated global tuna catches to their BGCP of origin. Similarly, the fraction of fishes in the FAO database previously assigned to the coral reefs of different countries (see above) would not require this procedure, as they would have been previously subtracted, along with Fonteneau's tunas.

Once (1) is completed, i.e., it will be straightforward to implement (2), i.e., to improve the maps for certain areas with better coverage than provided by the FAO catch statistics, e.g. the North Atlantic, where international data sets, from ICES and NAFO, and national data sets, from institutions such as DFO in Canada, NMFS in the USA, or IFREMER in France, are available.

Ecosystem Description using Ecopath

The Ecopath with Ecosim (& Ecospace) modeling approach has been recently reviewed in several contributions (Christensen and Pauly 1992, Walters et al. 1997, 1999, Pauly et al. 2000, Christensen and Walters, 2000), and there is no need here to present its working or outputs.

Ecopath models exist for numerous parts of the world (details in www.ecopath.org), including the North Atlantic. Currently, well over 100 models have been published, and more than 1800 colleagues in nearly 100 countries have registered as users of the Ecopath software system.

However, the ecosystem model coverage of various ocean basins is still spotty at best, hence precluding simple raising of flows and rates from ecosystem to basin scales. Thus, a stratification scheme is required, based on the geographic structure outlined above, and which can be used to scale models from the sampling area of the field data used to parameterize the models to the wider area that is assumed represented by these same models. The strata for the North Atlantic are presented in Figure 4.

LMEs (and other subprovinces) are seen here as providing the key level for ecosystem model construction. For each LME, an Ecopath model must be constructed to describe the ecosystem resources and their utilization, and to ensure that the total fisheries catch of each LME is used as output constraint (just as their primary production will be used as input constraint). In addition, our stratification scheme must be such that it can straightforwardly accommodate any number of additional Ecopath models for each LME. This can be done so as to simultaneously address the issue of parameter uncertainty, as briefly described below.

The LME Ecopath models require information on abundance, production and consumption rates and diets for all ecosystem groupings. Such information can be obtained from the following sources:

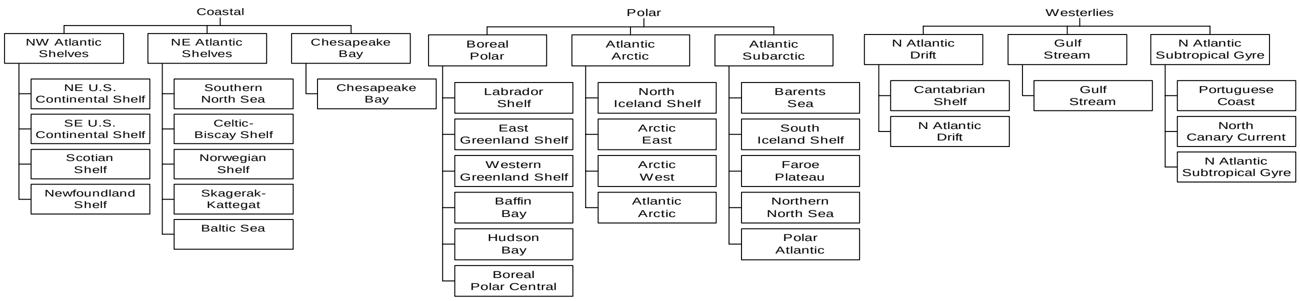


Figure 4. Proposed hierarchy of biomes, biogeochemical provinces and LME/subprovinces in the North Atlantic. Note implicit stratification, for use when, e.g., scaling up, from part of an LME, to basin or ocean-wide estimates; see also Figure 3).

- Abundance, production and consumption rates, and diets of marine mammals are available from the Sea Around Us database for all (117) species of marine mammals (see also Pauly et al 1998b, Trites and Pauly 1998);

- Fishery catches: available from the spatially structured catch database generated as described above, and covering all species groups;

- Occurrence, biology and ecology of marine fishes: soon to be available from FishBase (www.fishbase.org), presently available both at the BGCP level, and the LME/subprovince level as well. The relevant FishBase search routine option in question was designed for optimizing extraction of ECOPATH-relevant information, and is a result of the ongoing cooperation between FishBase and *Sea Around Us projects*;

- For marine invertebrates: only limited information (beyond the catches in the FAO database) is available from electronic databases, but a variety of publications provide extensive information. Production rates can be estimated from the well-founded empirical relationships of Brey (1999), now included in ECOPATH;

- Primary production estimates: establishment of a global database aimed at supplying fine grid level satellite based estimates of primary production is presently underway through a cooperation between the Space Applications Institute, EC Joint Research Centre, Ispra, Italy, and several authors of the present contribution.

The origin of each set of data (5 rate or state variable for each of the often 20-40 functional group in a model, plus a diet matrix) can be described and a related confidence interval assigned to each of the input parameters. Confidence intervals can also be estimated, as 'posterior distributions' for the output parameters of models. In addition a module of ECOPATH is designed to describe the 'pedigree' of ECOPATH models, i.e., the degree to which the models are rooted in locally sample and reliable data, (described in more details by Christensen and Walters, 2000). This module estimates, based on the pedigree of its input data, an overall quality index for each model, which in turn can serve as weighting factor, as required when dealing with discrepancies (e.g. between local vs. LME-wide catches), i.e., when raising one or several model(s) to the LME/subprovince level.

The LME/subprovince-level ECOPATH models will thus make up the backbone of our approach for addressing province, basin and global issues related to abundance, productivity, interaction and impact for ecosystem resources e.g., by trophic levels. Being based on the best available estimates of productivity and utilization of the upper trophic levels, and on productivity for the primary producers, the models are constrained from the top as well as from below.

Where possible the LME-level models will be supplemented with additional models. The procedure for this is:

- New models are assigned to strata, based on the proportion of area covered that falls within each of the depth strata < 10 m, 10-50 m, 50-200 m, 200-1000 m, and > 1000 m;

- For each new model, the confidence intervals of input and output parameters are estimated along with the pedigree index of the model;

- The LME/subprovince-level model is assigned to depth strata based using weights based on the relative primary productivity in each of the depth strata;

- Within each of the depth strata productivity, abundance, etc., are raised to the LME/subprovince level using the quality index of the models as weighting factors for the associated confidence intervals.

With this structure in place, it will be easy to add new models as they become available, and it is feasible to assign confidence intervals to all estimates derived from the analysis.

Conclusions

The ecosystem classification proposed here is not meant as a panacea that will solve all our biogeographical problems, or all spatial problems of fisheries. It should not be necessary to stress this; however, it is likely that some readers will think we believe it. We don't. However, we know that no telephone registry would ever emerge, if regular debates were held as to the optimal way to arrange the letters in the alphabet.

The ecosystem classification proposed here will soon be implemented globally by FishBase, which will thus assign all marine fish species so far described to their LME(s) or subprovince(s). It will also be used to give a geographic structure to an unofficial, spatialized, version of the FAO database of global fisheries catches (see above), thus complementing the atlas of tuna catches compiled by Fonteneau (1998), and allowing both to be related to estimates of primary production for example, mapped in similar fashion by Longhurst et al. (1995).

Moreover, this classification is fully compatible with the LME approach of Sherman and co-workers, which has led to an extensive documentation of management issues at LME scale (see references in Sherman and Duda 1999), and a number of field projects designed to address these issues, funded by various international granting agencies.

Thus, we invite colleagues to join us in expressing their results using the classification and definitions proposed here. To support this collaboration, we will supply, via the Internet, tables presenting the details of the classification by half-degree squares.

Acknowledgments

The members of the *Sea Around Us* Project thank the Pew Charitable Trusts for their support of the work leading to this contribution. Also, the first author acknowledges support of a grant from the Canadian National Science and Engineering Research Council.

References

Anon. (1991) *Bartolomew Illustrated World Atlas.* Harper Collin, Edinburgh.

Bonfil, R., Munro, G., Sumaila, U.R., Valtysson, H., Wright, M., Pitcher, T., Preikshot, D., Haggan, N. and Pauly, D. (1998) *Impacts of distant water fleets: an ecological, economic and social assessment. pp. 11-111 In: The footprint of distant water fleet on world fisheries.* Endangered Seas Campaign, WWF International, Godalming, Surrey, 111 pp.

Brey, T. (1999) A collection of empirical relations for use in ecological modelling. *Naga,* the ICLARM Quarterly, 23(3), 24-28.

Buddemeier, R.W. and Smith, S.V. (1999) Coral adaptation and acclimatization: a most ingenious paradox. *Amer. Zool.* 39:1-9.

Caddy, J.F., Carocci, F. and Coppola, S. (1998) Have peak fishery production levels been passed in continental shelf areas? Some perspectives arising for historical trends in production per shelf area. *J. North Atl. Fish. Sci.* 23, 191-219.

Charney, J.I. and Alexander, L.M.A. (eds) (1993) *International Maritime Boundaries.* The American Society of International Law. Kluwer, Dordrecht.

Christensen, V. and Pauly, D. (1992) The ECOPATH II - a software for balancing steady-state ecosystem models and calculating network characteristics. *Ecological Modelling* 61, 169-185.

Christensen, V. and Walters, C.J. (2000) Ecopath with Ecosim: Methods, capabilities and limitations In: Pauly, D. and Pitcher, T.J. (eds) Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. *Fisheries Centre Research Reports* 8(2), 79-105.

Ekman, S. (1967) *Zoogeography of the Sea.* Sidgwick and Jackson, London, 417 pp.

Fonteneau, A. (1998) *Atlas of tropical tuna fisheries.* Edition ORSTOM, Paris.

Froese, R. and Pauly, D. (eds) FishBase 2000: Concepts, Design and Data Sources. *ICLARM.* [see www.fishbase.org]

Golley. F.B. (1993) *A history of the ecosystem concept in ecology: more than the sum of its parts.* Yale University Press, New Haven and London, 254 pp.

Grassle, J.F. and Stocks, K.I. (1999) A global ocean biogeographic information system (OSIS) for the Census of marine life. *Oceanography* 12(3), 12-16.

Grove, J.S. (1985) Influence of the 1982-1983 El Niño event upon the ichthyofauna of the Galápagos archipelago. In: G. Robinson and del Pino, E.M. (eds.) *El Niño in the Galápagos Islands: the 1982-1983 event.* Publication of the Charles Darwin Foundation for the Galápagos Islands, Quito, 191-198.

Grove, J.S. and Lavenberg, R.J. (1997) *The Fishes of the Galápagos Islands.* Stanford University Press, Stanford, 863 pp.

Kleypas, J.A., McManus, J.W. and Meñez, L.A.B. (1999) Environmental limits to coral reef development: where do we draw the line? *Amer. Zool.* 39, 146-159.

Levin, S.A. (1990) Physical and biological scales and the modelling of predator-prey interactions in Large Marine Ecosystems. In: K. Sherman, K., Alexander, L.M. and Gold, B.D. (eds) Large Marine Ecosystems: patterns, processes and yields. Amer. Ass. Adv. Sci. Washington, D.C. , 179-187.

Longhurst, A.R. (1995) Seasonal cycles of pelagic production and consumption. *Progress in Oceanography* 36, 77-167.

Longhurst, A.R. (1998) *Ecological Geography of the Sea.* Academic Press, San Diego. 398 pp.

Longhurst, AR., Sathyendranath, S., Platt, T. and Caverhill, C.M. (1995) An estimate of global primary production in the ocean from satellite radiometer data. *Journal of Plankton Research* 17, 1245-1271.

Mahon, R. (ed). (1987) Report and Proceedings of the Expert Consultation on Shared Fishery Resources in the Lesser Antilles. *FAO Fish. Tech. Pap.* 383, 278 pp.

McManus, J. (1992) The Spratley Islands: a marine park alternative. *NAGA, the ICLARM Quarterly* 15(3), 4-8.

Munro, J.L. (1996) The scope of coral reef fisheries and their management. In: Polunin, N.V.C. and Roberts, C.M. (eds) Reef Fisheries. Chapman & Hall, London, 1-14.

NRC (1999) *Sustaining Marine Fisheries. National Research Council.* National Academy Press, Washington, D.C. 164 pp.

Pauly, D. (1998) Tropical fishes: patterns and propensities. In: Langford, T.E., Langford, J. and Thorpe, J.E. (eds) Tropical Fish Biology. *J. Fish Biol.* 53 (Suppl. A), 1-17.

Pauly, D. (1999) Review of A. Longhurst's "Ecological Geography of the Sea." *Trends in Ecology and Evolution* 14(3), 118.

Pauly, D., Christensen, V., Dalsgaard, J., Froese, R. and Torres, F.C. Jr. (1998a) Fishing down marine food webs. *Science* 279, 860-863.

Pauly, D., Trites, A., Capuli, E. and Christensen, V. (1998b) Diet composition and trophic levels of marine mammals. *ICES Journal of Marine Science.* 55, 467-481.

Pauly, D., Christensen, V. and Walters, C.J. (2000) Ecopath, Ecosim and Ecospace as tools for evaluating the ecosystem impact of fisheries. *ICES J. Mar. Sci.* 57, 697-706.

Polunin, N.V.C. and Roberts, C.M. (eds). *Reef Fisheries.* Chapman & Hall, London, 477 pp.

Platt, T., Caverhill, C.M. and Sathyendranath, S. (1991) Basin-scale estimates of oceanic primary production by remote sensing: the North Atlantic. *Journal of Geophysical Research* 96, 15147-15159.

Platt, T. and Sathyendranath, S. (1988) Oceanic primary production: estimation by remote sensing at local and regional scales. *Science* 241, 1613-1620.

Platt, T. and Sathyendranath, S. (1999) Spatial structure of pelagic ecosystem processes in the global ocean. *Ecosystems* 2, 384-394.

Platt, T., Sathyendranath, S., Ulloa, O., Harrison, W.G., Hoepffner, N. and Goes, J. (1992) Nutrient control of phytoplankton photosynthesis in the Western North Atlantic. *Nature* 356, 229-231.

Sala, O.E., Chapin, F.S., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D.M., Mooney, H.A., Oesterheld, M., Poff, N.L., Sykes, M.T., Walker, B.H., Walker, M. and Wall, D.H. (2000) Global Biodiversity Scenarios for the Year 2100. *Science* 287, 1770-1774.

Sathyendranath, S. and Platt, T. (1993) Remote sensing of water-column primary production. *ICES Mar. Sci. Symp.* 197, 236-243.

Sathyendranath, S., Platt, T., Caverhill, C.M., Warnock, R.E. and Lewis, M.R. (1989) Remote sensing of primary production: computations using a spectral model. *Deep-Sea Research* 36, 431-453.

Sherman, K., Alexander, L.M. and Gold, B.D. (eds). (1990) *Large marine ecosystems: patterns, processes and yields.* Amer. Assoc. Adv. Sci. Washington D.C. 242 pp.

Sherman, K., Alexander, L.M. and Gold, B.D. (eds). (1993). *Stress, mitigation and sustainability of large marine ecosystems.* Amer. Assoc. Advan. Sci., Washington D.C., 376 pp.

Sherman, K. and Duda, A.M. (1999) An ecosystem approach to global assessment and management of coastal waters. *Marine Ecology Progress Series.* 190, 271-287.

Smith, S.V. (1978). Coral reef area and the contribution of coral reefs to processes and resources of the world's ocean. *Nature* 273, 225-226.

Sverdrup, H.U. (1953) On the conditions for vernal blooming of the phytoplankton. *J. Cons. Perm. Int. Explor. Mer.* 18, 287-295.

Trites, A. and Pauly, D. (1998) Estimates of mean body weight of marine mammals from measurements of maximum body length. *Can. J. Zool.* 76, 886-896.

Walters, C., Christensen, V. and Pauly, D. (1997) Structuring dynamic models of exploited ecosystems from trophic mass-balance assessments. *Rev. Fish Biol. Fish.* 7(2), 139-172.

Walters, C., Pauly, D. and Christensen, V. (1999) Ecospace: prediction of mesoscale spatial patterns in trophic relationships of exploited ecosystems, with emphasis on the impacts of marine protected areas. *Ecosystems* 2, 539-554.

Wells, S.M. (ed) (1988) *Coral Reefs of the World.* International Union for the Conservation of Nature, Gland, and United Nations Environment Programme, Nairobi. 3 Volumes.

THE BASIS FOR CHANGE: PART 1 RECONSTRUCTING FISHERIES CATCH AND EFFORT DATA

R. Watson, S. Guénette, P. Fanning[a] and T.J. Pitcher

Fisheries Centre, UBC & [a]DFO, Halifax

ABSTRACT

Rational examination of marine policy requires an analysis of changes in the abundance of species and marine community structure with respect to past policy decisions. Abundance estimates themselves rely heavily on catch and effort statistics. There are official statistics of fish landings for many fisheries of the world. Fishing effort data is generally less available. Unfortunately, for a variety of reasons, landings data do not always reflect actual catches well. For example, discarded catches are left out of official statistics, which developed primarily to demonstrate the value of commercial landings. Illegal or unmandated (not subject to regulated reporting requirements) catches are seldom documented except in candid stock assessment discussions of major species. Through an exhaustive compilation of existing data sources and with the assistance of expert local consultants and/or partnerships, we can develop databases that present a more complete and accurate picture of the catches of marine species, including those of limited commercial significance. The importance of this process is demonstrated by our example from the Canadian North Atlantic fisheries. In this case a partnership arrangement has allowed the inclusion of the discards of fishes, crustaceans and marine mammals based on observer data. An outline of the database required to include and document 'adjustments' to official statistics is presented. This work will be extended to the entire North Atlantic region and beyond.

INTRODUCTION

This paper describes the methods employed in the collection, organisation and adjustment of fisheries catch and effort data used in the '*Sea Around Us Project*'. We will elaborate on our general approach using examples from Canada's Department of Fisheries and Oceans (DFO) and the Northwest Atlantic Fisheries Organisation (NAFO), and the International Commission for the Exploration of the Sea (ICES).

Many countries, particularly those fishing in the North Atlantic, have an excellent record of collecting and reporting fisheries statistics. There have, of course, been inevitable shifts in the format and content of these reports over the long time that seafaring nations have been reporting. These changes have caused numerous problems for the interpretation and analysis of this valuable information. Impacts have affected: species aggregation (typically with commercial species groups now being separated into species), species identification, the units of measure, definition of statistical areas (or their replacement with new systems), degree of coverage of the data collection system and other important measures. Commonly fishing effort data suffer more, as statistical systems evolve with developments in vessels, gear, and fishing practices. Fishing tactics and techniques change over time with targets and fishing areas, which means that 'days at sea' may have different interpretations. Some important measures are difficult to obtain.

Above all, the purpose for which the statistics were collected has changed in many cases. Initially to show the value and development of fisheries, these statistics are now used to manage and maintain stocks. This shift in objectives has caused many distortions, but fundamentally the change is, that before we wanted to know what was landed, now we want to know what was killed. That is, we require the inclusion of all sources of fishing mortality needed to assess the resource, especially if this is to be done in some ecosystem-based way[1]. Patterns of discarding (often altered by quotas or market factors), as well as unreported, misreported and unmandated catches are now important. The spatial distributions of fish stocks and fishing have become important as our understanding of the fishery and the biological processes has developed. These distributions have become valuable as we seek to manage individual fish stocks independently. Our ability to discern spatial patterns is limited by the spatial resolution of catch data and this has varied over time. Quotas, other management measures and international arrangements can all influence the spatial distribution of fishing, regardless of the underlying distribution of the biological stock.

The historical statistics from most countries have focused on the major commercial fisheries, and have largely ignored small scale, artisanal and

[1] Especially if the impacts of fisheries on marine ecosystems are to be evaluated. For example, total fish extractions are required as input to an ECOPATH model.

recreational fisheries. These fisheries are sometimes under jurisdictions other than the traditional national/international reporting bodies. Local knowledge is valuable in obtaining and interpreting statistics on these fisheries sectors and will be vital in reconstructing an historical record for them. The statistics have also frequently ignored or combined the information on catches of less valuable or less abundant species. Again, local knowledge may contribute to enhancing our knowledge of catches from these secondary species

The challenge is to apply informed procedures to improve and unify these historical statistics. Substituting a good guess as a default value for an element of the catch is likely more accurate than assuming by default that it was zero. Documentation of procedures for data adjustment, the basis for the estimates and the authority of the advice used in making the adjustments is essential. The process must be transparent, allowing identification of the reported data and the adjustments made to arrive at the final figures. Only with a transparent and fully documented procedure will it be possible for the agencies that are the primary owners of the data to assess and provide input on the magnitude and quality of the adjusted figures. Many times these agencies have considerable knowledge about discarding, misreporting and other sources of differences between the total catch and the landed catch.

The current uses of fisheries catch and effort data are many and varied. We are attempting to place the data from the North Atlantic in a system that is extendible to the fisheries of the world. As such, our choice of coding and structures has reflected a desire to incorporate fisheries from the extremely small scale to the largest of the factory ships. We are seeking to facilitate analysis of the energy consumption versus production in the fisheries (see Tyedmers, 2000) and this has demanded an approach to reporting fishing effort that is widely applicable (horsepower-days) and based on data that are widely available. Users of catch data will want to reconstruct historical and spatial patterns of exploitation, and in the case of modelling, fishing mortalities. For these purposes, we want to know all sources of fishing mortality (including those not reported in official landings), in the location in which they occurred. The spatial scale of data required for these models varies, but we will strive to produce statistics at the smallest practical scale to allow them to be integrated with larger scale ecological/oceanographic processes (see Pauly et al., 2000).

Our terminology for the various data resources we will discuss is as follows. A dataset refers to the data holdings of a given agency and may include a number of databases. For example, the DFO dataset includes a catch/effort database, an observer database, survey database and many more. A database is a single, coherent collection of data records that will usually be stored in several tables with relational links. Within a database all records will share common coding schemes, units and other standards.

Starting with the global dataset from the Food and Agriculture Organisation (FAO) for world fisheries landings, we have merged in other major datasets that have a geographically narrower scope and finer resolution such as those from DFO, NAFO and ICES. In addition to the greater detail and resolution, these often provide information on fishing effort. These datasets may also indicate different values for the landings data.

To this composite of databases we must make 'informed' adjustments and additions. These consist of additional data such as estimates of discards and other unreported catches. Justifications for these adjustments to the 'official' statistics come in a variety of forms. In the best cases, these adjustments will come from reliable and documented sources such as observer programs, but which are not included in official landings. In other cases, they may arise from a general discarding rate estimated for a specific fishery, or from estimates of illegal catches from industry or government sources. In some cases, such as ICES stock assessments, these additional sources of fish mortality have been compiled and are used in the stock assessments but are not available in official statistics. In many of these arrangements, the statistics supplied by member states cannot be officially altered even when they stretch the bounds of credibility. In all these cases it is our intention to make the appropriate adjustment, and to credit the source and document the methods used.

Data types

Fisheries data sources contain information of different types, including estimates of landings, measures of effort and a variety of classifiers describing the effort, such as the gear used, the area fished and others. In addition, some data sources attach estimates of economic value or price to the estimates of catch. Integration of data from the various sources, and subsequent adjustments, depends on the standardisation of measures and definitions for all the data types and sources.

Catch and Landings

The most important and fundamental information about fisheries for management purposes is the total catch (Gulland, 1983; Pauly, 1998). Catch is usually classified by species, area, fishing gear used, and other factors. What is officially reported as 'catch' should be nominal catches (the live weight equivalent of the landings). Data on the weight or numbers of animals that were taken but later discarded, even if collected, are not included as these statistics that were designed to describe the contribution of fisheries to the food supply and national economies. The reported catches may be the result of a census of fishing vessel landings, survey sampling, reporting by fishers, or estimated by proxies such as fishing effort. For this paper the reported landings, are nominal catches, and are treated in metric tonnes (t) of live weight (mass) equivalents.

Catch statistics are important for three reasons (1) the gathering of statistics increases knowledge of the fishery (tracking of vessels engaged in fishing, dockside sampling of these same vessels, etc.), (2) total catches determine the scale of the fisheries, both within and between sectors, in terms of their production and value; and (3) examining time series of catches allows for first-order assessment of fisheries, and of the status of the species and populations (stocks) upon which the fisheries depend (see Grainger and Garcia, 1996). Finally, assessments of fisheries and their impacts on fish stocks and the environment have evolved to include other sources of information. However, basic catch statistics are still essential to the process (see Alder et al., 2000).

Fisheries catches may be separated into three components: (1) nominal catches, reported to (and by) a monitoring agency (e.g. by member countries to FAO), (2) discarded bycatch, the non-targeted part of a catch, often consisting of the juveniles of targeted or other species, caught due to the unselective nature of the gear used, and usually thrown overboard rather than landed; and (3) an unreported component, consisting of categories not covered by the reporting system in question (examples may include sport fisheries, artisanal fisheries, or illegal catches). Thus, this last group may be composed of catches that a given agency is not mandated to gather and report ('unmandated catches'), and of catches that are misreported by fishers or others. A major task of our current work is to estimate unmandated and misreported catches, with both requiring the development of new protocols (see Pitcher and Watson, 2000).

Each fishery statistical system we deal with has evolved a set of procedures and conversion factors for reconstituting the original weight of fish landed in a wide range of product forms. The conversion factors (e.g. COFREPECHE, 1996) that are used in each agency's statistical processing are not explicitly considered in our adjustments. It is obvious, however, that if inappropriate conversion factors were used by the agencies providing the catch data, this would lead to significant errors in the live weight equivalents (e.g. converting lobster tails to whole body weights). Note that under quota management systems there may be a tendency for industry to seek adjustment of conversion factors (downward) in circumstances where live weight is being over-estimated. There is, however, no incentive for them to seek any adjustment in the case where live weight equivalents are being under-estimated.

Value and Price

Much of the original incentive for governments to systematically monitor fisheries was to determine the value and economic development of the fishing industry. In some national systems (e.g. Canada), the estimated value of catches (or equivalently, average price) is recorded with the catch data. In other systems (e.g. FAO), economic statistics are generated and reported independently of the catch data.

Effort

As with economic information, fishing effort may be measured and classified, by area, gear, etc, in the same process that records the catch data, or estimated independently. In either case, the effort must be matched to the corresponding catches within the basic statistical system.

The definition of fishing effort, unlike catch, is dependent on the nature of the fishing unit (e.g. boat, trap) and the amount of resources expended by that unit. The specific effort resources expended are routinely measured in units of time and/or amount of gear used but alternative definitions abound. Our work will use three units of effort. The conventional units 'days fished' and 'days at sea' will be compiled directly from the statistical sources. An alternative unit of 'horsepower-days' will be the product of the numbers of days at sea times the average horsepower of the vessels in the given block of effort.

Gear specific effort units, such as hours trawled or thousands of hook-hours seem to offer an apparent finer resolution of effort but they are not used here. Although such detail is available for many of the North Atlantic fisheries, this is not the case for many, or possibly most, fisheries in other parts of the world. Where it is available, the accuracy of gear-specific effort measures has been challenged for many reasons and often by the fishers themselves. Fishers have claimed they had falsified the original logbook data to appear to comply with management restrictions. Regardless of such a concern, the numbers of days fished is a statistic relatively easy to obtain and difficult to falsify. Finally, gear-specific effort units are also difficult or impossible to aggregate across gear types, e.g. relating total number of trawl-hours to hook-hours. On the other hand, horsepower-days offers a comparatively robust measure that can be compared across most fisheries of the world, even those where no vessels are involved.

PRIMARY DATA SOURCES

This methodology review paper deals with data from four primary sources: FAO, DFO, NAFO, and ICES. Each has its own strengths and weaknesses. In addition, information from other datasets, such as those from the U.S. government, the tuna commissions, etc. will be included in our project database. These data will be augmented by smaller, tightly focused datasets, prepared by consultants for a range of European inshore, small-scale and recreational fisheries.

FAO

Only one global database of fish catches presently exists from which inferences can be made that pertain to entire ocean regions: the database assembled by the FAO from reports supplied by member countries (FAO, 1980). This database consists mainly of annually updated catch time series, by countries and regions, for the year 1950 to the present. The quality of the data therein is highly variable, and ranges from accurate data on a single species basis for some countries to crude and over-aggregated estimates for others. Moreover, the catches are not assigned to the Exclusive Economic Zones (EEZ) of the countries for which they originated, but to the large FAO statistical areas for which it is reported.

Few scientists outside the FAO have made use of these statistics to draw inferences on fish stock status over large areas of the world's oceans (but see Alverson and Dunlop, 1998 for exceptions), but content themselves with citing assessments made by FAO staff. There has been little independent validation of this database against original or other data sources. Perhaps there is little criticism and crosschecking because so many countries and institutions contribute to this dataset that has engendered a strong sense of ownership. Nevertheless, its weaknesses in the face of current needs are understood. The FAO Advisory Council on Fisheries Research has admitted; "the current statistics collection system is limited primarily to landings and commodity statistics, whereas there is a critical need for data relevant to fleet capacity, participation in fisheries, economic performance and distribution" (Anon 1997). There have been calls for the FAO reporting areas and species groupings to be changed to reflect current fisheries practises, which would facilitate analysis of the economic efficiencies of fisheries (Pontecorvo, 1988). Such changes would probably facilitate improved biological analysis as well.

Canada Department of Fisheries and Oceans (DFO)

This dataset includes records of Canadian (that is Canadian vessels only) commercial catch and effort per species (marine finfish, invertebrates and plants) for Eastern Canada for years 1986-1998 broken down by spatial statistical regions called 'unit areas'. This dataset is obtained in the Zonal Interchange File Format (ZIFF) and has been compiled within the Atlantic Canadian fisheries regions to ensure consistency of coding and units from the four different statistical offices that operate in the zone. It includes date, target species, unit area, tonnage for each species landed, vessel characteristics (tonnage, tonnage class, length, horsepower) and gear. Records may not include complete vessel characteristics, and therefore, horsepower or tonnage may be missing. The *Sea Around Us Project* will aggregate the DFO data to the level of effort by month, unit area, tonnage class, fishery type and gear type. The catch is further classified by species. The DFO catch data includes all fishery catches with the exception of recreational catches (generally considered small with respect to the commercial fisheries). There are several small-scale fisheries that either have not collected effort data or have only begun to do so recently i.e., in the 1990's.

Northwest Atlantic Fisheries Organisation (NAFO) data

The NAFO dataset includes monthly catch (marine finfish and invertebrates) and effort by divisions only (which may comprise several unit areas in the DFO system) for Canadian and foreign vessels. The data is structured by fishing country, vessel and gear types, and species targeted. The information gathered by NAFO is a compilation of the catch and effort as declared by each member country. NAFO and its predecessor the International Commission for Northwest Atlantic Fisheries (ICNAF) provide a consistent statistical data series since 1960. In order to prevent duplication of records with the DFO dataset we have removed records of catch/effort by Canadian vessels after 1985. This dataset does not include any information for vessel horsepower. A significant number of foreign vessels have been recorded at one time or another in the DFO records of vessel characteristics, including horsepower, and these records will be used to estimate horsepower from the tonnage and other characteristics available in the NAFO vessel records.

International Commission for the Exploration of the Sea (ICES) North Sea data:

ICES data for the North Sea comes from two sources. Electronic data sets exist for all ICES areas (including the North Sea) landings back to 1973. These data are broken down by statistical area and reporting country for the major species. Data provided to us did not include fishing effort or vessel descriptions. There is no official electronic dataset of landings prior to 1973; we therefore used the records provided in ICES' *Bulletin Statistique des Pêches Maritimes (des pays du nord de l'Europe*) to enter landings for the North Sea from 1903 until 1974. From this written record we also extracted what exists of fishing effort records including breakdowns by tonnage class, and more rarely by vessel horsepower.

Consistency of Data Sources

Consistency comparisons between NAFO and DFO datasets will be made, as will DFO, NAFO and ICES with FAO. This will help to determine if the national and international reporting systems are treating data consistently and completely. Comparisons will be limited to the large-scale aggregates used in the FAO dataset. However discrepancies can be investigated with the more detailed data available in the other sources.

Consultations

Official catch and effort statistics are available for most areas of the world. An aggregated set of this data is usually provided to the FAO for inclusion in their global dataset. In order to provide complete details of fish effort and fishing fleet composition, it is usually necessary to access national databases directly. In the case of European Union countries, these data is compiled across member states and are available on the internet.

Obtaining records of small-scale (typically inshore), artisanal, and recreational fishery catch and effort statistics is more problematic. This usually requires either a co-operative or consultative arrangement with some agency/individuals within the country in question. Our project has engaged consultants to report on the inshore, small-scale and recreational catch for the majority of maritime nations in the North Atlantic region. At present we have consultants working on fisheries in Iceland, the U.K., the Irish Sea, Denmark, Norway, France, the Netherlands, Germany, Spain, Portugal and Morocco. Plans exist to extend these efforts soon to Belgium, Russia, and the Azores and Faeroe Islands.

Our co-operative arrangement with the DFO has allowed estimates of discarding to be made based on their observer program. These valuable collaborative arrangements, however, are rare. Alverson et al. (1994) provides a range of discarding for major fisheries that can be applied where appropriate. Such extrapolations from similar fisheries (with respect to gear and target species) must be carefully applied. However, these estimates, untested as they are in most cases, most likely yield a fairer interpretation of total mortalities than ignoring discards where they are known to occur.

Illegal catches are probably the most difficult information to obtain, as they are seldom discussed in official statistics (Creed 1996). Some attempts to make allowances for 'misreported' catches through modelling look promising (Patterson, 1998). Usually these catches can be inferred from other fisheries. Typically, however, interviews must be conducted with informed sources within the fishery or monitoring agency. Personal networks are invaluable for this. A generalized approach based on historical changes in fisheries management or other factors affecting

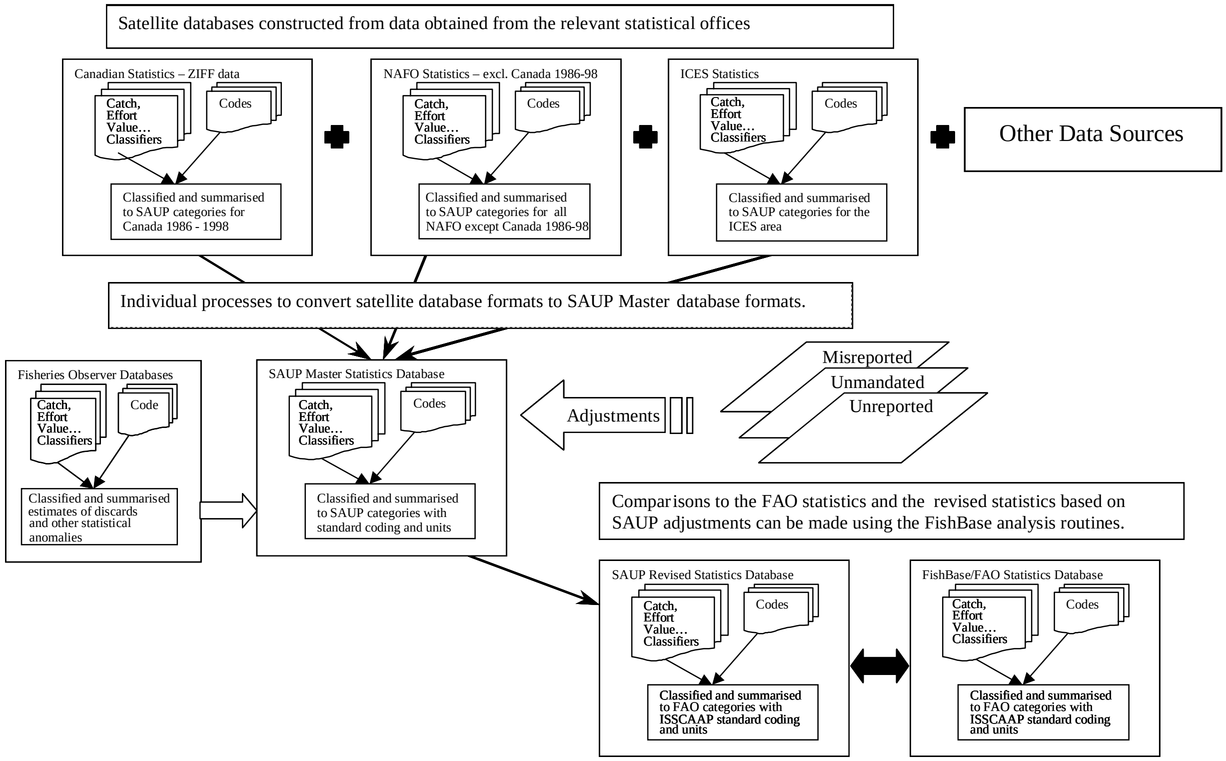


Figure 1. Overall data acquisition, data processing and data management supporting the *Sea Around Us Project's* reconstruction of actual fish catches from the North Atlantic.

incentives
to cheat can be informative here (Pitcher and Watson, 2000). Though many official statistics may be difficult to access the trend is for this to change. 'Freedom of information' acts have removed legal impediments, and improved information technology has contributed to widespread and simplified access in several countries including Canada and the U.S. Nevertheless the work of key individuals in each country being reviewed is invaluable. It provides a means of contacting local artisanal and recreational fisher groups who can be very co-operative if properly approached. It provides a means of accessing available port records, some of which have impressive historical spans. Most importantly, having a person within the country allows fishers and government officials to be interviewed 'off the record' so that estimates of illegal fishing, discarding and other vital 'unofficial' statistics can be elucidated.

DATABASE REQUIREMENTS

Design of the project's principal, i.e. 'Master', database was constrained by several imperatives. Unlike a conventional database developed by a government department or a business we could not scale our resources (money, personal etc.) with the scope of our coverage. Indeed some of our imperatives required a smaller, less-commercial approach (such as the choice of MS Access® as our database).

The first imperative was the strong desire to provide an output of summarised catch data compatible with the 15,000 species of marine fishes included in FishBase (Froese and Pauly 2000; www.fishbase.org). This would allow the wealth of descriptive data (taxonomic, life history, occurrence etc) and the significant investment in analytical procedures (trophic level comparisons etc.) to be utilised. Likewise we also wished to maintain, as far as possible, compatibility with the FAO global dataset including its ISSCAAP species codes. Updates from FAO will be valuable to our future work, as no other agency has a mandate or the resources to produce a truly global dataset.

The second imperative was for the database to allow allocation of catch and effort to spatial strata representing functional ecological entities such as large marine ecosystems (see Pauly et al. 2000). Meta-analysis of spatial data would certainly require the use of geographical information systems. Our database must facilitate the use of the data by experts developing models

of marine ecosystems.
Thirdly, we wanted to maintain a system of 'satellite' databases (Figure 1) recording the best estimates of catch and effort as supplied by the source agencies. Each satellite database will retain the codes, units and standards of its source agency, but the records will be processed to the *Sea Around Us Project* codes, units and standards into our master database. Thus each original satellite database record will be associated with a Master database record where all the subsequent adjustments and additions can be made in a rigorously documented manner. In this way an 'audit trail' will exist to link incoming data to our final estimates. It should be noted that the resolution of the data in the satellite databases (spatial, temporal, effort, gear, etc.) may vary, however, all will be processed to the standard resolution of the project master database as they are loaded.

The fourth and final imperative was that as rapidly as possible the information would be available to all, preferably on the Internet and with a map-based interface. In this way it would be used/improved by experts, and contribute to debates on the state of marine systems and marine policy in general.

DATABASE STRUCTURE

Catch

Species Codes
The database utilises the ISSCAAP codes used by FAO, but will allow synonymy with other coding systems. The codes, broadly compatible with the fish classification in FishBase (itself based on that of the California Academy of Science), will allow identification at a variety of taxonomic levels and allow processed products to be differentiated from whole products.

Catch Value
Catch values, based on average prices not corrected for inflation, for three broad periods (1950s, 1970s and 1990s), as well as their major markets (Sumaila et al. 2000), will be included for each taxonomic group to allow estimation of catch values.

Fishing Effort

The method of describing fisheries and fisheries effort used in this project was designed to be extendible to other fisheries around the world, some of which are very different from those dealt with in the North Atlantic. The 'taxonomic' approach allows any fishery to be characterized by its basic gear type, its location, the tonnage class of vessels used (if any), and the major target species. This system draws upon the descriptions of world fishing gear by Brandt (1984). Those fisheries that can be confused by two of these descriptors can be separated by the third. We plan to further characterize these fishing effort groupings by the average 'catching power', that is the amount of the target species typically landed for each fishing day or day at sea (when abundance is high) with the usual number and configuration of gear units (hooks, nets, whatever) employed. This will facilitate comparison of small and large-scale fisheries (see Ruttan et al., 2000).

Time Periods
Though some data sets provided to us, such as that from DFO Canada, contain detailed fishing effort aggregated to month, we have further aggregated these records to annual records as we are primarily interested in examining changes over longer periods. The original monthly information will assist in studies of the seasonal aspects of these fisheries, and allow us to formulate a more precise spatial allocation of catches and fishing effort. Some of the Canadian data is available by fishing gear set by the date of the fishing activity rather than the date of landing.

Fishing Areas
As with catch data, it is important to be able to aggregate fishing effort into spatial definitions such as large marine ecosystems (see Pauly et al., 2000) that we believe to be the correct scale to examine the impact of management changes. Data from DFO Canada was provided by 'unit area', these are smaller areas that nest within NAFO statistical areas. ICES data were broken down into ICES statistical areas. Where possible and appropriate, expert consultation will be used to determine appropriate rules to allocate catch and effort to the smaller units which will facilitate their re-aggregation into units of ecological or management significance (for example Large Marine Ecosystems).

Gear type
Although all the statistical systems record a wide variety of fishing gears, we have grouped them into a much smaller number (see list below) which ignores the details of gear construction but is based on the primary mode of fish capture or gear operation. For example, hand line and longline are in the same category because their efficiencies do not depend upon a particular boat

Table 1. Fishery types for the North Atlantic.

Groundfish	Demersals e.g. cod, flounders, redfish
Small Pelagics	herring, mackerel
Large Pelagics	tuna, swordfish
Sharks and Skates	Porbeagle, dogfish
Freshwater or Diadromous	alewife, smelt, eels
Bivalves	clams, quahaugs
Scallops	
Squid	
Lobster	
Shrimp	
Crab	
Miscellaneous	Seaweeds, lumpfish

size. Harpoon and spear are quite rare in this data set and include sealing and swordfish. The dredge group includes both hand-held and mechanical devices because the hand-held one is rare, and each of them will be used with distinctive vessel sizes.

Gear types are:

- bottom trawls,
- midwater trawls,
- mobile seines,
- surrounding nets,
- gillnets and entangling nets,
- hooks and lines, trap and lift nets,
- dredges,
- grappling/wounding, harpoons and spears, and
- other gear.

Fishery Types

Although approximately 50 species account for 95% of the nominal catch reported to FAO from the North Atlantic since 1950, there are still many more species that are caught but not landed or reported. In most catch statistics, the target species is in fact the more abundant species on a trip by trip basis. This is sometimes called the 'main species caught'. There are exceptions, however, such as tropical shrimp fisheries, which often take many times the quantity of small demersal fin fishes than shrimp target species. The number of target species for the fisheries of the world is potentially a very long list. Thus, it is more useful to group the target species into broader fishery types that reflect the choices that fishers are really making: fishing for groundfish, small pelagics, squids, etc. (Table 1).

With these categories, the assignment of effort to fishery types is less subject to interpretation than the assignment to species sought or even 'main species caught' groupings. The fishery types defined here serve as links to observer data that will provide a minimum estimate of discards produced by each fishery.

Vessel Size

The most widely available descriptor of vessel size is its overall length. Unfortunately, trends in vessel design, at least for the North Atlantic, have resulted in large increases in the displacement of vessels within regulated length groups. As a result, the relative fishing power over time is best described by tonnage. There are long standing tonnage classes (Table 2) in use on both sides of the Atlantic and they are used here. Where necessary, we will convert from vessel length to vessel tonnage.

Table 2. North Atlantic vessel tonnage classes.

Not known	(for Canada most are 0-24.9)
0-49.9	
0-24.9 25-49.9	This split used in Canada only
50-149.9 150-499.9 500-999.9	
1000-1999.9	
2000 or greater	

Adjustments

Adjustments made to catch and effort data (as imported from 'satellite' databases) will be documented on a species-time-area-gear basis so that changes in values can be review and updated.

CASE STUDY: CANADIAN NORTH ATLANTIC (DFO AND NAFO DATA)

Our approach can be illustrated by the process of reconstructing the catch and effort for the North Atlantic region under the jurisdiction of Canada's DFO and NAFO. This case demonstrates what is possible under co-operative arrangements in 'data-rich' fisheries. In other circumstances where the unaggregated data are not available, approaches based on more general considerations, e.g. average rates of discarding for aggregated areas or times (Alverson et al., 1994), are necessary. Even in circumstances where conventional datasets are complete and well maintained, the difficult task of estimating totals for discards, misreported, and unreported catches may call for a different approach (Pitcher and Watson, 2000).

Species Identification

Species codes and names were rendered uniform across three data sets: the DFO research, Canadian commercial catch, and NAFO. Coding inconsistencies were traced and corrected when possible. Because they are the direct links with the observer data, the DFO research species codes were kept. Nine categories were added to the research species list: marine plants, sub-products of already accounted for catches (e.g. seal and cod liver), and unconvertible products (Table 3). The unconvertible products category refers to products for which the yield of processed products varies significantly across area, fishers, and time, so as to make difficult the estimation of an accurate live weight. Marine mammals catches were not part of the fishery data and are described in a specific section (see Appendix 1).

Table 3. Categories of products unconvertible into live weight biomass.

Description	Examples
Sub-products of catches already accounted for elsewhere	seal and cod liver, seal oil
Unconvertible products	sea urchins roe
Marine plants	kelp, Irish moss, rockweed

Catch reconstruction

For the years 1986-1998, the Canadian catch was obtained from the 'zonal interchange files' (ZIF files) while the foreign catch came from NAFO data. The foreign catch is defined as the catch reported by vessels registered in other countries. For the years 1960-1985, all catches were obtained from the NAFO data set.

Catch was compiled by year, month, country, NAFO division, unit area, vessel size, gear type, fishery type, and species.

Not all aspects of marine harvest are covered equally by the DFO database. One component that is missing is the take of marine mammals, especially seals. A reconstruction of seal harvests is described in Appendix 1.

Effort Reconstruction

Canadian data 1986-1998

Effort is defined as the number of horsepower-days, that is, the number of days spent fishing (includes searching and fishing) or days at sea (includes fishing days plus travel time), structured by gear, year and month. The direct approach would normally be to match vessel characteristics to each fishing trip. However, because of frequent missing vessel characteristics, and because small Newfoundland vessels were not individually linked with their catches, several intermediate steps were necessary to generate estimates of horsepower.

Where missing, the vessels horsepower was replaced by an estimated value based on a linear regression using vessel length and tonnage. Vessels present in the database (actually fishing or not) were used if complete information for their length, tonnage and horsepower was available. A preliminary exploration of the data showed a skewed distribution for horsepower, warranting a fourth root transformation to stabilize the variance. The resulting linear predictor of transformed horsepower was

$$HP_l = 1.844 + 0.0379 * \text{length} - 0.0017 * \text{tonnage} \qquad \ldots 1)$$

Retransformation, accounting for the retransformation bias gives

$$HP = HP_l^{\,4} + 6\sigma^2 HP_l^{\,2} + 3\sigma^4 \qquad \ldots 2)$$

An alternative estimation of the horsepower using a generalised linear model and appropriate link may provide for improved precision without the problems of retransformation bias. A comparison of these two estimation approaches will be made for these data.

For each trip, horsepower was obtained by using the horsepower attributed to the vessel that reported that catch. Missing horsepower were replaced by the average value computed for each stratum (combination of year, month, tonnage class, gear type, and fishery type). The remaining missing values were replaced by averaging horsepower over progressively larger combinations of categories (blocks of effort) until all missing values were estimated (Table 4.)

Effort was then computed as the amount of horsepower multiplied by the number of fishing days spent. Because effort was often missing, a distinction is made between catch associated with and without effort so that total effort could be scaled from reported effort.

Effort adjustment for catch without effort

Total effort will be computed for each effort cell as the total catch for all species divided by the catch rate for all species. Each cell will have a unique factor and will be referenced in the database to our methods. Application will depend on the data source as some may have already applied effort prorating.

Table 4. Procedure used to estimate the average horsepower in each block of effort.

Descriptors used	Remaining missing values
Year, month, NAFO divisions, tonnage class, gear type, fishery	47,127 (41%)
Year, NAFO divisions, tonnage class, gear type, fishery	9,736 (9%)
NAFO divisions, tonnage class, gear type, fishery	5,061 (4%)
NAFO divisions, tonnage class, gear type	674 (<1%)
Tonnage class, gear type	0

Discarding

Observer Programs

The use of at-sea observers is a widespread practice in large-scale fisheries. In the Northwest Atlantic there are observer programs operated by Canada, based in Nova Scotia, Newfoundland and the Gulf of St. Lawrence, and by NAFO. At-sea observers supplement the much more limited amount of surveillance conducted by the fisheries enforcement agencies. Observer data contains a voluminous amount of information but caution is required in the analysis. Observers are not deployed in a random manner, nor in fact is there usually a sampling design intended to minimise variance or control bias. Observers are often deployed in a 'tactical' manner, meaning the enforcement agency is concerned about a particular area or the fishing practices of a particular vessel, and send an observer in response. Observer data has been challenged over the years with accusations of corruption. However, very few of these have ever been substantiated. The greatest challenge in analysis of observer data is the effect that the presence alone of an observer has, or may have, on the fishing practices of a vessel, i.e., an observer effect. One expected observer effect would be for captains to not commit infractions of the regulations while carrying the observer.

Fisheries observers routinely spend a full trip on board vessels. They record the positions fished, the effort used and the composition and fate of the catch taken. The data is far more detailed than is possible to collect with a logbook and is independent of either willful or negligent inaccuracies on the part of the ship's Captain. The characteristics of the vessel and gear are recorded at the beginning of the trip and any gear modifications are recorded as the trip progresses. The effort is recorded by date, time and position (latitude/longitude), the amount of gear and duration fished, conditions of weather and sea during fishing and any gear damage or other events arising during fishing. The catch is observed and total catch for each species is estimated, including amounts kept and discarded. As many vessels operate 24-hours a day, the observers update their records from the logbook whenever they were off-duty.

Many species that never appear in the reported catch statistics are recorded by the observers, for example, on the eastern Scotian Shelf (Nova Scotia, Canada) in 1986 there were 125 species observed in the catch, while there are only 42 reported in the corresponding NAFO database. Some of these 42 include groups of species such as sharks that are routinely separated by observers but others, such as skates, are completely unaccounted for.

A description and application of the approach to catch adjustment using observer data is worked out for a particular block of data in the following sections. The data is from 1986 and covers the groundfish fishery on the eastern part of the Scotian Shelf in NAFO divisions 4V and 4W (Figure 2). The catch statistics are obtained from the NAFO databases maintained by the Canadian Department of Fisheries and Oceans (DFO). These data have been included in The Sea Around Us database as described above. The observer data comes from the DFO observer database maintained at the Bedford Institute of Oceanography by Marine Fish Division, Maritimes Region of DFO. Most of the analysis was completed using database queries in ORACLE although the same results could be obtained by various other means. The block of data was selected for this example because it is data-rich and allows a good demonstration of the methods. It is acknowledged that many other fisheries, areas and times are not as well covered, and adjustments to such catches will be based on less data and broader application of the mean catches from other places and times.

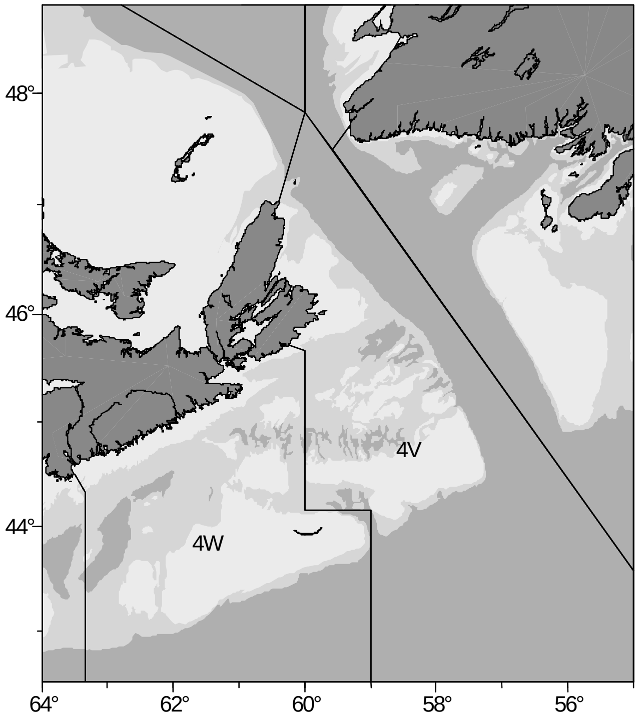


Figure 2. Area of discard estimation (NAFO's Divisions 4V and 4W) as outlined on the eastern portion of the Scotian Shelf off Nova Scotia. Canada.

A further point regarding the example block of data is that during 1986 it was <u>not</u> illegal to discard fish, in any quantity and of any species. For this reason, there was less likely to be an observer effect limiting this behaviour. It has been reported that fishing captains were still reluctant to discard excessively when observers were present, even in the absence of any specific regulation against it. Consequently, an observer effect, inhibiting discarding practices, cannot be ruled out and so the estimated discards, even for this period of time, must be considered minimum estimates, especially for target, high-valued species.

Estimation of Observer Coverage Proportions

All results from at-sea observers must be interpreted carefully in light of variable and often low coverage levels for certain fisheries. We have defined coverage rate to be the proportion of the reported landings for a given unit of data, i.e., country, area, month combination, which was observed as retained catch by the observers. The proportions reflect the total amount of retained catch of all species by weight, observed at sea with respect to the total amount of landings reported for the corresponding country, area and month. Proportions greater than 1.0 reflect observed catches in excess of the total reported landings. Thus, observer coverage proportion, $O_{c,a,m}$, on a catch basis is:

$$O_{c,a,m} = \Sigma_s \, kept_{c,a,f,y,m,s} \; / \; \Sigma_s \, C_{c,a,m,s} \qquad \text{...3)}$$

where $kept_{c,a,f,y,m,s}$ is the total observed landings and $C_{c,a,m,s}$ is the nominal catch (landings) in the NAFO data.

Table 5. Coverage proportions by Canadian fisheries at-sea observers during 1986, for the groundfish trawler fishery in NAFO 4VW in 1986. Empty cells have no observer coverage, 'no catch' indicate that fishing was observed but no catch was reported to NAFO, grayed cells mean that there was neither reported nor observed catch. Figures represent the ratio of a nation's observed catch to that nominal catch reported by that nation for the same month and statistical. Values exceeding 1.0 indicate that the observed catch exceeded the catch subsequently reported to NAFO.

	Canada		Cuba		France	Japan		USSR	
Month	4V	4W	4V	4W	4V	4V	4W	4V	4W
1	0.07				1.34				
2	0.09	0.28			0.10				
3	0.14	0.00			0.00	1.40	no catch		
4	0.02	0.01		0.47	0.77				
5	0.18	0.06		0.58	1.13				0.51
6			no catch	0.06		1.58		no catch	0.42
7	0.02	0.02	no catch	1.00		1.94		no catch	0.23
8	0.09	0.00		0.53		1.68	no catch		
9	0.14	0.01							
10	0.08	0.06				1.65			
11	0.06	0.00							
12		0.00							

Table 6. Estimates of discards (tonnes) by month and species in the groundfish trawler fishery in NAFO area 4VW in 1986. (* refer to unspecified species).

Common Name	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Annual Total
Alewife		0.05		0.12	0.00	0.00							0.17
American plaice	45.92	9.63	103.54	0.10	14.61	1.80	3.60	26.01	12.17	10.22	0.00		227.59
Argentine			0.01	0.06	0.14	1.83	0.02	7.57	0.00	0.07			9.70
Bigeye tuna										0.00	0.00		0.00
Bluefin tuna							1.28		0.00	0.00	0.00		1.28
Cetaceans						3.69							3.69
Cod	68.70	234.49	281.96	0.01	55.72	1.20	170.69	292.89	39.22	50.66	0.17	0.00	1195.72
Crustaceans	1.80	0.41	0.47	0.44	6.25	15.94	5.79	2.49	0.32	0.19			34.10
Cusk	0.00	2.08	0.00	0.15	0.11	0.51	0.00	2.60	0.00	0.18	0.00	0.00	5.63
Dogfish*	27.16	164.92	1608.20	462.03	476.99	33.03	2.56	23.36	0.40	0.67			2799.33
Flounder*	0.00	0.29	0.97	0.00	0.37	0.00	0.00	0.00	0.00	0.00			1.63
Grenadier								0.00					0.00
Groundfish	0.00		0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00		0.00
Haddock	48.10	109.63	93.55	0.00	29.06	2.89	50.66	162.14	30.96	89.54	38.00	0.00	654.53
Hake*			0.01	0.00	0.30	0.00	0.00	0.00	2.22	0.06			2.60
Halibut	0.00	0.25	0.02	0.00	0.00	0.16	0.00	12.60	0.00	0.00	0.00	0.00	13.03
Herring	0.00	0.17		0.00	0.67	0.48	0.00	2.53	0.01	11.69	8.00		23.54
Mackerel		0.04		0.00	0.09	7.89	9.21	5.36	0.04	0.03	0.00	0.00	22.65
Monkfish	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Other fish sp.	10.84	8.94	1.32	2.64	34.53	11.46	73.77	103.87	13.03	38.45	2.69		301.55
Other inverts	0.12	0.01	0.40	0.00	2.91	30.20	0.06	759.98	0.16	4.11			797.94
Pollock	11.26	115.62	64.99	0.11	61.17	0.94	60.60	19.64	4.93	6.25	0.00	0.00	345.51
Porbeagle		0.89		0.65	1.06	0.12				1.56			4.29
Red hake	0.00	75.15	4.58	0.71	1.02	5.54	0.01	3.58	0.02	0.11			90.72
Redfish	7.73	3.52	13.16	0.00	6.01	0.40	1.62	85.94	1.59	9.69	0.35	0.00	130.00
Seals*	0.11	0.32	4.48		0.79	4.77	0.32						10.78
Sharks*				0.04	12.33	1.19	2.98	1.79	0.37	26.66			45.36
Silver hake	5.23	17.09	1.52	32.68	47.85	200.94	11.10	11.60	1.90	81.18	0.00	0.00	411.10
Skates*	174.03	383.03	501.45	327.25	266.96	6.22	414.42	224.60	244.19	313.38	86.29		2941.81
Swordfish					0.00	0.48	0.00	0.18	0.00	0.00	0.00		0.66
Turbot		0.00	0.00		0.00	0.00	0.00	0.04	0.00	0.32			0.36
White hake	0.01	0.93	0.34	8.69	0.89	5.99	1.52	45.73	2.33	10.45	0.00	0.00	76.88
Winter flounder							0.00			0.00			0.00
Witch flounder	14.96	1.03	54.18	0.02	0.41	0.03	0.04	0.62	0.39	0.03	0.00	0.00	71.72
Wolffish*	0.98	1.46	0.54	0.01	1.01	0.10	0.05	27.72	1.47	3.64	0.00	0.00	36.98
Yellowtail fl.	2.11	0.02	0.14	0.00	10.46	1.19	90.21	77.68	55.62	71.23	0.00	0.00	308.67
Totals	419.08	1129.97	2735.83	835.72	1031.69	338.99	900.50	1900.51	411.35	730.38	135.50	0.00	10569.30

* unspecified

Table 5 highlights several of the problems inherent in this approach. Cells highlighted in grey contained neither reported catch nor observed catch, so do not represent a problem. Empty white cells correspond to catches with no observer coverage at all. Cells indicated as 'no catch' correspond to instances of observers reporting fishing, with retained catches, from times and areas but for which the country in question did not report any catch at all. In the six cases below, it is likely that the vessels were in part fishing in adjoining areas, i.e., 4V or 4W, and that their catches were reported as such. The other problem revealed is the occurrence of cases in which the observed catch exceeds the reported catch.

Estimation of discard catch rates

In this example, the observer data (Table 6) and the corresponding reported catches for the groundfish trawler fishery in NAFO 4VW in 1986 was the compiled weight of both kept and discarded catch by month (m), country (c), NAFO area (a), and species caught (s). Estimates of discard rates, $d_{c,a,m,s}$, were obtained by linking the observer program data and the reported catch for each block of effort,

$$d_{c,a,m,s} = disc_{c,a,m,s} \,/\, \Sigma_s \, kept_{c,a,m,s} \qquad \text{...4)}$$

where $disc_{c,a,m,f,y,s}$ is the total observed discards for species s and $kept_{c,a,m,s}$ is the observed landings. The estimated discards, $D_{c,a,m,s}$, are computed as:

$$D_{c,a,m,s} = d_{c,a,m,s} * \Sigma_s \, C_{c,a,m,s} \qquad \text{...5)}$$

Table 7. Summary of example results of observer-based estimates of total discards (tonnes) for all species combined, for the groundfish trawler fishery in NAFO 4VW during 1986.

Month	Canada		Cuba		France	Japan		USSR		Monthly Totals
	4V	4W	4V	4W	4V	4V	4W	4V	4W	
1	400	0			19					419
2	852	212			67					1130
3	1090	1632			0	14	0			2736
4	330	0		505	0	0	0			836
5	411	31		311	1				277	1032
6	0	0	0	243		5		0	92	340
7	868	42	0	0		4		0	8	921
8	713	1149		18		21	0			1901
9	387	24		0		0	0			411
10	598	128				5				731
11	69	66				0	0			135
12	0	0				0				0
Totals	5717	3284	0	1078	87	49	0	0	377	10592

where $C_{c,a,m,s}$ is the reported catch in the NAFO data. The individual estimates of discarded catch are summarised by species and month in Table 6, by month and country in Table 7 and totalled by species (Table 8) below. The difference between the total in Table 7 (10592) the other two (10569) occurred because a small amount of catch (23 t) had no species identity assigned to it. This is often accounted for in a category called NEI (not elsewhere included).

The analysis presented here will be extended to better estimate discards from cells without observer estimates through application of generalized linear modelling. One specific approach may be a logistic model for the rates. However, other alternatives will also be investigated. This approach opens the way to using the EM algorithm for filling in missing cells. Interviews of fishers participating in various fisheries provide a semi-quantitative means of estimating discard. However, these are usually specific to particular times and areas, and great care must be exercised when applying them to large aggregates of data. For this reason, data from this source should be given greater weight as a means of setting ‘anchor points’ in the more qualitative discard estimation of Pitcher and Watson (2000).

Illegal Catches

Enforcement and surveillance program

Estimates of illegal catches taken by both foreign and domestic vessels could potentially be estimated from fishing vessel surveillance data (DFO Conservation and Protection Branches in the Atlantic regions). Their data is confidential and considered sensitive but if kept anonymous, it may be possible to use and analyse their data to

Table 8. Comparison of estimated discards and reported landings for the groundfish trawler fishery in NAFO area 4VW for 1986, all countries combined. The table is ordered in descending order of the proportion (percent) of discards in the total catch (tonnes).

Common Name	Catch	Discard	Percent
Grenadier	1	0.0	0.0
Bigeye tuna	10	0.0	0.0
Groundfish (unspec)	76	0.0	0.0
Monkfish	2081	0.0	0.0
Swordfish	231	0.7	0.3
Silver hake	82466	411.1	0.5
Mackerel	2202	22.7	1.0
Halibut	1132	13.0	1.1
Cod	79084	1195.7	1.5
Redfish	7621	130.0	1.7
Cusk	326	5.6	1.7
Pollock	19296	345.5	1.8
Flounder (unspec)	68	1.6	2.3
Witch flounder	2382	71.7	2.9
Turbot	12	0.4	2.9
White hake (Squirrel)	2341	76.9	3.2
Haddock	16384	654.5	3.8
Wolffish (unspec)	309	37.0	10.7
Bluefin tuna	9	1.3	12.4
Red hake (Squirrel)	257	90.7	26.1
Yellowtail flounder	692	308.7	30.8
Alewife	0	0.2	100.0
Hake (unspe	0	2.6	100.0
Cetaceans (unspec)	0	3.7	100.0
Porbeagle	0	4.3	100.0
Argentine	0	9.7	100.0
Seals (unspec)	0	10.8	100.0
Herring	0	23.5	100.0
Crustaceans (unspec)	0	34.1	100.0
Sharks (unspec)	0	45.4	100.0
American plaice	0	227.6	100.0
Other fish species	0	301.6	100.0
Other invertebrates	0	797.9	100.0
Dogfish (unspec)	0	2799.3	100.0
Skates (unspec)	0	2941.8	100.0
Total/Average	216980	10569.5	4.6

obtain an estimate of illegal catches including areas outside the 200 miles limit (but see Pitcher and Watson, 2000). Such estimates would provide the basis for adjustments to the catches reported to NAFO.

Consultants are currently engaged by the Project to obtain catch records from the home ports of Portuguese and Spanish fleets which have fished in the Northwest Atlantic. These will be matched to records available from NAFO. Processes like this will be used to obtain better estimates of unreported and illegal fishing activities.

Discussion

Our approach is ambitious and relies upon considerable skilled collaboration. However, the need for the best, most complete records of total catch (=mortality estimates) and fishing effort is critical for the rational re-examination of fisheries policies in the light of historical stock collapses and current concerns. The collection of basic catch and fishing effort statistics is expensive and requires local knowledge to overcome errors in coding and interpretation. The role of consultants familiar with the fisheries in question is important to addressing any shortcomings in the official datasets and, particularly for inshore, artisanal, or recreational fisheries. Their data will be matched to records available from NAFO and other agencies.

Doubtless, mistakes were made when the official datasets were compiled but these are almost inevitable, and pale in comparison to deliberate omissions caused by functional or jurisdictional limitations. By functional limitations we mean that the data (at least in a usable form) do not exist, so it cannot be reported (because of the limited resources available). An example of this is illegal fishing; since estimates of such catches are not often made, they cannot be reported. In contrast, where data available, such as estimates of discarding, it may not be within the mandate of the reporting agency to include them in official catches. This is understandable as the need to report the value of landed catch underlay the genesis of most the world's fisheries statistical systems and this purpose still dominates all others. Jurisdictional boundaries, for example between tiers of government, may make the production of a comprehensive database, one that accounts for all sources of fishing mortality at all life-history stages, very difficult. Our Canadian case study from the Northwest Atlantic demonstrates that it is possible to make estimates of discarding for even non-commercial species if observer programs are in place. This would be very difficult based on the scaling of reported commercial landings alone. The estimates we have reported here can be improved through the use of general linear models or similar approaches that would allow estimates of blocks of time and space where no observer data exists. Using these methods it may also be possible to make estimates of discards for years when the observer program did not operate.

Our estimates confirm that discarding was not a minor phenomenon for vessels operating on Canada's Scotian Shelf (Table 8). Mortality estimates for many species would be significantly increased if discarding were included. Even so our estimate is acknowledged to be a minimum, especially for target species where we may have significant 'observer' effects. Our total of 10,569t of discards includes fishes, crustaceans, and marine mammals. Overall, however, the overall discard rate was only 4.9% (total discards/total landings). There was discarding of major target species such as cod and silver hake. We believe that the discarding estimates for non-target (generally non-quota) species are a minimal estimate. However, it is very likely that observer presence has caused estimates of the discarding rate of target species to be greatly underestimated. These results are, however, only for a small portion of the total North Atlantic fishing grounds and for only one year. Moreover, only the groundfish trawl fishery was examined here. Nevertheless estimating discarding rates of target and non-target species, even in light of these problems, will be required before total catch estimates can be attributed to ecosystems and nations' EEZs (see Pitcher and Watson, 2000).

Alverson et al. (1994) estimated that there were nearly 686,000t per year discarded in the Northwest Atlantic alone, but unlike our estimate, these did not include marine mammals. Based on our minimal estimates, discards in this region, based on the groundfish fishery alone, would have exceeded 120,000t for 1997. To reach the total of discards estimated by Alverson et al. (1994) for this region, our overall discarding rate would have to be nearer 30% than the 4.9% we calculated for the groundfish fisheries in statistical area 4VW in 1986. Likewise, estimates of discarding for species never landed would be very difficult to include without an observer programme. Nevertheless, individual estimates of discarding rates for the Northwest Atlantic groundfish trawl fisheries ranged from one in the top twenty of those fisheries with the highest “recorded” discard ratios, to four estimates in the lowest ten overall. The highest estimate was 5.28 to 1 (i.e. more than 5 kg of discards for each kg of target species landed). Of the several Northwest Atlantic fisheries listed, including the Hake Trawl, Cod Trawl, Redfish Trawl, and Plaice Trawl, all but the last had ratios below 10% (Alverson et at. 1994). The values reported in Alverson (1994) do not represent overall discarding rates but are simply the available individual estimates (pers. comm. D. L. Alverson). With 5.8% discarding by weight, the Northwest Atlantic Cod Trawl fishery had discarding rates comparable to those we calculated from observer data.

In addition to changes in the abundance and size structure of these species, changes to fishing policy, gear configuration, and fishing practise can greatly alter the numbers discarded. Rational discussion of fisheries policy, and its impacts, requires a reliable time series of discarding estimates. We have shown one example of how estimates of discards can be made with existing data sources. In ‘data-poor’ fisheries, we must rely on estimates from similar fisheries or our knowledge of the marine community that is likely to be impacted. In many cases the marine community will already have been highly altered – fortunately these changes also can be anticipated.

Changes to government policy in many countries of the world, whether mandated by international agreements or otherwise, have meant that estimates of bycatch and discarding are more available and are now discussed openly when stock management (for the major species) is considered. Illegal catches, on the other hand, are still taken as an admission of mismanagement, enforcement failure, or industry malfeasance, and are usually not included in official figures. In the cases where these estimates must be made and included in discussions, such as in international assessments of major commercial species, care is taken not be too specific as to which country’s vessels have taken the catch and where it was taken. Unfortunately not all species currently have even this level of candid analysis. We are in the process of obtaining estimates of illegal and unmandated catch for the major North Atlantic fisheries but results are slow in coming as networks of trust are established. The need to include estimates of these catches is not yet universally accepted.

It is a well-accepted axiom, that first we must find out what is happening before we can plan to do anything about it. So it is also with marine policy. Transparent improvements to catch and effort data series will improve the quality of the debate and contribute to the sustained development that the majority of countries have already adopted as national policy

Acknowledgements

The authors wish to thank the support of the Pew Charitable Trusts to the *Sea Around Us Project.* For assistance with the Canadian data used in our case study, we thank Becky Sjare, Garry Stenson, and Joe Firth (DFO St. John's, Nfld). We are grateful for the assistance of D. Pauly and the suggestions of reviewers.

References

Alder, J., Pitcher, T., Preikshot, D., Kaschner, K. and Ferriss, B. (2000) Rapfish estimates - how good is good? In: Pauly, D. and Pitcher, T.J. (eds) Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. *Fisheries Centre Research Reports* 8(2), 136-182.

Alverson, D.L. and Dunlop, K. (1998) *Status of world marine stocks.* Fisheries Research Institute, University of Washington, FRI-UW-9814, 29 pp.

Alverson, D.L., Freeberg, M.H., Murawski, S.A. and Pope, J.A. (1994) A global assessment of fisheries bycatch and discards. *FAO Fisheries Technical Paper* No. 339, 233 pp.

Anon (1997) A report on the First Session of the Advisory Committee on Fishery Research, Rome, Italy, 25-26 November 1997. *FAO Fisheries Report* (571), Rome, 34 pp.

Anon (1998) Report of the joint ICES/NAFO working group on harp and hooded seals. ICES, ICES Headquarters, Copenhagen, Denmark, *ICES CM* 1998/Assess:3, 37 pp.

Anon (1999) Report of the joint ICES/NAFO working group on harp and hooded seals. ICES, ICES Headquarters, Copenhagen, Denmark,, *ICES CM 1999/ACFM*:7, 33 pp.

Brandt, A. von (1984) Fish catching methods of the world. *Farnham: Fishing News* 3rd Ed. 418 pp.

COFREPECHE (1996) Comparative study of conversion coefficients used for estimating the live weight of fish caught by Community fishing vessels. *Final Report of Project 95/02, Centre Ifremer de Brest, BP. 70 F-29280* PLOUZANE, Commission of the European Communities, Directorate General for Fisheries. 127 pp.

Creed, C. (1996) Social responses to ITQs: cheating and stewardship in the Canadian Scotia-Fundy inshore mobile gear sector. In: Report of the second workshop on Scotia-Fundy groundfish management. Burke, D.L., O’Boyle, R.N. and Sinclair, M. (eds) Canadian Technical Report of Fisheries and Aquatic Sciences 2100, 72-82

Froese, R. and Pauly, D. (eds) (1998) FishBase 2000: Concepts, design and data sources. *ICLARM*, Manila.

FAO (1980) *Yearbook of fishery statistics: catches and landings. Vol. 50.* Food and Agriculture Organization of the United Nations, Rome. 386 pp.

Grainger, R.J.R. and Garcia, S. (1996) Chronicles of marine fisheries landings (1950-1994): trend analysis and fisheries potential. *FAO Fisheries Technical Paper* No. 359, 5 pp.

Gulland, J.A. (1983) *Fish stock assessment: a manual of basic methods. Vol. 1.* FAO/Wiley Series on Food and Agriculture. 223 pp.

Hammil, M. O. and Stenson, G. B. (in press) Estimated prey consumption by Harp seals (*Phoca groenlandica*), Grey seals (*Halichoerus grypus*), Harbour seals *(Phoca vitulina)* and Hooded Seals (*Cystophora cristata*) in the Atlantic Canada. Journal of Northwest Atlantic Fisheries Science.

ICNAF (1970) Sealing statistics for 1937-1968. *Statistical Bulletin* 18, 124-140.

Patterson, K.R. (1998) Assessing fish stocks when catches are misreported: model simulation tests, and application to cod, haddock, and whiting in the ICES area. *ICES Journal of Marine Scien*ces 55, 878-891.

Pauly, D. (1998) Rationale for reconstructing catch time series. *European Commission Fisheries Cooperation Bulletin* 11(2), 4-7.

Pauly, D., Christensen, V., Longhurst, A., Sherman, K. and Watson, R. (2000) Mapping fisheries onto marine ecosystems: proposal for a consensus architecture allowing inferences at local, basin and global scales. In: Pauly, D. and Pitcher, T.J. (eds) Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. *Fisheries Centre Research Reports* 8(2), 13-22.

Pitcher, T.J. and Watson, R. (2000) The basis for change 2: estimating the total extractions from marine ecosystems. In: Pauly, D. and Pitcher, T.J. (eds) Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. *Fisheries Centre Research Reports* 8(2), 40-53.

Pontecorvo, G. (1988) The state of worldwide fisheries statistics: a modest proposal. *Marine Resource Economics* 5, 79-81.

Ruttan, L.M., Gayanilo, F. C. Jr., Sumaila R. U. and Pauly, D. (2000) Small versus large scale fisheries in the Gulf of Maine and George's Bank (U.S.A.): a multispecies, multifleet model for evaluating their potential to conserve fishes and fisheries. In: Pauly, D. and Pitcher, T.J. (eds) Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. *Fisheries Centre Research Reports* 8(2), 64-78.

Sergeant, D. E., (1991) Harp seals, man and ice. *Can Spec Publ Fisheries and Aquatic Sciences* 114, 153p.

Stenson, G. B., Sjare, B. and Wakeham, D. (1999) Catch-at-age of northwest Atlantic harp seals. *DFO Atlantic Fisheries Research Documents*, 99/105, 21p.

Sumaila, R., Chuenpagdee, R., Munro, G. (2000) Tracking fisheries landings in the North Atlantic. In: Pauly, D. and Pitcher, T.J. (eds) Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. *Fisheries Centre Research Reports* 8(2), 114-122.

Tyedmers, P. (2000) Estimating the industrial energy inputs and related ecological costs associated with the commercial fishing fleets of the North Atlantic. In: Pauly, D. and Pitcher, T.J. (eds) Methods for evaluating the impacts of fisheries on North Atlantic ecosystems . *Fisheries Centre Research Reports* 8(2), 123-135.

Zeller, D. and Pauly, D. (2000) Life history patterns and depth zone analysis. In: Pauly, D. and Pitcher, T.J. (eds) Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. *Fisheries Centre Research Reports* 8(2), 54-63.

APPENDIX 1: MARINE MAMMALS

Marine mammal catches are not recorded in a central database at DFO as happens for fish and invertebrates. The majority of seal hunting activity is aimed at harp and hood seals. The grey seal was the object of a bounty hunt from 1967 to 1984. The catches of other species of seals were aggregated as "other seal". The principal task was to obtain the data from different sources by contacting researchers working on one or several species. Second, the catch information that was available was the number of animals in the catch by age groups. The grouping differs depending on the hunting grounds and the species. In order to convert the catch in number to yield, the mean body weight for each broad size category was derived from growth curves in the best cases, and from more general data on adult sizes in other cases. Additional sources of removals, largely unaccounted for, are the animals wounded or killed but never recovered ("struck and lost" in seal hunt terminology), and are included in the catch statistics. Research has been undertaken to estimate the number of harp seals it represents (G. Stenson, pers. comm.).

Table 9. Weight for each commercial category for Harp seal.

Stage	Age	Weight (kg)	Rationale
pups	15 –30 days	30	Stewart and Lavigne 1980; Sergeant, 1991 p.27
adult	ages 5-10 most likely assume sex ratio 50%	57-84	Based on catch at age data and Gompertz growth equation (April weight)

Reconstructing Harp Seal Catch

Data sources

Harp seal are the most abundant catch of marine mammals in western Atlantic Canada. Since harp and hooded seals are migrating from the Gulf of St. Lawrence and Newfoundland to Baffin Bay, southeastern Greenland and Hudson Strait, the catch of both Canadian and Greenland waters should be considered as sources of mortality on the same population. The Joint ICES/NAFO Working Group on harp and hood seals considers catches from West Greenland and half of Southeast Greenland derived from the Northwest Atlantic harp seal stock (Anon., 1998). Effort data for Norwegian and Russian hunting directed on West Greenland Ice were obtained from Appendix IV of the ICES document (Anon., 1999). Catch at age for the years 1952-1998 for each region, Gulf, eastern Arctic, and Greenland, were obtained from (Stenson et al., 1999). Catches for years 1950-1951 were obtained from ICNAF (1970).

Catch biomass

The harp seal catches are now recorded in two size categories, pups and 1+. Pup weight was obtained from Sergeant, 1991 (his Table 8). Mean weight of the catch was obtained by using the catch-at-age data for years 1952 to 1998 (Stenson et al. 1999) and the weight at age W_a computed from a Gompertz curve (Hammil and Stenson, in press).

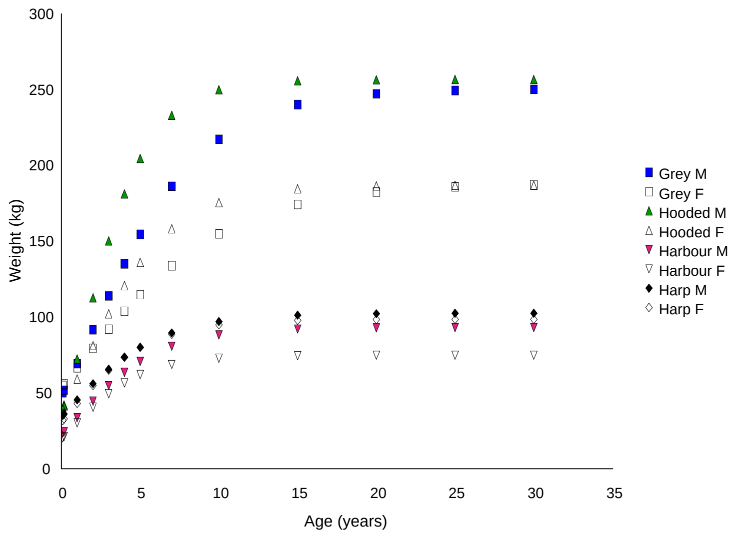


Figure 3. Weight at age relationship for Canadian seal species.

$$Biomass_y = Catch_{pup,y} * W_{pup} + \Sigma\, rop_{a,y} * W_a * Catch_y$$

where $prop_{a,y}$ is the estimated age composition of the yearly catch ($Catch_y$), expressed in percentages. The age composition for years 1950-1951 was assumed to similar to that of the year 1952. The Gompertz curve (Figure 3) was computed from specimens examined in April when seals are leaner than in the winter (Sergeant, 1991). The resulting mean weight of adult seals varied from 57 to 84 kg over the years (Table 8). Pup weight was estimated at 30 kg where seals are 15 to 30 days old. The hunting period varies within Canadian regions but the error in taking the April weight is probably smaller than error in the estimation of the catch at age (G. Stenson, DFO, St. John's Newfoundland, pers. comm.).

THE BASIS FOR CHANGE 2: ESTIMATING TOTAL FISHERY EXTRACTIONS FROM MARINE ECOSYSTEMS OF THE NORTH ATLANTIC

Tony J. Pitcher and Reg Watson

Fisheries Centre, University of British Columbia

Abstract

The reason for estimating total extractions of fish is to able to account for their impacts on marine ecosystems. Such an evaluation has not been attempted before, since ecosystem modelling techniques suitable for this purpose have only recently become available. Putting a figure on total extractions entails the difficult task of estimating, in addition to reported landings, discards, illegal, and unmandated catches, including disreported catches. These unreported extractions cast various types of shadows, many of which may be tracked and estimated quantitatively. Official figures often have an implicit assumption that such categories are zero, an unacceptable option for an ecosystem-based project. Some examples of adjustments for unrecorded catches are reported. We describe an innovative, well-funded NGO that tracks and publicizes illegal catch in the Southern Ocean and which may provide a model for other areas of the world such as the North Atlantic. We present an adjustment procedure based on a simple spreadsheet, divided into categories of unreported annual catch. Adjustment factors are based on reports from observers, confidential correspondents and on information published in a variety of sources. Over time the adjustment factors respond to changes in regulatory regime and hence the incentives and disincentives to mis-report. Once in place, this method provides preliminary estimates that may be refined without disruption. Preliminary estimates, set up as a 'straw man' for Atlantic Canada, suggest average figures since 1960 of around 30% for unreported extractions of cod and over 100% for herring. Although at first sight an adjustment procedure for illegal catch may appear controversial, we argue that such transparency is not only an essential part of a new fisheries regime that mimimizes deleterious impacts to marine ecosystems, but is also in conformity with the treatment of other kinds of fraud in contemporary society.

"Shame to him that speaks not forth: for never was the time so good as now"

Robert le Coq, Bishop of Laon, 1356, denouncing the anarchy that prevailed under misrule by the Dauphin of France.

INTRODUCTION

In order to evaluate the impacts of fisheries on North Atlantic ecosystems, the total annual amount of fish killed, from all species and by each fishery, has to be estimated. Obtaining these figures is not a trivial exercise because some items are not recorded, for a variety of reasons, in published catch statistics. In this paper we aim to present a methodology for making such estimates of unreported catch, following on from the database methodology presented in *The Basis for Change 1* (Watson et al. 2000). Our analyses will touch on controversial topics, and can be expected, in some cases, to be at variance with conventional assessments or official positions.

CATEGORIES OF UNREPORTED CATCH

Fisheries catches may be separated into three components:

1) nominal catch, that reported to a monitoring agency, generally to national body that itself reports to the FAO (Food and Agriculture Organization of the United Nations);
2) discarded by-catch, the non-targeted part of a catch, often consisting of the juveniles of targeted or other species, caught due to the unselective nature of the gear used, and usually thrown overboard rather than landed and generally. At least in recent years, in North Atlantic fisheries, this is estimated by some sort of observer program;
3) unreported catch, consisting of categories not covered by the reporting system in question

Category 3, unreported catch, as illustrated in Figure 1, may be composed of:

1) unreported discards: fish of species or sizes not wanted by the fishing vessel. Discards may be in excess of quota, high grading, and may or may not be illegal, but are amounts not reported by observers.
2) unmandated catches: catches that a given agency is not mandated to report, either on account of the small size of the vessel (catch is not recorded from small inshore vessels in the UK), or the nature of the species (a by-

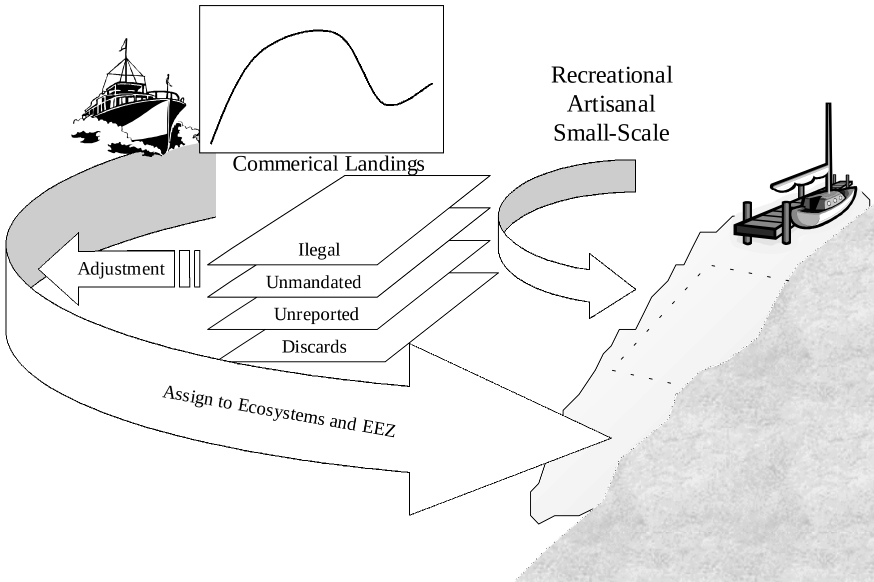


Figure 1. Illustrating how various categories of unreported catches maybe used to adjust reported landings and discards to estimate total extractions from a marine ecosystem.

catch of dogfish sharks often goes unreported). It may include discards of species not considered important enough to record, such as pelagic species like herring in some groundfish fisheries. A further example is catch from sport fisheries, which is often unmandated (it is not included in the FAO database) but can have significant impacts (see Walters 1995).

3) illegal catch: catches that contravene a regulation from the regulatory body. May be unreported by being landed away from the home port, or trans-shipped to foreign flagged vessels at sea. Includes disreported catches: catches whose identity (by species or size) may be deliberately misreported and concealed. Disreporting usually conceals quota violations, such as haddock reported as cod, or salmon concealed under surface layers of hake.

In the developed countries of the North Atlantic, the catch of fish of each commercially important species is routinely estimated by sampling at the ports of landing (Shepherd 1988), but this can be a difficult task, especially with scattered small-scale artisanal fisheries. Most of the above categories are missed by official fisheries catch statistics gathered in many countries, whose statistical systems were generally set up to track landings for economic purposes rather than the amount of fish killed by fisheries. Log books and sales figures kept by fishing captains or owners provide an alternative system, which has the advantage of also giving data on fish discarded at sea before landing, and on fishing effort. Interviews with fishers may provide historical information (Pauly 1998). But even the most plausibly diligent fishers can make mistakes under difficult conditions, and data from poorly-paid officials or observers employed to record landings can be less than accurate.

An assumption of zero is unacceptable

Where landing or catch data does not provide amounts of discards, or estimate unreported catches such as llegal and unreported catch, transshipments, or unmandated catches, it is important to realize that an implicit assumption has been made that such categories are zero. It is not our purpose to comment on the effect that such assumptions may have on conventional stock assessments, and in fact estimates of some catches, sometimes called 'unassigned', are often made and used in both the ICES and NAFO arenas at closed stock assessment workshops. Presumably for fear of embarrassing state governments, these figures generally remain confidential, or lie concealed in semi-private stock assessment working papers. In any event, they are not attributed to nations or locations but only to the fish stocks under examination. But leaving these figures at zero, as databases in the public domain tend to do, is unacceptable when trying to examine the impact of fisheries on marine ecosystems where total extractions must be estimated. Political pressures maybe such that even FAO's own, well-founded study of discards (Alverson et al. 1994) are omitted from the published FAO catch database.

Hence, the assumption of a zero adjustment to reported landings should not be used (Pauly 1998). Any percentage estimate of unreported catch by category, based on validated information, will be closer to the truth, and so should be used as a default in estimating the total catch figure for North Atlantic ecosystems modelled in Sea Around Us project. It is hoped that improvements to our default figures may well be stimulated by its publication.

As well as unreported and illegal catches, the total mortality experienced by a stock also includes ghost ('cryptic') fishing mortality and other unaccounted sources of mortality. This topic is comprehensively reviewed by Alverson et al. (2000), building on the work of ICES (1995), and is not considered in detail here.

EXAMPLES OF HOW UNREPORTED CATCH HAS BEEN DEALT WITH

Lake Malawi

- In Lake Malawi, usipa, a small, streamlined, silvery pelagic zooplanktivore belonging to the carp family, is the subject of a considerable artisanal seine net fishery. The fish are caught at dusk and through the night with the aid of lights. There are small local markets for the fresh fish, but the bulk of the catch is sun-dried and exported from the lake shore, the local variety of a traditional and important food commodity known in central Africa as 'kapenta'. Official FAO statistics record a total catch of 3,000 to 5,000 tonnes of usipa per year, but this figure seemed low according to the suspicions of experienced fishery biologists.

  For eight months in 1985/6, Lewis and Tweddle (1990) stationed observers on the only two roads leading out of the Nankumba peninsula, situated in the heart of the usipa fishery, who censused all trucks and their sacks of dried usipa. Local consumption and usipa exported by lake steamer was also estimated. The catch from the peninsula, which represents only 5% of the lake shoreline, was calculated as five times greater than the official catch for the whole lake. Scaling up the Nankumba catch to an estimate for the whole lake involved a number of assumptions, but the total catch in 1985/6 was probably between 50,000 and 100,000 tonnes, contrasting with the official figures of 5,573 tonnes from beach recorders.

Ecuador

- In the late 1980s the tropical chub mackerel fishery in Ecuador landed over 500,000 tonnes per year, caught by a fleet of small vessels of 20 to 350 tonnes, most of which sell their fish directly to fishmeal factories at three ports along the coast. Official landing figures were suspect and a log book system had proved unreliable. Since catches and catch-per-unit-effort for this economically important fishery have been declining markedly, an accurate assessment of the fishery using reliable catch data was urgent (Patterson 1990, Pitcher and Stokes 1990) and indeed the stock collapsed soon afterwards (Patterson et al. 1993). The catch was cleverly estimated from the numbers of sacks of fishmeal output from the fishmeal factories (Patterson et al. 1990). The weight of fish input to the fishmeal process was back-calculated from the conversion ratios at each stage of the industrial process. The number of fishing vessels in each month was estimated from official permits issued each day ('zarpes'). Knowledge of the fleet structure allowed an estimate of the catch which did not go through this route (approximately 15%). Not only were the final catch estimates about double the official catch statistics, but disconcertingly there was poor correlation between the two sets of figures.

Peru

- During the heyday of the Peruvian anchovy fishery, in the late 1960s and early 1970s, it was realized that official statistics massively underestimated true catches, and that fishmeal plants were operating at much less than their mandated conversion efficiency. While the official figures were never revised (and are still cited, D. Pauly, pers. comm.), structured interviews of 40 former participants in the industry by one of the former participants pointed out the need to revise the official catch figure from 12 million tonnes in 1970 to 16 million tonnes, the actual value. Indeed, only the corrected catches are compatible with the true conversion efficiency of the reduction plants, and with fishmeal exports (Castillo and Mendo 1987).

North Atlantic

- In 1997 it is estimated that more than 75 % of the reported Spanish catch of 37,000 tonnes of swordfish was illegal. ICCAT's own records show that Spain exceeded its catch limit in both the North and South Atlantic in every year from 1996 when the ICCAT quotas were introduced. For Bluefin tuna, Spain exceeded the catch limits of about 8000 tonnes by 19% in 1995, 58% in 1996 and 51% in 1997. Moreover, France, Italy, Japan and Morocco are reported as having illegal catches for Bluefin tuna and swordfish as large as those of Spain (Raymakers and Lynham 1998).

- Patterson (1998) used an "adapt" type of nonlinear-least-squares tuned VPA model in comparison with standard ICES VPA in order to estimate unreported catch. The Patterson model is able to provide good estimates of stock size and therefore catch, even when catches are under-reported. The method was used with three gadoid fisheries, North Sea cod and west Scotland cod and whiting. Patterson concluded that the West Scotland stocks, but not those in the North Sea, had been substantially under-reported since 1991 by a factor of 30-60%.

- In Scotland and France, large quantities of 25-30 cm cod are illegally landed as "blue greens", and under a different name, in France [2 correspondents].

- In western Ireland, the catch of large midwater trawlers targeting herring and mackerel is estimated to be at least 100% of the reported catch, with the consequence that the true catch was likely double the quota of 50,000 tonnes [1 correspondent].

- At least 50% of the catch of Scottish purse seiners is said to be illegal [1 correspondent].

- Unreported catch is said to equal reported catch for Humberside fisheries, and higher figures applied to historical periods of distant water fleets before the EEZs. [1 correspondent].

- In Denmark, cod landings are often disreported as dogfish shark. [1 correspondent].

- In Canada, the arrest of a Spanish trawler (the *Estai*) in 1995, revealed a secret specially-constructed hold that concealed unreported, illegal and undersized catch. There were two sets of log books, each reporting different catch figures. From the skipper's secret logbook, total catch was found to be 100% underreported [Harris 1998]. Moreover, 98% of the catch was undersized (and hence illegal).

- A significant amount of catch from the *Estai* was recorded in the logbook of another Spanish vessel, the *Patricia Nores* [Harris 1998]

- 45% of all Spanish catches of flounder are said to be discarded at sea and not reported [Harris 1998].

- In the late 1980s, every haul of the trawl by Russian vessels was estimated to be under-reported by at least 10 tonnes [Internal DFO document, quoted by Harris 1998].

Harris (1998), who appears to have had access to a considerable amount of privileged information, reports many instances of discards and disreported catch. His book can therefore be used to provide preliminary figures for Canadian waters. We are preparing a corrigenda from his book that may be used to tune estimates of discards and illegal catch for his region. We realize that it is easy to journalists' reports, but we would hope for better figures from those who have better knowledge.

An NGO tracking illegal fish catch

The 1996/7 annual quota for Patagonian toothfish (*Dissostichus eleginoides*), served as 'Chilean Sea Bass' in expensive seafood restaurants world-wide, was set at 17,000 tonnes by CCAMLR *(Commission for the Conservation of Antarctic Living Marine Resources)*, illegal catches taken around Heard and McDonald Island (Australia), Kerguelen Island (France) and Prince Edwards and Marion Island (South Africa), appear to have exceeded the legal quota by a factor of 500%. These illegal catches and sales of toothfish have been traced by an NGO, ISOFISH *(International Southern Oceans Longline Fisheries Information Clearing House)*.

Based in Hobart, Tasmania, and associated with CCAMLR, ISOFISH is funded by the Australian fishing industry. ISOFISH aims to track and report the unlicensed fishing activities of toothfish longliners and monitor the trade in illegally caught fish in cooperation with national authorities and the international regulatory body, CCAMLR.

The ISOFISH web site lists over 90 named individual boats and their owners, many with detailed records of their illegal activities. A newsletter dated March 1999 examines the Chilean fishing industry and names the 'pirate king' of the industry, (Roberto Verdugo, former Under-Secretary of State for Fisheries in a Chilean government) worth US$100 million in exports (80% to Japan) from Chile in 1997. Along with seven other Chilean companies, over 50 fishing vessels sell illegal toothfish catches. A

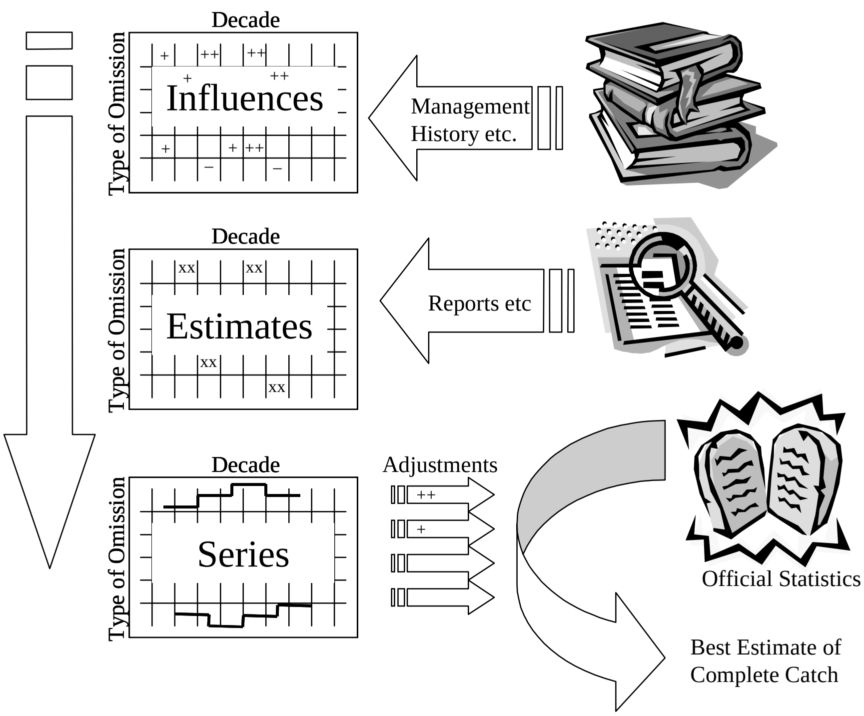


Figure 2. Illustrating adjustments to landings data to construct total fishery extractions from a marine ecosystem. A shifting climate of influences and point estimates at top lead to adjustment factor matrix at bottom of diagram.

1999 report states, "ISOFISH has enough evidence to publicly identify these companies as knowingly and persistently involved in and benefiting from toothfish poaching." By 1998, to its credit, government counter-measures in Chile were aimed at exposing the trade. However, a consequence was the re-flagging of many of these vessels in Belize, Panama, and Honduras. Moreover, port and trade authorities in Uruguay, Mauritius, Mozambique, Namibia and the French island of Réunion are identified as "providing unquestioning support" to the poachers, and being involved in trans-shipments of illegally caught fish.

ISOFISH is a good model of what may achieved, with adequate funding, in identifying specific illegal fishing and tracking the trade in illegally-caught fish that drives such activities.

Proposed Method for the SAU Project

Basis of the adjustment method

We present an adjustment procedure based on a simple spreadsheet, divided into categories of unreported annual catch (Figure 2). Adjustment factors are based on reports from observers, confidential correspondents and on information published in a variety of sources. Over time, the adjustment factors respond to changes in regulatory regime and hence the incentives and disincentives to mis-report. Once in place, this method provides preliminary estimates that may be refined without disruption, and offers a basis for collaboration and discussion. Figure 2 illustrates the general principles of the procedure. In Table 1 (a to f) we show a hypothetical example of the adjustment process. In each case we show five sections of catch adjustment: discards that are reported by observers (or in some other fashion); discards that are unreported (for example in the absence of observers); unmandated catches (defined as above); disreported catches and illegal landings (fish ultimately landed and sold somewhere in the world).

For each species these categories are shown for domestic and for foreign fleets. Table 1a lists a set of influences on misreporting, mapping the 'incentive climate' as it were, tabulated in 5-year periods. Table 1b contains some estimates used as anchor points that have some reasonable validation, obtained from surveillance, informants or other sources. Each anchor point is documented as to its source (as far as is possible). Table 1c shows adjustment factors interpolated between the point estimates of 1b using influences from 1a. Interpolations here are simply performed linearly between the points with information – obviously more sophisticated statistical methods could be used.

Total officially reported landings are listed in Table 1d: this data is extracted from official databases. Missing catches in Table 1e are estimated by multiplying the factors from1c by the landings in 1d. Hence Table 1f provides estimates of total extractions.

The most difficult part of the work is developing Tables 1a and 1b. It is important to emphasize that all anchor points at stage b are explicitly footnoted, even if exact sources cannot be revealed in some cases. Beyond this point the method flows fairly automatically and in such a way that most criticism is forced by the scheme to

Table 1. Illustrating the catch adjustment process. (a) climate of factors influencing misreporting; (b) documented point estimates (anchor points) of misreporting from informants or others; (c) interpolated adjustment factors; (d) landings (and recorded discards) data; (e) missing catch data; (f) estimated total fishery extractions from ecosystem.

Species	**Jurisdiction**	**Type**				**Period**			
(a) INFLUENCE FACTORS			**1960s**	**1970-74**	**1975-79**	**1980-84**	**1985-89**	**1990-94**	**1995-99**
Species A	Domestic	Obs discards	None	Some	Lots	Heaps	?	Lots	Some
		Obs effect discards	None	Some	Lots	Heaps	?	Lots	Some
		Unmandated	?	Some	Lots	Heaps	?	Lots	Some
		Disreported	None	Some	Lots	Heaps	Lots	Lots	Some
		Illegal	None	Some	Lots	Heaps	Lots	Lots	Some
	Foreign	Obs discards	None	?	Lots	Heaps	Lots	?	?
		Obs effect discards	None	Some	Lots	Heaps	Lots	Lots	?
		Unmandated	None	?	Lots	Heaps	Lots	?	?
		Disreported	None	Some	Lots	Heaps	Lots	Lots	?
		Illegal	None	Some	Lots	Heaps	Lots	Lots	?
Species B	Domestic	Obs discards	None	Some	Lots	Heaps	?	Lots	Some
		Obs effect discards	None	Some	Lots	Heaps	?	Lots	Some
		Unmandated	?	Some	Lots	Heaps	?	Lots	Some
		Disreported	None	Some	Lots	Heaps	Lots	Lots	Some
		Illegal	None	Some	Lots	Heaps	Lots	Lots	Some
	Foreign	Obs discards	None	?	Lots	Heaps	Lots	?	?
		Obs effect discards	None	Some	Lots	Heaps	Lots	Lots	?
		Unmandated	None	?	Lots	Heaps	Lots	?	?
		Disreported	None	Some	Lots	Heaps	Lots	Lots	?
		Illegal	None	Some	Lots	Heaps	Lots	Lots	Some
(b) ANCHOR POINTS (%)									
Species A	Domestic	Obs discards[L]							7[A]
		Obs effect discards							10[B]
		Unmandated		10[C]					
		Disreported	0[D]			100[E]			25[F]
		Illegal			30[G]				
	Foreign	Obs discards[L]							
		Obs effect discards							
		Unmandated							
		Disreported							
		Illegal						40[H]	
Species B	Domestic	Obs discards[L]							
		Obs effect discards							
		Unmandated			25[I]				
		Disreported							
		Illegal							
	Foreign	Obs discards[L]							
		Obs effect discards							
		Unmandated	0[J]						
		Disreported				80[K]			
		Illegal							

Notes on sources for Anchor Points (examples)

(A) Informant A. (B) DFO surveillance reports. (C) Harris (1998). (D) Harris (1998). (E) Informant B. (F) Informant A. (G) DFO estimate, Anon. (H) Harris (1998). (I) Word Bank Study 1990. (J) Informant A. (K) Informant A. (L) This is the portion of the observer discards that are discarded when no observer is present.

c) INTERPOLATIONS

Species A	Domestic	Obs discards	0.07	0.07	0.07	0.07	0.07	0.07	0.07
		Obs effect discards	0.10	0.10	0.10	0.10	0.10	0.10	0.10
		Unmandated	0.00	0.10	0.30	1.00	0.30	0.40	0.25
		Disreported	0.00	0.10	0.30	1.00	0.30	0.40	0.25
		Illegal	0.00	0.10	0.30	1.00	0.30	0.40	0.25
	Foreign	Obs discards	0.07	0.07	0.07	0.07	0.07	0.07	0.07
		Obs effect discards	0.10	0.10	0.10	0.10	0.10	0.10	0.10
		Unmandated	0.00	0.10	0.30	1.00	0.30	0.40	0.25
		Disreported	0.00	0.10	0.30	1.00	0.30	0.40	0.25
		Illegal	0.00	0.10	0.30	1.00	0.30	0.40	0.25
Species B	Domestic	Obs discards	0.07	0.07	0.07	0.07	0.07	0.07	0.07
		Obs effect discards	0.10	0.10	0.10	0.10	0.10	0.10	0.10
		Unmandated	0.00	0.10	0.25	1.00	0.30	0.40	0.25
		Disreported	0.00	0.10	0.25	1.00	0.30	0.40	0.25
		Illegal	0.00	0.10	0.25	1.00	0.30	0.40	0.25
	Foreign	Obs discards	0.07	0.07	0.07	0.07	0.07	0.07	0.07
		Obs effect discards	0.10	0.10	0.10	0.10	0.10	0.10	0.10
		Unmandated	0.00	0.10	0.25	1.00	0.30	0.40	0.25
		Disreported	0.00	0.10	0.25	0.80	0.30	0.40	0.25
		Illegal	0.00	0.10	0.25	0.80	0.30	0.40	0.25

(d) LANDINGS

Species A	Domestic	Landings	12000	12000	12000	12000	12000	12000	12000
	Foreign	Landings	8000	8000	8000	8000	8000	8000	8000
Species B	Domestic	Landings	11500	11500	11500	11500	11500	11500	11500
	non-CDN	Landings	400	400	400	400	400	400	400

(e) MISSING CATCH

Species A	Domestic	Obs discards	840	840	840	840	840	840	840
		Obs effect discards	84	84	84	84	84	84	84
		Unmandated	0	1200	3600	12000	3600	4800	3000
		Disreported	0	1200	3600	12000	3600	4800	3000
		Illegal	0	1200	3600	12000	3600	4800	3000
	Foreign	Obs discards	560	560	560	560	560	560	560
		Obs effect discards	56	56	56	56	56	56	56
		Unmandated	0	800	2400	8000	2400	3200	2000
		Disreported	0	800	2400	8000	2400	3200	2000
		Illegal	0	800	2400	8000	2400	3200	2000
Species B	Domestic	Obs discards	805	805	805	805	805	805	805
		Obs effect discards	80	80	80	80	80	80	80
		Unmandated	0	1150	2875	11500	3450	4600	2875
		Disreported	0	1150	2875	11500	3450	4600	2875
		Illegal	0	1150	2875	11500	3450	4600	2875
	Foreign	Obs discards	28	28	28	28	28	28	28
		Obs effect discards	3	3	3	3	3	3	3
		Unmandated	0	40	100	400	120	160	100
		Disreported	0	40	100	320	120	160	100
		Illegal	0	40	100	320	120	160	100

(f) ESTIMATED TOTAL EXTRACTIONS

Species A	Total	**21540**	**27540**	**39540**	**81540**	**39540**	**45540**	**36540**
	Percentage Unreported	*7.70*	*37.70*	*97.70*	*307.70*	*97.70*	*127.70*	*82.70*
Species B	Total	**12816**	**16386**	**21741**	**48356**	**23526**	**27096**	**21741**
	Percentage Unreported	*7.70*	*37.70*	*82.70*	*306.35*	*97.70*	*127.70*	*82.70*

be constructive by way of improving the interpolations. Moreover, the revised total extractions are not so controversial because they are no longer identified by country of origin, rather, they are articulated upon the ecosystem in question

A preliminary example: Atlantic Canada

A preliminary influence table for fishery catches in Atlantic Canada is shown in Table 2. This table is an example: a more complete table has to be assembled with more information for a wider range of species. Similar tables will be drawn up for each major area of marine ecosystems in the North Atlantic.

Table 2. Summary of influences on the incentives to misreport fishery catches from Atlantic Canada from 1960 to present day (with thanks to Sylvie Guénette).

	1960s	1970-74	1975-79	1980-84	1985-89	1990-94	1995-99
Regulatory regimes		ICNAF quotas overestimated	EEZs	NAFO quotas		1992 cod moratorium	cod fishery still closed
Non-Canadian catch	No incentive to misreport. Slight discarding of juveniles. Discarding high for some unused species[1]	Higher misreporting				100% unreported turbot catch outside EEZ (from arrested Spanish vessel *Estai*)	
Canadian unreported catch[2]	Moderate discarding by inshore fishery when plant capacity exceeded. Discarding may be high for some unused species[1]			Offshore vessels: strong incentive to discard after enterprise allocations put in place. Inshore: moderate cod discards at wharf[3]	Offshore: high incentive to discard; Inshore: Gill nets in water too long, increased soak times decreased proportion of marketable fish. Large discards at wharf- but decrease in minimum fish size accepted by buyers[3].	Illegal catch of cod during moratorium Low discards	Illegal catch of cod during moratorium (lower after 'sentinel' and food fishery opened)
unmandated catch			lanternfish and monkfish in scallop fishery				
disreported catch						Disreported catch of cod	High for Canadian and non-Canadian (outside the EEZ) for groundfish sector

Notes:
1. Skates for example, on Georges Bank in 1951, the average capture rate for Barndoor skates was as high as 21 per tonne of cod trawled (Bigelow, H.B., Schroeder, W.C., in Casey and Myers, 1998 (this has decreased now as their abundance has decreased)
2. Unreported catch defined as: fish in bad condition, for gill nets the catch is retained for household use, for traps, the fish are too small or are dumped when the processing plant's capacity is exceeded.
3. From Hutchings and Ferguson (ms submitted).

Table 3 presents our first attempt to quantify the effects of the factors presented in general terms in Table 2 for two species caught in the Scotian Shelf fishery, cod and herring. In Table 3b it is important to try to have at least one anchor point in each row of the table. In this example, unmandated cod and herring catches do not exist, so all the values, and the anchor point, are zero.

Note that herring are targeted by the pelagic purse seine fishery but are also caught as largely unreported bycatch in the demersal trawl fishery. Our percentage figure refers here to the target herring fishery, not the trawl fishery in which herring are a bycatch. This is different to

Table 3**. Estimations of total extractions of cod (*Gadus morhua*) and herring (*Clupea harengus*) from the 4VW region of Atlantic Canada from 1960 to present day.**

(a) INFLUENCE FACTORS

Species	**Jursidiction**	**Type**	**1960s**	**1970-74**	**1975-79**	**1980-84**	**1985-89**	**1990-94**	**1995-99**
Cod	Domestic	Obs discards	Low	Low	Medium	High	Medium	Low	Low
		Obs effect discards	High	High	High	High	High	High	High
		Unmandated	None	None	None	None	None	None	None
		Disreported	None	None	None	None	None	Low	Low
		Illegal	Lots	Lots	Some	Low	Low	More	More
	Foreign	Obs discards	Medium	Medium	Medium	High	Medium	Low	Low
		Obs effect discards	High	High	High	High	High	High	High
		Unmandated	None	None	None	None	None	None	None
		Disreported	None	None	Low	Low	Low	Medium	Medium
		Illegal	Some	Lots	Huge	huge	Lots	Some	Some
Herring	Domestic	Obs discards	Lots	Lots	Lots	Lots	Lots	Lots	Lots
		Obs effect discards	High	High	High	High	High	High	High
		Unmandated	None	None	None	None	None	None	None
		Disreported	None	None	None	None	None	None	None
		Illegal	Low	Low	Low	Low	Low	Low	Low
	Foreign	Obs discards	Lots	Lots	Lots	Lots	Lots	Lots	Lots
		Obs effect discards	High	High	High	High	High	High	High
		Unmandated	None	None	None	None	None	None	None
		Disreported	None	None	None	None	None	None	None
		Illegal	Low	Low	Low	Low	Low	Low	Low

(b) ANCHOR POINTS (%)

Species	Jursidiction	Type	1960s	1970-74	1975-79	1980-84	1985-89	1990-94	1995-99
Cod	Domestic	Obs discards					2[A]		
		Obs effect discards					1000B[B]		
		Unmandated	0[C]						
		Disreported							0.5[D]
		Illegal					0.5[E]	1[E]	1.5[E]
	Foreign	Obs discards					2[E]		
		Obs effect discards					1000[F]		
		Unmandated	0[C]						
		Disreported						0.5[E]	
		Illegal							5.0[E]
Herring	Domestic	Obs discards						7G	
		Obs effect discards						10H	
		Unmandated	0[C]						
		Disreported	0[C]						
		Illegal						1[E]	
	Foreign	Obs discards						5[E]	
		Obs effect discards						50[E]	
		Unmandated	0[C]						
		Disreported	0[I]						
		Illegal						5[E]	

Notes on sources for anchor points (examples)
(A) Informant A. (B) DFO surveillance reports. (C) Unmandated category not applicable to cod in this region. (D) Informant B. (E) Harris (1998). (F) Informant A. (G) DFO estimate, Anon. (H) estimate based on similar fisheries reported elsewhere. (I) Disreporting for herring from Informant C.

c) INTERPOLATIONS

Cod	Domestic	Obs discards	0.005	0.005	0.02	0.05	0.02	0.005	0.005
		Obs effect discards	10.0	10.0	10.0	10.0	10.0	10.0	10.0
		Unmandated	0.0	0	0	0	0	0	0
		Disreported	0	0	0	0	0	0.005	0.005
		Illegal	0.04	0.04	0.01	0.005	0.005	0.001	0.015
	Foreign	Obs discards	0.02	0.02	0.02	0.05	0.02	0.01	0.01
		Obs effect discards	10	10	10	10	10.0	10	10
		Unmandated	0.0	0	0	0	0	0	0
		Disreported	0	0	0.01	0.01	0.01	0.05	0.05
		Illegal	0.05	0.1	0.3	0.3	0.1	0.05	0.05
Herring	Domestic	Obs discards	0.07	0.07	0.07	0.07	0.07	0.07	0.07
		Obs effect discards	10.0	10.0	10.0	10.0	10.0	10.0	10.0
		Unmandated	0.0	0	0	0	0	0	0
		Disreported	0	0	0	0	0	0	0
		Illegal	0.001	0.001	0.001	0.001	0.001	0.001	0.001
	Foreign	Obs discards	0.05	0.05	0.05	0.05	0.05	0.05	0.05
		Obs effect discards	50.0	50.0	50.0	50.0	50.0	50.0	50.0
		Unmandated	0.0	0	0	0	0	0	0
		Disreported	0	0	0	0	0	0	0
		Illegal	0.05	0.05	0.05	0.05	0.05	0.05	0.05

(d) LANDINGS (1000 tonnes per annum)

Cod	Domestic	225	144	164	269	261	110	8
	Foreign	51	126	76	19	13	18	23
Herring	Domestic	1144	759	320	384	217	65	2
	Foreign	214	281	121	180	72	28	0

(e) MISSING CATCH (1000 tonnes per annum)

Cod	Domestic	Obs discards	1	1	3	13	5	1	0
		Obs effect discards	11	7	33	134	52	5	0
		Unmandated	0	0	0	0	0	0	0
		Disreported	0	0	0	0	0	1	0
		Illegal	9	6	2	1	1	0	0
	Foreign	Obs discards	1	3	2	1	0	0	0
		Obs effect discards	10	25	15	9	3	2	2
		Unmandated	0	0	0	0	0	0	0
		Disreported	0	0	1	0	0	1	1
		Illegal	3	13	23	6	1	1	1
Herring	Domestic	Obs discards	80	53	22	27	15	5	0
		Obs effect discards	801	531	224	269	152	45	2
		Unmandated	0	0	0	0	0	0	0
		Disreported	0	0	0	0	0	0	0
		Illegal	1	1	0	0	0	0	0
	Foreign	Obs discards	11	14	6	9	4	1	0
		Obs effect discards	536	702	302	451	180	69	0
		Unmandated	0	0	0	0	0	0	0
		Disreported	0	0	0	0	0	0	0
		Illegal	11	14	6	9	4	1	0

(f) ESTIMATED TOTAL EXTRACTIONS (1000 tonnes per annum)

Cod	Total	**312**	**324**	**318**	**453**	**338**	**139**	**37**
	Percentage unreported	*12.8*	*20.0*	*32.5*	*57.5*	*23.0*	*8.2*	*17.4*
Herring	Total	**2797**	**2356**	**1001**	**1330**	**643**	**214**	**4**
	Percentage unreported	*106.0*	*126.5*	*127.2*	*135.6*	*122.7*	*132.1*	*77.1*

conventional fishery work where there could be no percentage discard estimate as there is no catch reported by that particular fishery. The percentage figure here refers to the percentage of unreported by-catch of herring extracted from the ecosystem, by whatever gear may catch it.

The final results show an average of 30% and for cod and 157% for herring over the whole time period, although in the most recent half-decade with data reports these figures are 17% and 77% respectively. We emphasize again that these values are intended here only as 'straw men' to be refined and improved by those more knowledgeable about these fisheries than us.

Conclusions

Unreported extractions cast various kinds of shadows on fisheries and their associated activities. These shadows can help us track them. Patterson (1998) tracked the numerical shadows of illegal catch using a VPA technique. Illegal catch generates profits that may be revealed with suitable financial scrutiny. Transshipments may be observed directly by aerial surveillance or may create unexpected landings at ports complicit in such dealings (like the deep-sea Antarctic toothfish landings in tropical Mauritius). Without VMS or human observer schemes, the shadow of discards at sea may be more difficult to track, as often the only direct observers are seabirds and marine mammals. But even here, over time, mass-balance ecosystem models may reveal shadows of extractions that need to be explained.

As set out in this paper, our method of attempting to quantify unreported catches has some advantages. When setting the anchor points, for example, informants may be asked to rank the severity of unreported catches. In fact humans are quite good at ranking things presented in pairs, asking the question "which is the better and which is the worse?" A series of paired questions might be developed for a more formal protocols here.

The method has its difficulties, for example, in that we use a percentage of the reported catch. How do we deal with the problem where no catch is reported, yet discards and illegal catch are known to occur? Patterson (1998) considers it easier to estimate catch 'reporting efficiency' (i.e., accuracy) than to make absolute estimates of unreported catch. But the key here is that we are interested in an annual value for whole ecosystems. And this in itself makes some of the issues raised by identifying the sources of anchor point estimates less controversial. Therefore figures in tonnes can be raised to annual values and compared with the annual catch of the species over the whole system.

Publicizing or covering up illegal catches in the North Atlantic?

Creating an organization similar to ISOFISH in the North Atlantic would be of great value. Keeping illegal catch under wraps is what governments tend to want to do for fear, it seems, of causing political embarrassment to allies. Even Canada, famous for the 1995 arrest, instigated by the fisheries minister Brian Tobin, of a Spanish trawler, whose secret, specially constructed hold concealed 100% unreported, illegal and undersized catch, is coy about revealing illegal fishing activities. When asked, Australia rapidly provided lists of other vessels arrested for illegal fishing such information, but this information is difficult to obtain. One study on illegal catch in Scotland (data summaries reported in Beddington et al. 1997) is a confidential document, and not obtainable by the public or other scientists.

Murawski (1996) has looked at factors influencing discards in data from the US and Canada. General linear models were fitted to discard rates, total catch, species richness, species diversity evenness, together with operational variables associated with the fishing process (codend mesh, vessel size, tow duration, total catch, target species, year, month, depth and statistical area). Variances were high, but fisheries managed by mesh and fish size generally had higher discard rates. Year classes with high abundance influenced discard rates disproportionately. Murawski worked with observer estimates of discards, whereas the focus of this paper is to suggest a method to use when such data is not available.

In the ICES area, estimates of illegal fishing are routinely made by the stock assessment working parties that regularly perform single-specie stock assessment. Yet, it is an unwritten but strictly imposed tradition that the basis of such adjustments are not made public, even when officials have direct knowledge of specific events. Such a policy of secrecy would likely be news for the public of the countries involved. Covering up for illegal fishing would be unthinkable if this were illegal drug running in North Atlantic countries. Bank staff who defraud the public of millions of dollars are not protected by a shield of anonymity – so why should this protection be afforded to illegal fishers?

Evaluation by FAO of IUU fishing

While our work was in progress, and following a series of discussions in international fora such as the International Maritime Organization (IMO), FAO convened a working group with mandate to evaluate, 'illegal, unreported and unregulated' catch (IUU: Bray 2000). Leading this initiative, Bray reviews IUU experience world wide, and points the finger at flag states for not providing adequate human and financial resources to tackle the problem.

Unfortunately, the three FAO categories do not map easily into the operational categories we use in our algorithm. Illegal catch includes both a reported element (disreported), an estimated element (e.g. observer and other estimates of discards) and an unreported component. Moreover the unregulated catch category seems ill-defined, and overlaps with our unmandated category. The term 'unauthorized' fishing is also used, but also does not easily link to our categories, except as an overarching term for all unreported and misreported catches.

In this work, however, FAO has published a very strong message concerning the critical importance of IUU fishing to the sustainability of benefits from capture fisheries. For example, Evans (2000) considers that IUU fishing distorts and devalues information from compliant fisheries, lowers allowable catches set using the precautionary approach, and increases uncertainty and the risk of overexploitation. Evans considers that, at national scales, there is often complacency about the intractability of the problem, echoing our concerns expressed above. Evans considers some fisheries, where new technology has recently made deepwater or marginal stocks vulnerable, to be underreported by as much as 75% , and in the case of stocks on the high seas, over 100%. Evans sees compliance with FAO Code of Conduct for Responsible Fisheries (see Doulman 1998; Edeson 1996) as an essential first step in improving the situation.

Doulman (2000) also considers IUU to be major flaw in present fisheries management, leading to a loss of economic and social benefits, and, in extreme cases, to the collapse of stocks. Doulman calls for a protocol that can operate regionally, sub-regionally and nationally, and be applicable to different types of fisheries and stock distributions. Hence we offer the method set out in draft here as candidate.

Finally, Edeson (2000) reviews the legal remedies available to combat IUU fishing. In particular, the possible role of the FAO Code of Conduct as an instrument of international law and a part of an International Plan of Action. Within the EEZs of nations, although some national laws might be improved, the problem is more a lack of implementation of existing regulations. Edeson considers this situation might be improved by explicit adoption of the FAO Code of Conduct. The possibility of enforcement by the flag state of the vessel is also under discussion

Benefits from a transparent new method

Obtaining estimates of the total extractions from an ecosystem as essential for a rational evaluation of the impact of fisheries When total extractions from an ecosystem are estimated, ECOPATH and ECOSIM modelling can reveal anomalies when models fail to balance, or simulated hindcasts do not fit biomass survey data. These methods can suggest alternative values for stock biomass. In some cases existing catch and biomass figures may be mutually incompatible where trophic webs cannot support them. We anticipate a number of anomalies of this kind arising from our total catch estimations.

Transparency is the only way that the many difficulties this new method will face can be reduced to a minimum. The database for SAU, together with its assumptions and modifiers used to infer total catches will be available on the World Wide Web, in order to allow the retracing of each step involved in arriving at certain conclusions. In so doing, the SAU team makes its conclusions not only reproducible in principle, as scientists always should, but also in practice. The only exception to this would be to protect the anonymity of certain informants, e.g., concerning illegal catches.

Cheating is widespread in fisheries, and the penalties are low, and the risk of detection is often low as the participants are well aware. Unfortunately, political disincentives lead many concerned with fisheries to downplay their knowledge of this cheating. Where government and official sources have strong links, and even funding, from industry, we may expect these disincentives to be stronger. Fraud on this scale has not only contributed to the depletion of North Atlantic ecosystems and contributed to disastrous stock collapses, but has foreclosed options for the future generation of wealth and sustainable benefits from marine resources. Like any other criminal act, we need to estimate its true magnitude and encourage its disclosure.

Conclusion

Our method stands or falls by the explicitness and quality of the anchor points. These need to be defendable scientifically and to withstand scrutiny by scientists, fisheries, regional and government agencies, managers and informants. Ideally, in the public interest, an analysis would obtain the support of all of these constituents

Acknowledgements

We are most grateful for discussion of the basis of this paper with Jaqueline Alder, Sylvie Guenette, Paul Fanning, Nigel Haggan and Daniel Pauly and to two referees for their helpful comments.

References

Alverson, D.L., Freeberg, M.H., Murawski, S.A. and Pope, J.G. (1994) A global assessment of fisheries by-catch and discards. *FAO Fisheries Technical Paper* 339, 233 pp.

Alverson, D.L. et al. (2000) *Report of the Fishing Technology and Fish Behaviour Working Group (FTFB) on unaccounted mortality in fisheries.* ICES, FTFB, Haarlem, the Netherlands, April 2000. 76 pp.

Beddington, J., Lorenzen, K. and Payne, I. (1997) Limits to exploitation of capture fisheries. In: Hancock, D.A., Smith, D.C., Grant, A. and Beumer, J.P. (eds) *Developing and Sustaining World Fisheries Resources: the State of Science and Management.* CSIRO, Collingwood, Australia, 529-536.

Bray, K. (2000) A global review of illegal, unreported and unregulated (IUU) fishing. Expert consultation on illegal, unreported and unregulated fishing, *FAO: IUU*/2000/6, 53 pp.

Casey, J.M. and Myers, R.A. (1998) Near extinction of a large, widely distributed fish. *Science* 281, 690-692.

Castillo, S. and Mendo, J. (1987) Estimation of unregistered Peruvian anchoveta (*Engraulis ringens*) In: official catch statistics, 1951 to 1982. In: Pauly, D. and Tsukayama, I. (eds) The Peruvian anchoveta and its upwelling ecosystem: three decades of change. *ICLARM Studies and Reviews* 15, 109-116.

Doulman, D. (1998) *The Code of Conduct for Responsible fisheries: the requirement for structural change and adjustments in the fisheries sector.* [www.fao.org].

Doulman, D. (2000) Illegal, unreported and unregulated fishing: mandate for an international plan of action. Expert consultation on illegal, unreported and unregulated fishing, *FAO: IUU*/2000/4, 16 pp.

Edeson, W.R. (1996) Current legal development: The Code of Conduct for Responsible Fisheries: an introduction. *Intl J. Mar. Coastal Law* 11: 233-38.

Edeson, W. (2000) Tools to address IUU fishing: the current legal situation. Expert consultation on illegal, unreported and unregulated fishing, *FAO: IUU*/2000/8, 13 pp.

Evans, D. (2000) The consequences of illegal, unreported and unregulated fishing for fishery data and management. Expert consultation on illegal, unreported and unregulated fishing, *FAO: IUU*/2000/12, 9 pp.

Harris, M. (1998) *Lament for an Ocean: the collapse of the Atlantic cod fishery.* McLelland and Stuart, Toronto, Canada. 342 pp.

ICES (1995) Report of the study group on unaccounted mortality in fisheries. *ICES CM* 1995/B:1, 27 pp.

ISOFISH (1999) The Chilean Fishing Industry: its involvement in and connections to the illegal, unreported and unregulated exploitation of Patagonian toothfish in the Southern Ocean. ISOFISH. Hobart, Tasmania, *ISOFISH Occasional Report* No. 2, 98 pp.

Lewis, D.S.C. and Tweddle, D. (1990) The yield of usipa (*Engraulicypris sardella*) from the Nankumba Peninsula, Lake Malawi (1985-1986). Coll. Rep. Fish. *Res. Malawi, Occ. Pap.* 1, 57-66.

Murawski, S. (1996) Factors influencing by-catch and discard rates: analyses from multispecies/ multifishery sea sampling *J. Northw. Atl. Fish. Sci.* 19, 31-39.

Patterson, K.R. (1998) Assessing fish stocks when catches are misreported: model simulation test, and application to cod, haddock and whiting in the ICES area. *ICES Journal of Marine Sciences* 55, 878-891.

Patterson, K.R., Castello, M., Santos, M. and Barcia, M (1990) [Validation and recomputation of catch and effort data in the nearshore fleet of Ecuador for the period 1981 to 1989]. Report to INP, Guayaquil, Ecuador, 47 pp. [in Spanish]

Patterson, K.R., Pitcher, T.J. and Stokes, T.K. (1993) A stock collapse in a fluctuating environment: the chub mackerel *Scomber japonicus* (Houttuyn) in the eastern central Pacific. *Fisheries Research* 18, 199-218.

Pauly, D. (1998) Rationale for constructing catch time series. *European Community Fisheries Cooperation Bulletin* 11 (2), 4-5.

Pitcher, T.J. and Stokes, T.K. (1990) *Assessment of Ecuadorean Commercial Pelagic Fisheries. Contract Report to the Overseas Development*

Administration, United Kingdom. Marine Resources Assessment Group, Imperial College, London, 60 pp.

Raymakers, C. and Lynham, J. (1999) *Slipping the net: Spain's compliance with ICCAT recommendations for Swordfish and Bluefin Tuna.* World Wildlife Fund Endangered Seas Campaign, 58 pp.

Shepherd, J.G. (1988) *Fish stock assessments and their data requirements*. In: Gulland, J.A. (ed) Fish Population Dynamics, Wiley, London, 35-62.

Walters, C.J. (1995) Fish on the Line: the future of Pacific fisheries. *Suzuki Foundation Report*, Vancouver, 87 pp.

Watson, R., Guénette, S., Fanning, P. and Pitcher, T. J. (2000) The Basis for Change I: reconstructing fisheries catch and effort data. In: Pauly, D. and Pitcher, T.J. (eds) Methods for assessing the impact of fisheries on North Atlantic ecosystems. Fisheries Centre Research Reports 8(2), 23-39.

HOW LIFE HISTORY PATTERNS AND DEPTH ZONE ANALYSIS CAN HELP FISHERIES POLICY

Dirk Zeller and Daniel Pauly

Fisheries Centre, University of British Columbia, 2204 Main Mall, Vancouver, B.C., Canada, V6T 1Z4

ABSTRACT

The life-history patterns of fish species are complex. But much of this complexity can be captured in simple diagrams of coastal transects, where juveniles usually occur in larger numbers in shallow waters, while adults generally inhabit deeper, offshore waters. Such coastal transects can be used to show how different fisheries sectors (e.g. small versus large scale) may exploit different parts of the life history of the same species or stock. Thus, a species may 'connect' small with large scale fishery sectors through their life history patterns. We show how this can be visualized through iconographic representations of generalized life history patterns and depth profiles, with specific key life-history parameters. Relevant patterns include spawning areas, nursery/juvenile distributions, adult distributions and spawning migrations. Four preliminary case studies presented here illustrate some general patterns with regard to water depth and distance from shore. The diagrams allow us to incorporate into management the concept of life history interconnectivity between different fishery sectors. This contributes to sustainable ecosystem-based approaches by informing policy options when faced with decisions to rationalize overcapitalized fisheries.

INTRODUCTION

The stock of an exploited species may be utilized by more than one fisheries sector (such as inshore, small-scale fisheries and offshore, large-scale fisheries) during different stages in the species life history (see Ruttan et al. 2000). Life history patterns are generally viewed as multi-dimensional in scale, with complex interactions between components defined by ecology, oceanography, time and geography. Often this complexity has made it difficult to assimilate potential effects of multiple fishery sectors on a species and the industry it supports. This may be either due to the perception of multi-dimensional complexity thought to be intractable, or because of an oversight of basic patterns.

Here, we argue that this multi-dimensional complexity can be reduced to a simpler, generalized two-dimensional life history pattern, while still capturing the essential information. Both Charles Darwin and Alexander von Humboldt used the method of reduced dimensionality to focus one's attention to the key issues while capturing most of the significant information concerning the topic at hand. For example, after reviewing much literature, Darwin concluded that "latitude is a more important element than longitude" for explaining the distribution of organisms (Barrett et al. 1987). This concept has recently been revisited in a latitudinal analysis of population dynamics and ecology of flatfishes (Pauly 1994). It was Humboldt, however, who in his classic *Voyage aux régions équinoxiales du Nouveau Continent* first used a transect technique to visualize the advantage of reduced dimensionality in explaining observed patterns in distribution (Gayet p. 2284-2287 in Tort 1996). In fisheries science, a classic example of data suitable for reduced dimensionality was presented by Garstang (1909) for the North Sea plaice (*Pleuronectes platessa*, Figure 1). Heincke (1913) re-expressed this as a 'law' wherein size increases with distance from shore (and depth), while numbers declined.

The life history characteristics of many species and stocks show generalized two-dimensional patterns, involving water depth and/or distance from shore. Pauly (1982) indicated such a pattern for a tropical bay and mangrove estuary, and FAO (1972) used this approach for many species in their *Atlas of the Living Resources of the Seas*. It is recognized that for many applications an inshore/offshore axis may better convey information on structure and processes than an alongshore axis or general geographic map view (Pauly and Lightfoot 1992). A good example of this is demonstrated by comparison of Garstang's map-view of plaice size distribution in the North Sea (Figure 1) with our representation of the same information for the same species and area (Figure 5). Such a transect approach permits the use of icons to represent key processes or patterns, as well as standardization of axis (e.g. log scale), which enables most species or stocks to be directly compared across extensive depth and distance scales.

The visualization of two-dimensional life history patterns is clearly only a small part in our evaluation of ecosystem effects of fishing (see Pauly and Pitcher 2000). Other components of the *Sea Around Us Project* assess the yield as well as economic benefits gained and foregone through non-optimal stock use by each fisheries sector (small scale versus large scale, incorporating gear type, vessel size and area of operation) for each area in the North Atlantic (see Ruttan et al. 2000, Munro and Sumaila 2000). We will be super-imposing the various scales of operation (depth of fishing and distance from shore) of each fisheries sector onto the life history

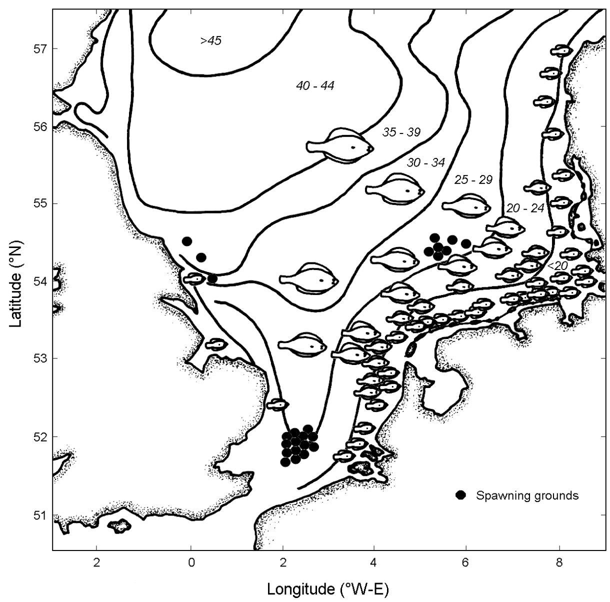


Figure 1. Schematic representation (geographic map view) of the distribution of plaice (*Pleuronectes platessa*) in the North Sea. Mean sizes (cm TL) are given for each depth isobar (modified after Garstang 1909).

illustrations of each species concerned. Thus, by utilizing standardized graphical illustrations in conjunction with a presentation of existing (or potential) 'life history/fisheries integration' problems, we hope to provide some additional impetus, as well as visual clarity, to future policy and management decisions, particularly with regards to rationalization of over-capitalized fisheries.

It might be questioned why we chose iconographic visualization as our preferred vehicle to present these patterns and the message associated with them? A clear advantage of standardized, two-dimensional graphs is that they permit comparison between different examples at one glance (Pauly and Lightfoot 1992). Tufte (1983) suggested that: "...of all methods for analyzing and communicating information, well-designed data graphics are usually the simplest and at the time the most powerful. Excellence in statistical graphics consists of complex ideas communicated with clarity, precision, and efficiency." According to Tufte (1983), graphical displays should:

"• Show the data
- Induce the viewer to think about the substance rather than about methodology or graphic design
- Encourage the eye to compare different pieces of data
- Avoid distorting what the data have to say
- Give the viewer the greatest number of ideas in the shortest time with the least ink in the smallest space."

Adhering to this theory of information presentation, our graphs are designed to be easy to decode (incorporating hues chosen to permit easy decoding by color deficient viewers, Tufte 1983 p. 183), contain key information (four major life history segments) and are standardized in scale, to permit direct comparison between species and areas.

The distance from the coast (and depth) of major population components determines their relative vulnerability to small-scale (often coastal) and large-scale (often offshore) gear and hence the existence and intensity of interactions and (potential) conflicts between these different fishery sectors. We anticipate that the final product (including the superimposed scales of operation of various fishery sectors, see Ruttan et al. 2000) will provide visual clarity on how separate fishery sectors act simultaneously on the same stock through their spatial and gear-selective fishing effort on different key life history stages. Furthermore, the depth distribution patterns will help us assign catch data assimilated by the project to the areas of the marine ecosystem classification to be used by this project. Thus, appropriate transects may be generated, if necessary, for different stocks with regards to the specific marine ecosystems as defined for the *Sea Around Us Project* (see Pauly et al. 2000).

METHODS

The species and stock specific data summarized in the coastal transects were obtained through standard literature searches, as well as species specific searches of FishBase (www.fishbase.org). The information for the Barents Sea deepwater redfish (*Sebastes mentella*) stock was augmented through a personal communication from Dr.

Konstantin Drevetnyak at the Russian Polar Institute in Murmansk, Russia.

The depth transects were obtained from ground-truthed depth data (standardized to Mean Sea Level; P. Sloss, NOAA-NGDC, pers. com.) with a two-nautical mile resolution based on satellite sources ('Global Relief' NOAA-NGDC, MGG Division, Boulder, Colorado,USA). The depth contour data was used in the *Surfer* geo-statistical program to calculate depth contours between locations relating to the general geographic area being considered. Thus, individual bottom contour transects represent typical depth contour transects derived from the stock specific geographic area. Graphs are standardized to log-scales to permit most species and stocks to be directly comparable across extensive depth (1-10,000 m) and distance scales (0.1-1,000 km), while simultaneously permitting shallow water, near-shore recreational fisheries sectors to be represented where applicable.

Four key life history stages are being used: Larval dispersal indicated by black dotted arrows (from hatching or larval extrusion to settlement or early juvenile stage), juvenile stages in blue (from post-larval to pre-fishery-recruitment stages), adult stage in brown (recruited to fishery) and spawning depth strata in red (representing depth zones used preferentially for spawning). Additional arrows indicate ontogenetic movements (blue) and regular spawning migrations (brown). The larval stage is being represented as a flow arrow only, to illustrate the link, via larval stage, between spawning areas and juvenile nursery habitats.

CASE STUDIES

In this paper we present four species as case studies, i.e., Atlantic herring (*Clupea harengus*), North Sea plaice (*Pleuronectes platessa*), Atlantic cod (*Gadus morhua*) and deepwater redfish (*Sebastes mentella*), each of which is associated with important fisheries and ecosystems of the North Atlantic.

Species with inshore / offshore patterns

Atlantic herring

Herring populations (*Clupea harengus*) often display complex feeding and spawning migrations (Iles 1971). They are separated into numerous local 'races', often identified by spawning locations and spawning periods (e.g. North Sea spring spawning herring; Muus and Dahlstrøm 1977, McKeown 1984, Blaxter 1985). Areas suitable for spawning by herring are banks and coastal areas with stony and rocky bottom and depths less than 250 m (Runnstrøm 1941a and Dragesund 1970 in Slotte and Fiksen 2000). Herring eggs are demersal (Blaxter 1985) and larval duration range from 2-6 months depending on stock (Houde and Zastrow 1993, FishBase 1999, M. Sinclair, pers. com.). Research on herring life history indicates that in many cases there is considerable mixing both in the nursery areas and feeding grounds of many stocks or 'races', while segregation occurs during spawning and early larval stages (Iles 1971, Iles and Sinclair 1982, Sinclair and Iles 1985, Sinclair et al. 1985).

Arcto-Scandian/Norwegian Spring Spawning herring stock (Figures 2 and 3)

Historically (i.e. pre-1970s, Figure 2), the Arcto-Scandian herring stock displayed extensive seasonal and ontogentic migrations. Spawning areas are along the south-western and western Norwegian coast, juvenile nursery areas are primarily in the Barents Sea, and adult feeding and over-wintering areas are offshore as far as Faroe Islands, Jan Meyers Island and Iceland (FAO 1972 maps 2.2 and B.2, Muus and Dahlstrøm 1977, Slotte and Fiksen 2000).

In recent years the Norwegian Spring Spawning herring stock (formerly called Arcto-Scandian stock) has recovered from near extinction in the late 1960s early 1970s, and appears to have re-established its previous patterns (Figure 2). For nearly 25 years after the collapse, the oceanic (Barents Sea, Iceland and Norwegian sea) nursery, feeding and wintering areas were abandoned, and the entire life cycle was spent in Norwegian coastal waters and fjords (Figure 3, Dragesund et al. 1980, Holst et al. 1998, Rottingen 1990, Hamre 1990 in Slotte and Fiksen 2000). During the 1990s, the feeding area has again expanded westwards to the Norwegian Sea (Holst et al. 1998, Slotte and Fiksen 2000), which is indicative of a return to the pattern illustrated in Figure 2. Herring larvae drift to a variety of nursery grounds in coastal fjords and the Barents Sea, and mix as adults on selected spawning grounds irrespective of nursery origin.

North Sea herring stocks (Figure 4)

The North Sea stock has generally been sub-divided into three groups, the northern North Sea summer-spawning, the central North Sea autumn-spawning and the Southern Bight winter-spawning groups (McKeown 1984). Depth-related generalized life history patterns are very similar for all three groups. Here, the Southern Bight winter-spawning group is illustrated as representative (Figure 4). Juveniles spend their early life in shallow, inshore areas. Once they reach approximately 10 cm in size, they move further offshore into deeper waters mainly to the south and east of Dogger Bank in the southern North Sea. Sexually immature but larger fish generally move further north and feed in the northern part of the North Sea. Adults migrate between the southern spawning area and northern feeding areas on an annual basis (McKeown 1984 and references therein).

Plaice (Figure 5)

The Plaice (*Pleuronectes platessa*) is a right-eyed flatfish occurring commonly in the North East Atlantic (Garstang 1909, McKeown 1984, FishBase 1999). In the North Sea four major spawning subgroups are recognized: Scottish east coast, Flamborough, Southern Bight and German Bight spawning group (McKeown 1984 and references therein). They spawn in 25-75 m depth, eggs and larvae are pelagic for approximately 3-8 weeks, and metamorphose to juveniles which settle in nursery areas in shallow, coastal waters (Muus and Dahlstrøm 1977, McKeown 1984, Figure 5). Juveniles remain in shallow waters (<20 m) for the first few years, then start moving into deeper waters. Plaice reach sexual maturity at 3-4 years, then undertake their first migration to spawning areas. Thereafter they disperse over a larger area, mainly in deeper waters, with overlap with other plaice stocks (McKeown 1984).

Atlantic Cod (Figures 6 and 7)

The Atlantic cod (*Gadus morhua*) is generally a diurnally schooling, demersal or benthopelagic species, occurring from shoreline to 500-600 m depth (FAO 1972 map B.1, Muus and Dahlstrøm 1977, Cohen et al. 1990, FishBase 1999). It can undertake long-distance migrations (FAO 1972 map 2.1, Cohen et al. 1990). Spawning takes place in 50-150 m depth for Barents Sea stock (Mukhina et al. 1995, Figure 6) and Gulf of Main/Georges Bank stocks (Serchuk et al. 1994, Figure 7). During the spawning season adults are highly aggregated and closely associated with banks or shelf-edge features (spawning areas), whereas during the non-spawning season distribution is more widely dispersed (Frank et al. 1994). Cod eggs are pelagic and concentrate in the 0-10 m depth strata , larvae hatch within 2-4 weeks of spawning, and settlement occurs after 3-5 months at 3-6 cm in size (Muus and Dahlstrøm 1977). Historically, sexual maturity was reached at between 4-15 years, however, presently this is reduced to 1-7 years due to overfishing (Serchuk et al. 1994, Longhurst 1998). Historic longevity was approximately 25 years, maximum size ~200 cm (Muus and Dahlstrøm 1977). Cod in northern Norway (Figure 6) are considered as two entities, although managed as a single stock: Norwegian Coastal Cod and Barents Sea stock (Fyhn et al. 1994, Loken et al. 1994). Loken et al. (1994) compared Barents Sea cod with Coastal Cod stocks in Norway, and found different early life histories, but no conclusive indication of different stock structure. Barents Sea cod juveniles remain planktonic for longer and settle far to the north and east in the Barents Sea (McKeown 1984, Helle 1994, Loken et al. 1994), while coastal cod juveniles settle earlier in very shallow coastal waters where the macroalgal belt might provide protection from predation (Loken et al. 1994). Similar shallow water settlement is also observed in North Sea cod (Riley and Parnell 1984 in Loken et al. 1994). Juvenile cod (1-year-old) have been reported to inhabit the shore slope of fjords between 10-30 m depth (Svendsen 1995). In the western Atlantic (e.g. Georges Bank, Figure 7), as well as on other shelf areas, most cod larvae appear to be retained on the banks used as spawning areas due to hydrodynamic patterns (Anderson et al. 1995) and the early stage of larval activity assisting movements shoalwards (Serchuk et al. 1994). Coastal cod within the Gulf of Maine (Figure 7) appear to maintain their own spawning grounds (e.g. Sheepscot Bay), and show an affinity to shallower (< 100m) coastal areas (Perkins et al. 1997).

Species with offshore pattern only

Deep water redfish (Figures 8 and 9)

In the North Atlantic there are two main species of redfish, *Sebastes mentella* (deepwater redfish, ocean perch) and *S. marinus* (golden redfish), which overlap in occurrence (FAO 1972 map A.1, Christensen and Pedersen 1989). A third species (*S. viviparus*) is generally found in shallower waters, and is the most common redfish in the North Sea and the Skagerrak (Anon. 1998).

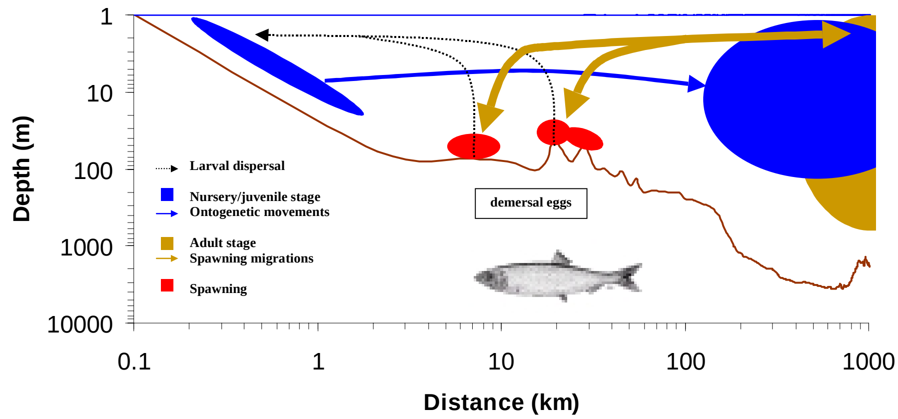


Figure 2. Generalized life history pattern by depth zone for Norwegian Spring Spawning herring (*Clupea harengus*) prior to the stock collapse in the late 1960s early 1970s, and the currently re-established pattern. Brown line represents typical depth transect from approx. 63°N, 8°E to 67°N, 11°W.

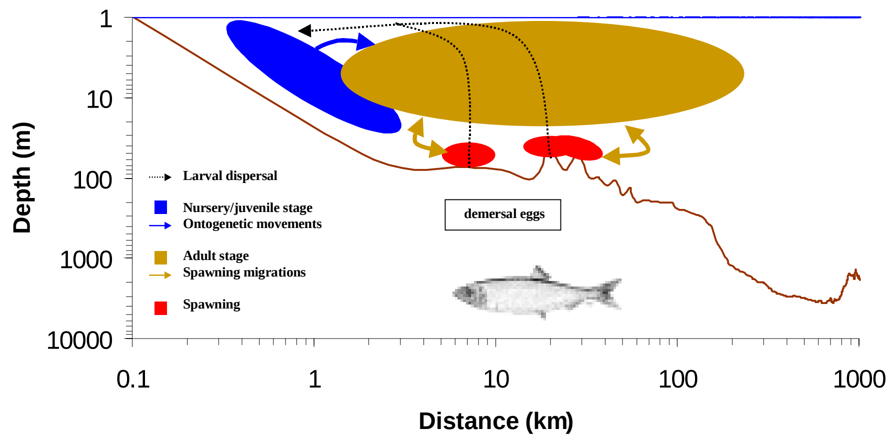


Figure 3. Generalized life history pattern by depth zone for Norwegian Spring Spawning herring (*Clupea harengus*) representative of the 25 years after the stock collapse in the late 1960s early 1970s. Brown line represents typical depth transect from approx. 63°N, 8°E to 67°N, 11°W.

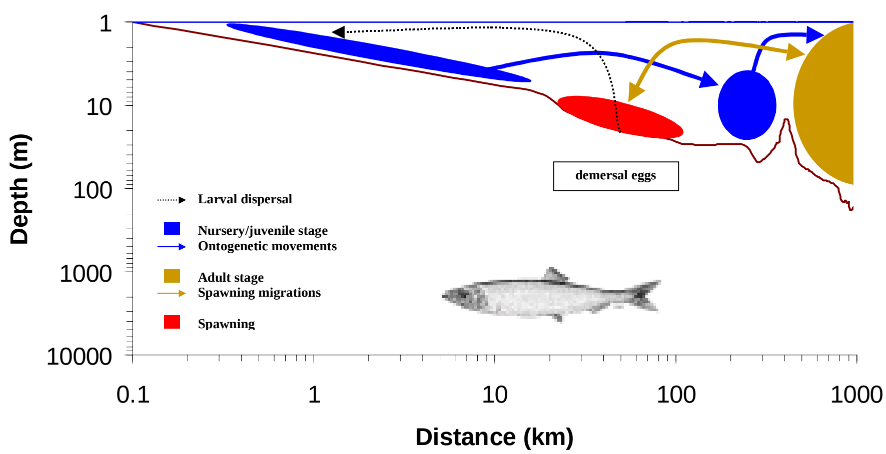


Figure 4. Generalized life history pattern by depth zone for Southern Bight winter spawning herring in the North Sea (*Clupea harengus*). Brown line represents typical depth transect from approx. 51°N, 3°E to 63°N, 2°E.

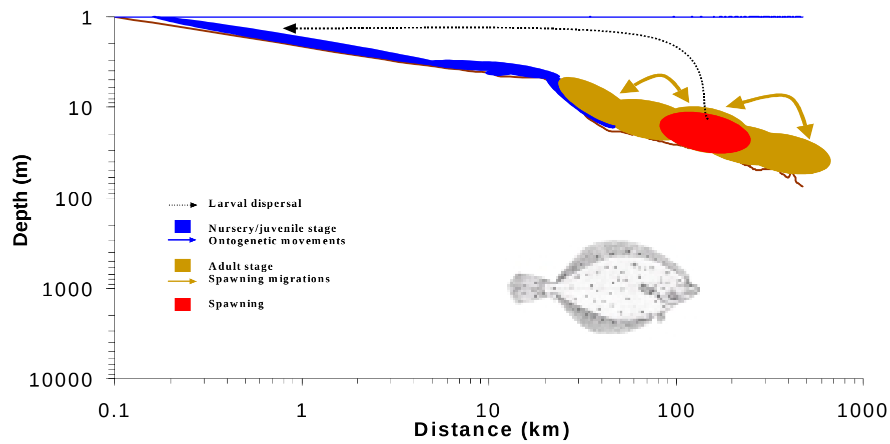


Figure 5. Generalized life history pattern by depth zone for North Sea plaice (*Pleuronectes platessa*). Brown line represents typical depth transect from approx. 53ºN, 8ºE to 56ºN, 3ºE.

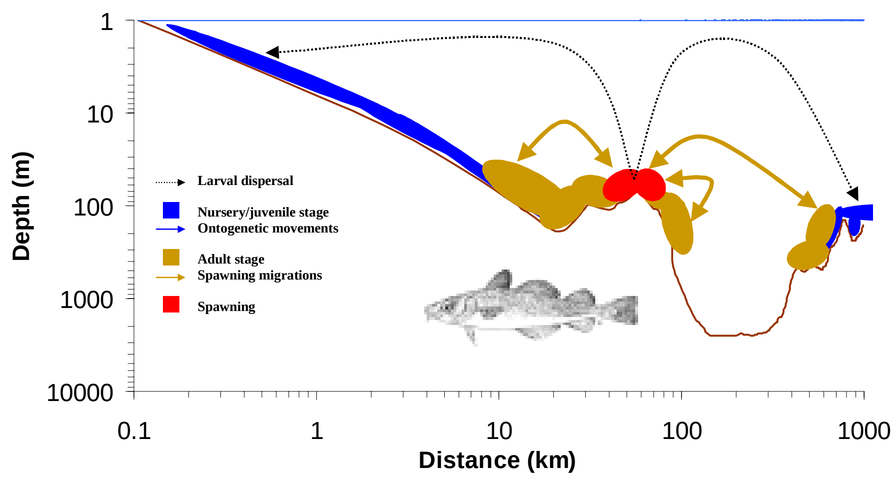


Figure 6. Generalized life history pattern by depth zone for Barents Sea and Norwegian Coastal Cod (*Gadus morhua*). Brown line represents typical depth transect from approx. 68ºN, 13ºE to 76ºN, 18.ºE.

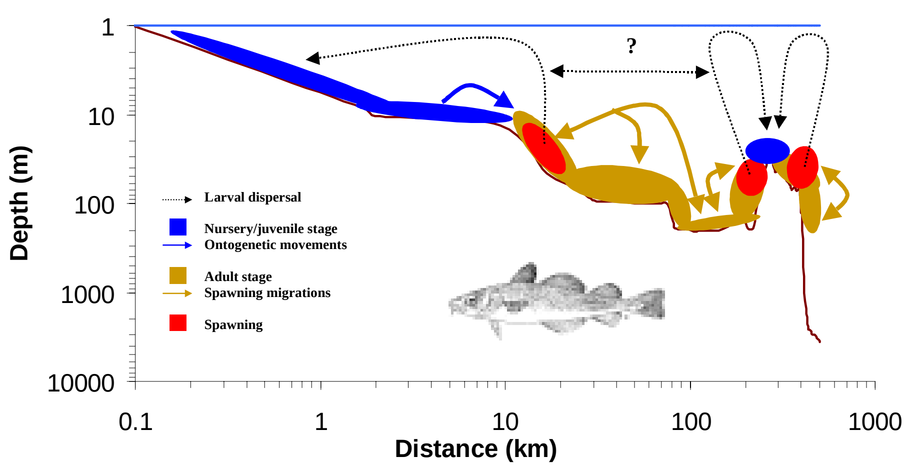


Figure 7. Generalized life history pattern by depth zone for Gulf of Maine and Georges Bank cod stocks (*Gadus morhua*). Brown line represents typical depth transect from approx. 42ºN, 70ºW to 40ºN, 65ºW.

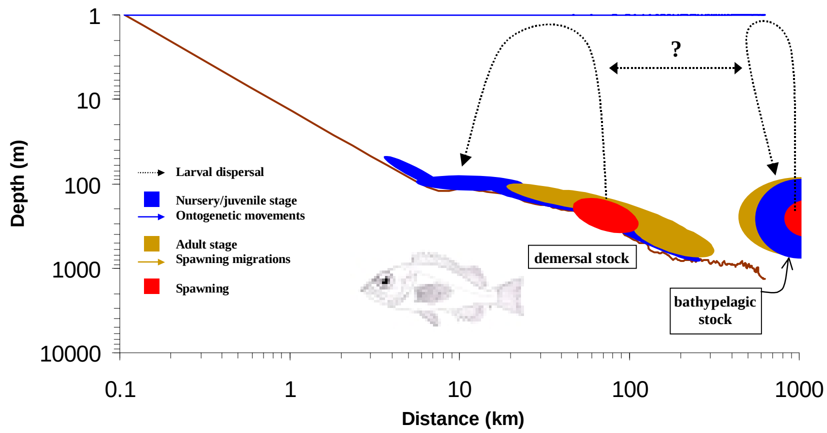


Figure 8. Generalized life history pattern by depth zone for Irminger Sea deepwater redfish stocks (benthic and mesopelagic *Sebastes mentella*). Brown line represents typical depth transect from approx. 63ºN, 22ºW to 59ºN, 30ºW.

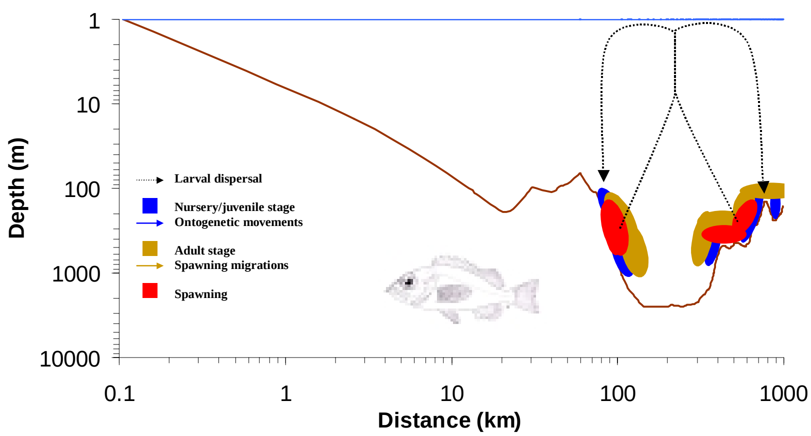


Figure 9. Generalized life history pattern by depth zone for Barents Sea deepwater redfish (*Sebastes mentella*). Brown line represents typical depth transect from approx. 68ºN, 13ºE to 76ºN, 18ºE.

Similarly, a third species (*S. fasciatus*, Ascadian redfish) occurs in the western Atlantic, also primarily in shallow waters in inshore areas (mainly 10-30 m depth, Kelly and Barker 1961 in Kenchington 1991), and is very common in the Gulf of Maine (Scott and Scott 1988). Here we concentrate on the first species, the deepwater redfish *S. mentella* (Figures 8 and 9). As its common name suggests, *S. mentella* is a deepwater, predominantly benthic species that rises off the bottom during the night (Scott and Scott 1988). However, mesopelagic groups have been documented in the Irminger Sea (Figure 8), and might represent separate stocks (Bel'skiy et al. 1987, Christensen and Pedersen 1989). Depth range of occurrence for *S. mentella* is 130-900 m (FishBase 1999), and stocks often show stratification by depth, with smaller individuals generally more shallow (Christensen and Pedersen 1989). Immature individuals have been recorded widely distributed down to 500 m (Drevetnyak 1993). However, no change in average length with depth was recorded for depths between 150-200 m, but average size did increase with depths > 200m (Magnusson et al. 1990). Redfish are ovoviviparous and larvae are born (extruded) at approximately 7 mm size after absorbing the eggsack. During the larval extrusion period adults were found to concentrate in the 250-700 m depth range (Drevetnyak 1993), with the majority of extrusions occurring at 250-400 m depth (Magnusson et al. 1990, Mukhina et al. 1992, 1995). The larval stage is pelagic in surface waters in 0-50 m depth (Christensen and Pedersen 1989, Herra 1989, Mukhina et al. 1992). Nursery areas are found mostly at depths between 50 and 350 m (Anon. 1998). At approximately 25 mm in size they start moving into deeper waters (Christensen and Pedersen 1989). Redfish grow to 7-8 cm during first year, thereafter approximately 2.5 cm per year until about 10 years of age, after which growth slows down (Scott and Scott 1988). Sexual maturity is thought to be reached at 8-10+ years of age, with a longevity of 40+ years (Christensen and Pedersen 1989).

DISCUSSION

The visualization of two-dimensional life history patterns, using the coastal transect method presented here, represents only a small component of the assessment of ecosystem effects of fishing undertaken by the *Sea Around Us Project*. Application of this method in the context of this project will require drawings of similar transects for the major commercial species of the North Atlantic, as defined by those species contributing 90% of the FAO database landings for FAO areas 21, 24 and parts of 31 and 34. This list will be augmented by species of regional significance based on 90% of the landings in the ICES and NAFO databases (e.g American lobster). It is anticipated that this might result in 40-50 species.

The purpose of these generalized life-history transects is not to present a detailed, quantitative depth distribution analysis. However, these graphics lend themselves to inclusion of such quantitative data in the form of vertical and horizontal data graphics that can be incorporated into the existing transects. Such quantitative information can be obtained from various sources, such as depth stratified survey data (e.g. Mahon and Sandeman 1985, Mahon et al. 1998).

Within the framework of the project, these coastal transect distributions will help assign catches to areas such as those described in the classification systems of the Large Marine Ecosystems (Sherman and Duda 1999) and 'biogeochemical provinces' (Longhurst 1995). A consensus synthesis approach to these classification systems is being considered by the *Sea Around Us Project* (see Pauly et al. 2000). The catch data allocation algorithm may also use augmentative data on geographic distribution and quantitative depth information where available (e.g. cumulative distribution frequency curves in Perry and Smith 1994). Within the context of the *Sea Around Us Project*, present day (1990s) as well as 'historic' transects (1950-60s) may need to be produced for stocks whose range of distribution may have changed significantly (e.g. Norwegian Spring Spawning herring present day versus 1970s, Figures 2 & 3). Additional information, such as seasonal variation in distribution or temperature iso-lines can also be accommodated, for example through multi-panel graphics. However, given the temporal and spatial scale of interest in this project (annual ecosystem models of large marine ecosystems) the present generalized 'snapshot' covering a distinct time period (e.g. 1990s) is considered appropriate.

Furthermore, Ruttan et al. (2000) will provide a method for assessing the yield and economic benefits gained and foregone through non-optimal use of resources by each fisheries sector (small scale versus large scale, incorporating gear type, vessel size and area of operation) for the different areas in the North Atlantic. The area and species specific information on the various scales of operation of different fishery sectors can thus be visually superimposed on the coastal transects, and coastal transects of fish distributions be used to show how different species ‘connect’, through their life history patterns, different fisheries sectors, such as small with large scale fisheries (e.g. inshore versus offshore).

Thus, we consider the present approach may be useful for visualizing the existence, interaction and potential conflicts between different fishery sectors for species or stocks whose life history patterns illustrate the need for improved integration of management of the different fishery sectors. This may apply in particular to rationalizations of overcapitalized fisheries. The proposed visualization may be used by management to incorporate the concept of life history interconnectivity between different fishery sectors and may assist in the formulation of more informed policy options for ecosystem-based management of North Atlantic fisheries.

Acknowledgements

We would like to thank the Environment Project of The Pew Charitable Trusts for their funding of the *Sea Around Us Project*. Special thanks to Drs J. Blaxter, K. Frank and M. Sinclair for their reviews and critical comments, which greatly improved this contribution. Thank you also to the reviewers participating in the Sea Around Us Method Review workshop. Drs Tony Pitcher, Reg Watson and Lore Ruttan provided assistance with the depth transects and commented on an earlier version of this paper. Particularly thanks to Dr Konstantin Drevetnyak from the Russian Polar Institute, Murmansk, for sharing his knowledge on Barents Sea Redfish.

References

Anderson, J.T., Dalley, E.L. and Carscadden, J.E. (1995) Abundance and distribution of pelagic 0-group cod (*Gadus morhua*) in Newfoundland waters: inshore versus offshore. *Can. J. Fish. Aquat. Sci.* 52:115-125.

Anon (1998) Report of the study group on redfish stocks. Living Resources Committee. ICES CM 1998/G:3

Barrett, P.H., Gautrey, P.J., Herbert, S., Kohn, D. and Smith, S. (eds) (1987) Charles Darwin’s Notebooks, 1836-1844: geology, transmutation of species, metaphysical enquiries. British Museum (Natural History), London and Cornell University Press, Ithaca and New York, 747 pp.

Bel'skiy, Y.I., Gerber, E.M., Demenin, A.A. and Safronov, A.M. (1987) On the vertical distribution of the deepwater redfish, *Sebastes mentella*, in the oceanic pelagic zone of the North Atlantic. *J. Icht.* 27(6), 96-100.

Blaxter, J.H.S. (1985) The herring: a successful species. *Can. J. Fish. Aquat. Sci.* 42 (suppl. 1), 21-30.

Christensen, V. and Pedersen, J. (1989) White fish resources and catch possibilities in the North Atlantic for foreign vessels. Danish Fisheries Technology Institute Report. 42 pp.

Cohen, D.M., Inada, T., Iwamotot, T. and Scialabba, N. (1990) FAO species catalogue. Vol 10. Gadiform fishes of the world (Order Gadiformes). An annotated and illustrated catalogue of cods, hakes, grenadiers and other gadiform fishes known to date. FAO Fish. Synop. (125, Vol 10), 442 pp.

Dragesund, O., Hamre, J. and Ulltang, O. (1980) Biology and population dynamics of the Norwegian spring spawning herring. *Rapp. Counc. Int. Explor. Mer.* 177, 43-71.

Drevetnyak, K.V. (1993) Russian investigation of redfish (*Sebastes mentella*) from the Norwegian-Barents Sea stock in 1992. ICES CM 1993/G:61.

FAO (1972) Atlas of the living resources of the seas. FAO Fisheries Circular No. 126.

FishBase (1999) A global information system on fishes. http://www.fishbase.org

Frank, K.T., Drinkwater, K.F. and Page, F.H. (1994) Possible causes of recent trends and fluctuations in Scotian Shelf/Gulf of Maine cod stocks. *ICES mar. Sci. Symp.* 198, 110-120.

Fyhn, U.E.H., Brix, O., Naevdal, G. and Johansen, T. (1994) New variants of the haemoglobins of Atlantic cod: a tool for discriminating between coastal and Arctic cod populations. *ICES mar. Sci. Symp.* 198, 666-670.

Garstang, W. (1909) The distribution of the plaice in the North Sea, Skagerrak and Kattegat, according to size, age, and frequency. *Rapp. Counc. Int. Explor. Mer.* 1, 136-138.

Heincke, F. (1913) Investigation on the plaice. General Rapport I. Plaice fishery and protective regulation. Part I. Rapp. P.-v. Réun. *Counc. Int. Explor. Mer.* 17A, 1-153.

Helle, K. (1994) Distribution of early juvenile Arcto-Norwegian cod (*Gadus morhua* L.) in relation to food abundance and watermass properties. *ICES Mar. Sci. Symp.* 198, 440-448.

Herra, T. (1989) Larval redfish drift migration in relation to hydrographic features. Rapp. P.-v. Réun. *Cons. int. Explor. Mer.* 191, 92-99.

Holst, J.C., Arrhenius, F., Hammer, C., Hakansson, N., Jacobsen, JA., Krysov, A., Melle, W. and Vilhjalmson, H. (1998) Report on surveys of the distribution, abundance and migrations of the Norwegian spring-spawning herring, other pelagic fish and the environment of the Norwegian Sea and adjacent waters in late winter, spring and summer of 1998. ICES CM 1998/D:3.

Houde, E.D. and Zastrow, C.E. (1993) Ecosystem- and taxon-specific dynamics and energetics properties of fish larvae assemblages. *Bull. Mar. Sci.* 53(2), 290-335

Iles, T.D. (1971) Growth studies on North Sea Herring III: The growth of East Anglican herring during the adult stage of the life history for the years 1940 to 1967. *J. Cons. int. Explor. Mer.* 33, :386-420.

Iles, T.D. and Sinclair, M. (1982) Atlantic herring: stock discreteness and abundance. *Science* 215, 627-633.

Kenchington, T.J. (1991) Vertical distribution and movements of larval redfishes (*Sebastes* spp.) in the southern Gulf of St. Lawrence. J. *Northw. Atl. Fish. Sci.* 11, 43-49.

Loken, S., Pedersen, T. and Berg, E. (1994) Vertebrae numbers as an indicator for the recruitment mechanism of coastal cod of northern Norway. *ICES mar. Sci. Symp.* 198, 510-519.

Longhurst, A. (1998) Cod: perhaps if we all stood back a bit? *Fish. Res.* 38, 101-108.

Magnusson, J., Kosswig, K. and Magnusson, J.V. (1990) Further studies on young redfish in the east Greenland shelf area. ICES CM 1990/G:43

Mahon, R. and Sandeman, E.J. (1985) Fish distributional patterns on the continental shelf and slope from Cape Hatteras to the Hudson Strait – A trawl's eye view. In: R. Mahon (ed) Towards the inclusion of fishery interactions in management advice. *Can. Tech. Rep. Fish. Aquat. Sci.* No. 1347, 137-152.

Mahon, R., Brown, S.K., Zwanenburg, K.C.T., Atkinson, D.B., Buja, K.R., Claflin, L., Howell, G.D., Monaco, M.E., O'Boyle, R.N. and Sinclair, M. (1998) Assemblages and biogeography of demersal fishes of the east coast of North America. *Can. J. Fish. Aquat. Sci.* 55, 1704-1738.

McKeown, B.A. (1984) Fish Migration. Croom Helm, London. 224 pp.

Mukhina, N.V., Drevetnyak, K.V. and Dvinina, E.A. (1992) Redfish spawning grounds in the Barents Sea and adjacent waters. ICES CM 1992/G:54

Mukhina, N.V., Drevetnyak, K.V. and Nikiforov, A.G. (1995) Influence of oceanographic fields on the abundance and distribution of *Sebastes mentella* from the Barents Sea stock and of Arcto-Norwegian cod at early life history. ICES CM 1995/Q:14.

Munro, G. and Sumaila, U.R. (2000) Restoration of overexploited capture fishery resources: an economic/ecosim modelling approach. In: Pauly, D. and Pitcher, T.J. (eds) Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. *Fisheries Centre Research Reports* 8(1), 106-113.

Muus, B.J. and Dahlstrøm, P. (1977) Sea fishes of Britain and North-western Europe. Collins, London. 244 pp.

Pauly, D. (1982) The fishes and their ecology. In: Pauly, D. and Mines, A.N. (eds). Small-scale fisheries of San Miguel Bay, Philippines: biology and stock assessment. ICLARM Technical Report 7, 15-33.

Pauly, D. (1994) A framework for latitudinal comparisons of flatfish recruitment. *Neth. J. Sea Res.* 32(2), 07-118.

Pauly, D. and Lightfoot, C. (1992) A new approach for analyzing and comparing coastal resource systems. *NAGA* 15(2):7-10.

Pauly, D. and Pitcher, T.J. (2000) Assessment and mitigation of fisheries impacts on marine ecosystems: A multidisciplinary approach for basin-scale inferences, applied to the North-Atlantic. In: Pauly, D. and Pitcher, T.J. (eds) Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. *Fisheries Centre Research Reports* 8(2), 1-12.

Pauly, D., Christensen, V., Froese, R., Longhurst, A., Platt, T., Sathyendranath, S., Sherman, K. and Watson, R. (2000) Mapping fisheries onto marine ecosystems: A proposal for a consensus approach for regional, oceanic and global integrations. In: Pauly, D. and Pitcher, T.J. (eds) Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. *Fisheries Centre Research Reports* 8(2), 13-22

Perkins, H.C., Chenoweth, S.B. and Langton, RW. (1997) The Gulf of Maine Atlantic cod complex, patterns of distribution and movement of the Sheepscot Bay substock. *Bull. Nat. Res. Inst. Aqua.* 0(Suppl. 3), 101-107.

Perry and Smith (1994) Identifying habitat associations of marine fishes using survey data: an application to the northwest Atlantic. *Can. J. Fish. Aquat. Sci.* 51, 589-602.

Rottingen, I. (1990) A review of variability in the distribution and abundance of Norwegian spring spawning herring and Barents Sea capelin. In: Erikstad, K.E., Barrett, RT. and Mehlum, F. (eds), What determines the distribution of seabirds at sea? Polar Res, 33-42.

Ruttan, L.M., Gayanilo, F.C., Sumaila, RS. and Pauly, D. (2000) Small scale versus large scale fisheries: a multipecies, multifleet model for evaluating their potential to conserve fishes and fisheries. In: Pauly, D. and Pitcher, T.J. (eds) Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. Fisheries Centre Research Reports 8(2), 64-78.

Scott, W.B. and Scott, M.G. (1988) Atlantic Fishes of Canada. University of Toronto Press. 731 pp.

Serchuk, F.M., Grosslein, M.D., Lough, RG., Mountain, DG. and O'Brien, L. (1994) Fishery and environmental factors affecting trends and fluctuations in the Georges Bank and Gulf of Maine Atlantic cod stocks: an overview. *ICES mar. Sci. Symp.* 198, 77-109

Sinclair, M. and Iles, T.D. (1985) Atlantic herring (*Clupea harengus*) distributions in the Gulf of Maine-Scotian Shelf area in relation to oceanographic features. *Can. J. Fish. Aquat. Sci.* 42(5), 880-887.

Sinclair, M., Anthony, V.C., Iles, TD. and O'Boyle, RN. (1985) Stock assessment problems in Atlantic herring (*Clupea harengus*) in the Nortwest Atlantic. *Can. J. Fish. Aquatic. Sci.* 42, 888-898.

Slotte, A. and Fiksen, O. (2000) State-dependent spawning migration in Norwegian spring-spawning herring. *J. Fish Biol.* 56, 138-162.

Svendsen, Y.S. (1995) Tracking juvenile cod (*Gadus morhua* L.) in northern Norway using acoustic tags. *Fisheries research* 23 (3-4), 311-318.

Tufte, E.R. (1983) The visual display of quantitative information. Graphics Press, Cheshire, Connecticut. 197 pp.

Tort, P. (ed) (1996) Dictionnaire du Darwinisme et de l' évolution. Volume 2. Presses Universitaires de France.

Small Versus Large-scale Fisheries: A Multi-species, Multi-fleet Model For Evaluating Their Interactions And Potential Benefits

L. M. Ruttan*, F. C. Gayanilo Jr., U. R. Sumaila* and D. Pauly*

Fisheries Centre, University of British Columbia, Vancouver, B.C., Canada, V6T1Z4
Present address:
Workshop in Political Theory and Policy Analysis, Indiana University, 513 N. Park Ave., Bloomington IN, 47408-3895 USA

Abstract

In this paper, we present a method for evaluating the economic losses and biological impacts of a lack of co-ordination of effort on the part of small versus large-scale fisheries. We illustrate our method using fisheries of the Gulf of Maine and the George's Bank (USA). There are several novel methodological components of this work. First, we use an approach for defining which fisheries are small and which are large on a scale that is specific to political units since gear that is large-scale in one country may be categorized as small-scale in another. Second, we present a multi-species, multi-fleet, yield-per-recruit model that incorporates gear selection curves for each gear type. This permits an evaluation of the economic benefits of trade-offs in effort between the two small and large-scale fleets. Optimal combinations of effort by the two fleets are identified by subtracting costs of fishing effort from the gross value calculated by the model. Third, we estimate the value of foregone profits by comparing the rents produced at such an optimum with those produced by the current fishery. Finally, we identify a Nash bargaining solution that would be obtained if both sectors chose to cooperate by coordinating their levels of effort.

Introduction

Throughout the world, fishing fleets are becoming progressively larger in scale and fisheries are becoming serially depleted. Vessels are becoming larger and faster, are traveling farther and farther from their homeports, are using more sophisticated (and expensive) technologies and are catching fish in shorter periods of time. The economic incentives for this trend are well understood. The open access nature of past fisheries clearly invited overcapitalization (Gordon 1954). These incentives persist in most modern day 'regulated access' and 'regulated restricted access' fisheries (Wilen and Homans 1997) and even in fisheries with individual quotas (Maurstad 2000). In addition, present declines in fish abundance requires indebted fishers to search ever further for fish. Government subsidies, based on the presumption that large-scale operations enjoy greater economies of scale, further accelerates this trend (see Milazzo 1998 for a general discussion of subsidies). However, evidence for the greater economic efficiency of large-scale gears is inconclusive (P. Tyedmers pers. comm. with respect to fuel efficiency) and there are clear social costs to these trends that are borne both by individuals and by society as a whole. Few studies consider the full range of hidden costs when assessing the desirability of supporting one or the other fishing sectors. A more detailed treatment of the 'ecological footprint' with respect to fuel inputs of each type of fishery is presented in Tyedmers (2000).

In this paper, we compare the economic profitability of small and large-scale sectors by identifying what combination(s) of effort by these operations generates the highest gross and net revenues. Our method of analysis is a multi-species, multi-fleet, value-per-recruit model that has been developed expressly for this purpose (Figure 1) (see appendix for details). Our analysis indicates that optimal combinations of effort differ greatly depending on whether net or gross returns are considered and hence, it is critical to incorporate cost estimates when evaluating management plans. Second, our use of a bio-economic approach allows us to estimate the rents lost to society when non-optimal levels of effort are applied to the fishery, or in other words, when the small and large-scale sectors do not cooperate. Having determined the optimal combination of effort obtainable if sectors do cooperate, we are able to identify a Nash equilibrium with side payments[1] using a modified version of the method developed by Nash (1953) and refined by Munro (1979). Our method differs in that we consider two sectors of a fishery rather than two countries competing for a trans-boundary resource and we use a multi-species

[1] A Nash equilibrium is one where each individual should *not* wish to change strategy even if the other player does. In cooperative games with asymmetrical payoffs, players may reach agreements whereby one player pays the other some portion of the benefits obtained though cooperation.

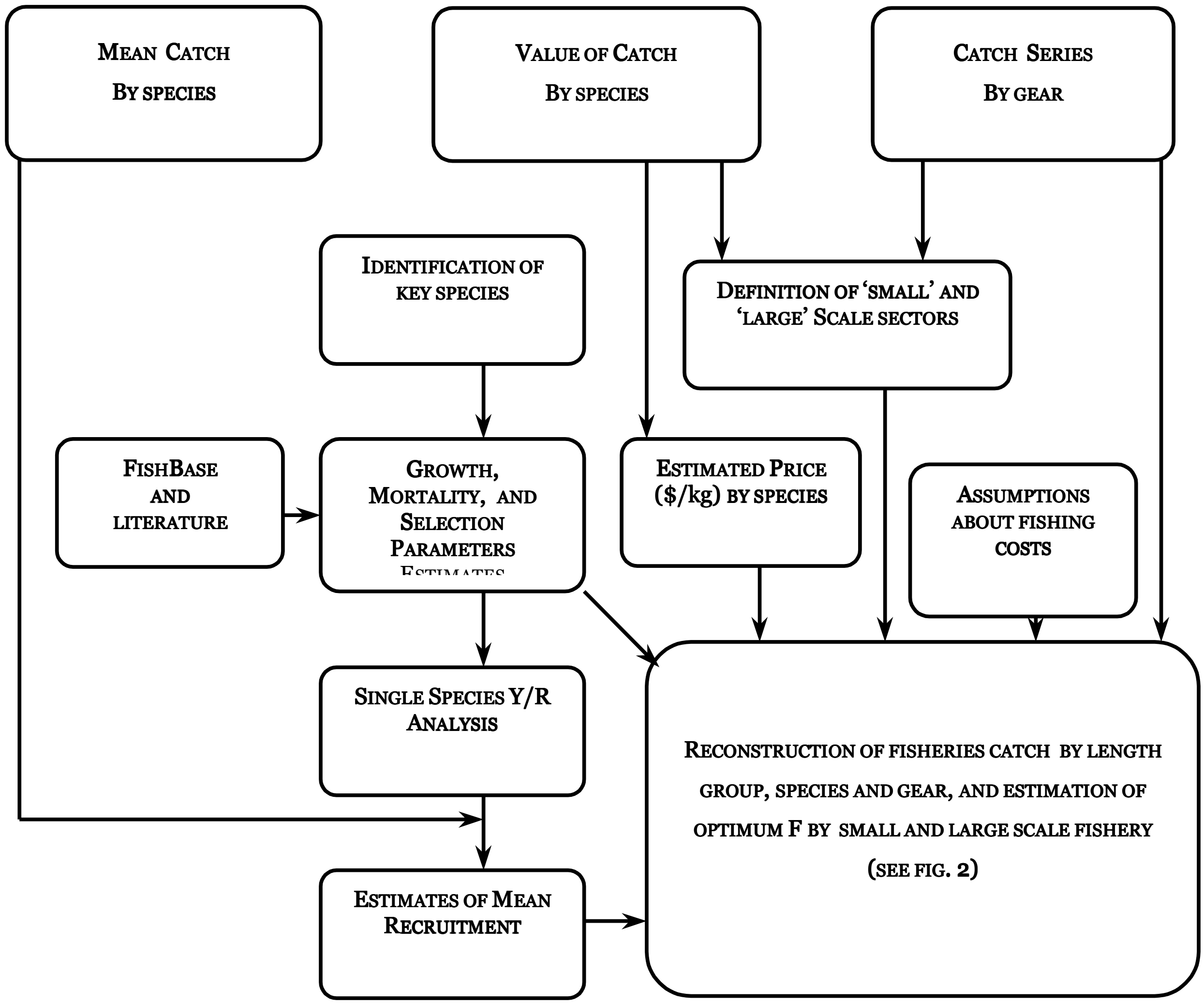


Figure 1. Flowchart of operations implied in the method described in this contribution.

value per recruit model rather than a surplus production model to predict optimal levels of effort (see also Sumaila 1997).

Here we illustrate this method using data from the Gulf of Maine and George's Bank (USA). Numerous assumptions must be made, chief among them that there is constant recruitment, that costs scale with effort, and that rents are driven to zero, any of which may distort the results of any one analysis but may not be problematic when this method is applied to a large number of fisheries since overestimates in one may compensate for underestimates in another.

MATERIALS AND METHODS

Gulf of Maine and George's Banks as a Study Area

The Gulf of Maine is a deep and cold body of water bounded on the South and West by the US states of Massachusetts, New Hampshire, and Maine, on the North by the Canadian provinces of New Brunswick and Nova Scotia, and on the East by the George's Banks. The latter is a shallow water bank rising at the edge of the continental shelf and capable of very high productivity (Sissenwine et al. 1984). Nearly 140 species are landed in the US states bordering the Gulf bringing in a total value of close to $650 million per year during the 1990s However, the once abundant sea life in both areas has been progressively depleted and the average trophic level of catches is declining (Steneck 1997). According to Steneck, where predatory

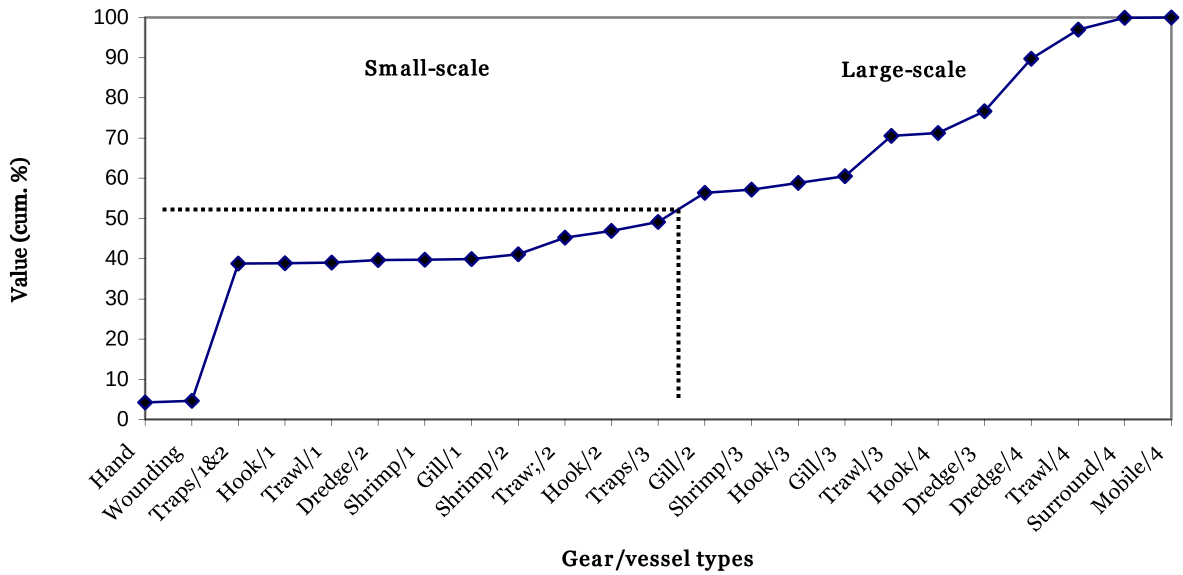


Figure 2. Cumulative % of landings value, by gear/vessel type first ranked from 'small' to 'large', with 50% line separating small from large-scale fisheries (see text).

groundfish were once the dominant species, now less valuable sculpins, dogfish and skates constitute the majority of the finfish catch. Data from the US National Marine Fisheries Service (NMFS) indicates that lobster, though always an important fishery, is now the largest fishery both in terms of value and tonnage and debate exists as to whether the current abundance is due to environmental changes or declines in the abundance of cod, one of its principal predators.

Source of Landings Data

Commercial and recreational landings recorded by gear and species from the states of Maine, New Hampshire and Massachusetts during the period 1989-98 were downloaded from the NMFS Office of Statistics and Technology website (http://www.st.nmfs.gov). The landings and their values were each averaged across the time period and then summed over the three states. These figures are henceforth referred to as 'total catches' or 'total values'. Undoubtedly a portion of these landings are caught in areas besides the Gulf of Maine and George's Banks. However, there is no simple method for apportioning them. It should be emphasized that no data on catch at length, or value of various size classes was available from the source we consulted, nor from other sources.

Defining Fishing Scale

There is no single definition of what is a small-scale fishery. However, the term is popularly used in reference to subsistence and artisanal fisheries, the latter typified by small, multi-species catches that are caught using small vessels taken on short fishing trips (Charles 1989, Munro 1980). Based on such a definition, we would find that most small-scale fisheries are found in the 'South', i.e. developing countries, while most large-scale fisheries are in the developed 'North'. On consideration it is clear, however, that many inshore fisheries in the developed world are much smaller in scale than the largest fisheries in those same areas. Thus we choose to categorize fisheries as small or large on a relative rather than absolute scale. The particular scale we use is catch per vessel per year. Our justification is that low catches are associated with smaller boats that travel shorter distances. Thus this scale captures the essence of 'smallness' with just one figure, although smallness also implies smaller crew, more limited range, etc.

In practice, we define particular fisheries as gear/vessel combinations. There are three steps. First, we categorized all fisheries as belonging to one of ten categories of gear and one of four categories of vessel size using the same tonnage categories as the NMFS (Table 1) and gear categories that are nearly identical to those used by Watson et al. (2000). The categories differ from theirs in that available data on catch rates necessitated aggregating bottom and mid-water trawlers, and permitted shrimp trawlers to be given their own categories; it was desirable to keep shrimp trawlers separate from other trawlers because they use different mesh size. Second, gear/vessel combinations were ranked in ascending order according to annual catch per vessel. Third, a cumulative percentage distribution is constructed with these ranked fisheries. The group of fisheries that provides the first 50% of landed value are then classified as 'small-scale' and the remainder as 'large-scale'

(Figure 2). The fisheries are divided into just two groups, large and small, using a cutoff point of 50% of cumulative landed value in order to provide a standard for comparison involving other variables such as employment, fuel consumption, etc. Finally, the cut-off point is examined to see whether all gears employed by the same size category vessels fall on the same side of the cut-off and the rankings adjusted if they do not. The justification for this is that many fishers, especially small-scale ones, use multiple gear types on the same boat.

To accomplish the second step, data on annual catch per vessel were obtained from the Status of the Fishery Resources of the Northeastern United States (NOAA n.d.) which contained such information for the years 1994-96 and the entire Northeast region of the US (covering all states managed by the New England and the Mid-Atlantic Fishery Management Councils). The data for some categories of vessel size were not available from this source due to there being low numbers of vessels in these categories (and thus sampling and privacy issues). However, information on landed value and numbers of vessels was available for the missing categories. These latter figures were used to estimate annual catch per vessel for the missing vessel size categories.

There were five categories of gear for which no information on catch/vessel/year was available. Fortunately, two of these (hand gear and wounding/grappling gear) would clearly have the lowest catch rates and hence they were inserted into the beginning of the ranking. A third category, traps 1&2, was also assumed to be relatively small in scale and hence was inserted just after wounding gear. (Note that it was assumed that all trap vessels of size class 3 were offshore lobster vessels, a category for which catch/year/vessel was available). The other two categories (mobile seines and surrounding gear) can reasonably be assumed to have high if not the highest catches/vessel/year and thus they were inserted into the end of the rank order.

After ranking the gear/vessel combinations based on the 1994-96 data, each combination's total value for the 1989-98 period was calculated by partitioning the total value for a particular gear type among vessel categories in the following manner. Since total value for the 1994-96 period was available for both gear and vessel categories, it was possible to calculate the percentage of value that each vessel category produced for a given gear type. These percentages were then multiplied by the 1989-98 total values for each gear type. Once these figures were calculated it was possible to complete the third step of the process of defining which fisheries are small by constructing a cumulative percentage distribution using the ranked gear types and their associated total values. We used total value rather than total landings for pragmatic reasons. Since the smallest scale gears typically have small but highly valued catches, the use of tonnage would lead to two thirds of gears being classified as small scale. This is intuitively wrong. We therefore deliberately chose to use value and thereby minimize the numbers of gears that we consider to be small scale. In this particular case, the classification of the two boundary fisheries (traps/3 and gill/2) were switched so that all gears employed be size class 2 vessels are defined as small-scale and all size class 3 vessels are large-scale.

Table 1. Gear and vessel categories.

Code	Gears
11	Shrimp trawl
12	Bottom or midwater trawl
21	Mobile seines
31	Surrounding nets (e.g. purse seines)
41	Gillnets and entangling nets
51	Hooks and lines
61	Traps and lift nets
71	Dredges
81	Grappling and wounding (e.g. harpoons)
90	Other gear (e.g. hand lines, hoes etc.)
	Vessels (gross registered tons)
1	<5
2	5 - 50
3	51-149.9
4	150+

Species Characteristics

Landings and Recruitment by species

The species included in this model were chosen by ranking all species landed in the three states from highest to lowest in terms of total value of landings. A cumulative percentage distribution of total values for each species was calculated and the species generating the first 95% of the value of all landings were included in the initial sample (33 species). All sessile species as well as 2 species of worms were then removed. Three additional species are not included in the analysis due to a paucity of easily available information on their population dynamics (American eel, *Anguilla rostrata*; bay scallop, *Argopecten irradians*; and hagfish *Myxine glutinosa*) leaving a final sample of 21 species (Table 2).

Fifty-seven different gears were listed in the initial data set. These were aggregated into the ten gear categories described in Table 1.

Table 2. Total observed landings, value, US $/kg, fishing mortality (F, year $^{-1}$), mean length at first capture size (L_c, cm), and number of recruits for species included in the model.

Common Name	Scientific Name	Landings (t)	Value ($$10^6$)	$/kg	F	L_c	Recruits (10^6)
BASS, STRIPED	*Morone saxatilis*	987.7	3.1	3.15	2.0	67.1	4.7
COD, ATLANTIC	*Gadus morhua*	30,723.1	53.7	1.74	2.0	103.6	22.7
FLOUNDER,SUMMER	*Paralichthys dentatus*	588.1	2.4	4.01	5.0	109.6	0.2
FLOUNDER,WINTER	*Pseudopleuronectes americanus*	4,689.9	13.6	2.64	3.0	35.2	42.4
FLOUNDER,WITCH	*Glyptocephalus cynoglossus*	2,009.8	7.5	3.45	2.0	32.8	157.8
FLOUNDER,YELLOWTAIL	*Limanda ferruginea*	4,475.5	11.3	2.59	2.5	35.0	87.0
GOOSEFISH	*Lophius americanus*	14,521.8	18.7	1.31	2.0	137.9	9.3
HADDOCK	*Melanogrammus aeglefinus*	1,618.1	4.2	2.62	3.0	65.6	5.0
HAGFISH	*Myxine glutinosa*	1,746.1	1.1	0.95	0.0		
HAKE, WHITE	*Urophycis tenuis*	6,031.3	6.5	1.11	3.0	58.8	7.6
HERRING, ATLANTIC	*Clupea harengus*	65,211.7	8.0	0.24	3.0	25.2	5449.0
LOBSTER, AMERICAN	*Homarus americanus*	24,158.0	155.3	7.68	1.5	15.2	83.8
PLAICE, AMERICAN	*Hippoglossoides platessoides*	4,467.1	11.4	2.40	2.5	48.1	44.8
POLLOCK	*Pollachius virens*	8,953.8	11.9	1.30	2.5	85.6	11.4
SCALLOP, SEA *	*Placopecten magellanicus*	6358.0	69.4	13.07	1.5	12.8	578.7
SCUP	*Stenotomus chrysops*	23.9	0.1	4.11	4.0	29.7	0.5
SEA URCHIN	*Strongylocentrotus droebachiensis*	12,515.0	22.1	1.45	2.0	15.1	14342.7
SHARK, SPINY DOGFISH	*Squalus acanthias*	18,354.8	6.0	0.34	2.0	57.7	206.3
SHRIMP, NORTHERN	*Pandalus borealis*	4,730.2	8.9	2.89	3.5	12.2	6303.6
SQUID, LONGFIN	*Loligo pealeii*	1,706.4	2.1	1.35	5.0	26.8	398.9
SWORDFISH	*Xiphias gladius*	1,259.0	7.9	6.60	2.5	186.9	0.1
TUNA, BLUEFIN	*Thunnus thynnus*	929.6	16.9	12.38	2.0	280.8	0.0
Total		**209,700.8**	**442.2**				**27,756.6**

- landings converted from meat to shell weight using a 1:9 ratio (Caddy 1989)

Table 3. Parameters used in calculations of yield per recruit and sources of information.

Common Name	L_∞ (cm)	K (yr^{-1})	W_∞ (g)	a	b	M (yr^{-1})	L_r (cm)	t_0 (yr)	T °C	FishBase Population or References
BASS, STRIPED	95.8	0.188	5,440	0.006	2.907	0.29	4.80	0.000	10.0	Coos Bay
COD, ATLANTIC	148.0	0.121	36,600	0.007	3.101	0.18	7.40	0.000	10.0	Gulf of Maine/George's Banks
FLOUNDER, SUMMER	137.0	0.843	29,949	0.007	3.117	0.65	6.90	0.000	10.0	USA, Delaware Bay, 1966-71
FLOUNDER, WINTER	44.0	0.400	1,380	0.021	3.000	0.39	2.20	0.000	5.0	Canada, East Coast
FLOUNDER, WITCH	46.9	0.150	786	0.002	3.390	0.29	2.30	0.000	10.0	Norway, Hekkingen, Malangen
FLOUNDER, YELLOWTAIL	50.0	0.335	1,183	0.009	3.000	0.48	2.50	0.000	10.0	USA, South New England
GOOSEFISH	197.0	0.060	53,952	0.017	3.000	0.11	9.90	-0.080	10.0	Canada, Bay of Fundy
HADDOCK	72.9	0.352	4,214	0.011	3.000	0.44	3.60	0.000	9.4	Gulf of Maine
HAKE, WHITE	84.0	0.218	13,685	0.004	3.147	0.17	6.80	-0.280	0.0	S. Gulf of St.Lawrence
HERRING, ATLANTIC	36.0	0.210	350	0.008	3.000	0.35	1.80	0.000	8.0	Norway, Atlanto-Scandian
LOBSTER, AMERICAN	25.3	0.056	13,783	0.003	3.015	0.13	6.00	-0.772	10.0	Campbell (1986); Estrella &McKiernan (1989 p.7 & 13); Townsend (1986)
PLAICE, AMERICAN	80.2	0.076	5,550	0.004	3.204	0.16	4.00	0.000	10.0	ICNAF Res.Div.3L 1969-72
POLLOCK	107.0	0.190	11,634	0.008	3.000	0.21	5.40	0.000	6.0	Norway, Norwegian Sea
SCALLOP, SEA	14.2	0.317	47	0.016	3.000	0.10	1.80	1.385	10.0	Caddy (1975 p.1316; 1989 p. 569)
SCUP	42.4	0.170	1,723	0.023	3.000	0.32	2.10	0.000	10.0	USA, Northwest Atlantic
SEA URCHIN	18.9	0.122	12	0.081	2.905	0.13	4.50	0.050	10.0	Longhurst and Pauly (1987); Russell et al. (1998 p. 46, 150); Swan (1958 p.512-13)
SHARK, SPINY DOGFISH	96.1	0.067	3,580	0.004	3.004	0.09	4.80	-5.000	10.0	Georgia Strait, BC
SHRIMP, NORTHERN	17.5	0.390	17	0.003	3.080	0.65	4.57	-0.100	10.0	Fournier et al. (1990 p.596); Haynes and Wigley (1969 p. 69, 74); Parsons and Frechette (1989 p. 74); Shumway et al. (1985 p.39)
SQUID, LONGFIN	38.30	0.590	103	0.046	2.118	0.87	2.10	0.000	10.0	Lange and Johnson (1981), Pauly (1985)
SWORDFISH	267.00	0.120	274691	0.014	3.000	0.15	13.40	-1.680	10.0	USA, Atlantic Coast
TUNA, BLUEFIN	468.00	0.050	1726165	0.037	2.870	0.15	23.40	0.000	12.0	USA, Cape Cod-Long Island

In the Gulf of Maine/George's Bank area this results in 23 different gear/vessel combinations. Since we had data on catch by gear type but not by vessel size, catches for each gear type were allocated to vessel categories by a method similar to that used to allocate values as described above (in *Defining Fishing Scale*). Most species had a category of 'uncoded' or multiple gears. These were given the same gear code as the most common gear used to catch that species. No dollar value was given for recreational fisheries and thus it was assumed that recreational and commercial fishermen using hook gear obtain the same price per kilogram.

Most growth parameters for finfish species were taken from FishBase 99 (Table 3). In cases where asymptotic weight (W_∞) was not available from FishBase (*Paralichthys dentatus*, *Stenotomus chrysops, Thunnus thynnus*) length and weight records were taken from Bigelow and Schroeder (1953) and used to calculate parameters a and b of length-weight relationships. These were then used to estimate W_∞ from the asymptotic length (L_∞). Mesh selection factors were determined by calculating each species' depth ratio from drawings available in FishBase 99 and in Bigelow and Schroeder (1953). The selection factor was then estimated using a nomogram in Pauly (1984, p. 11). Similar parameters for invertebrate species were gathered from the literature. Natural mortality estimates were obtained from FishBase 99 either as values associated with selected sets of growth parameters, or via the built-in estimation procedure based on the empirical equation of Pauly (1980), which uses L_∞, K and mean water temperature to estimate M. A value of 10°C was used as an input for all such estimates.

The numbers of recruits were obtained using Beverton and Holt yield per recruit analysis built into the spreadsheet software (Table 2). Inputs include the growth parameters described above as well as estimates of fishing mortality, F, and the mean length at first capture, L_c. The latter two parameters were, in turn, obtained from the FishBase 99 yield per recruit module. This module provides a graphical interface permitting one to easily identify the values of F and L_c associated with stable recruitment and the highest yield per recruit; these values were chosen since our goal is compare current rents against maximal possible rents.

Selection Characteristics

Each species/gear combination was assigned a mean length at first capture (L_{50}, in cm) that was either equal to the minimum legal size of capture (if available) or calculated using a known mesh size of that particular gear and the species' selection factor. It was assumed that the L_{50} of all non-mesh gear would be equal to the minimum legal size (e.g. hook gear, traps and pots, etc.). For some species, there is no minimum legal size and thus, for two of these (*Thunnus thynnus* and *Xiphias gladius*) an initial estimate of L_{50} was based on the minimum size caught in length-frequency data available from FishBase 99.

Selection and de-selection curves were calculated. On the selection side, values of L_{75} were calculated for finfish as equal to L_{50}*1.25 for all non-selective gear and equal to L_{50}*1.10 for all selective gear (here selective gears refers to size selectivity and include gillnets, hooks and lines, and traps; all other gears were considered non-selective). For invertebrates, $L_{75} = L_{50}*1.01$, the justification for this much steeper selection curve being that either the animals are hand picked out of the gear and there are minimum legal size limits (e.g. lobster, sea urchins and sea scallops) or that the nature of invertebrate body form justifies a steeper curve (e.g. shrimp and squid).

On the de-selection side, we set the D_{50} to be equal to L_∞*0.95 for non-selective gears and equal to L_∞*0.90 for selective gears. One exception is for lobster (*Homarus americanus*) where the D_{50} was set to be equal to the maximum legal size. Another exception was for cases where such a D_{50} ended up being smaller than the L_{50}. In these cases, the L_{50} was adjusted downwards. The D_{75} for all species is equal to D_{50} - 0.1cm.

Computation of Gross and Net Values

Finally, these species parameters, catches, recruitment values, classifications of gear as large or small, and selection curves for species/gear combinations were entered into a multi-species, multi-fleet spreadsheet solution (Figure 3). The gist of the model is to get around the lack of catch-at-length data by first constructing combined, weighted, selection curves for each sector's catch of each species. From natural mortality rates and using a selection curve, the relative distribution of population length can be estimated. This distribution is then 'raised' to allow for both natural mortality and the observed landings, which provides the corresponding pattern of fishing mortality at length ('F-pattern'), similar to Jones (1984). Then, effort applied to each species by each fleet can be varied systematically with an effort multiplier, the f-factor (see Appendix).

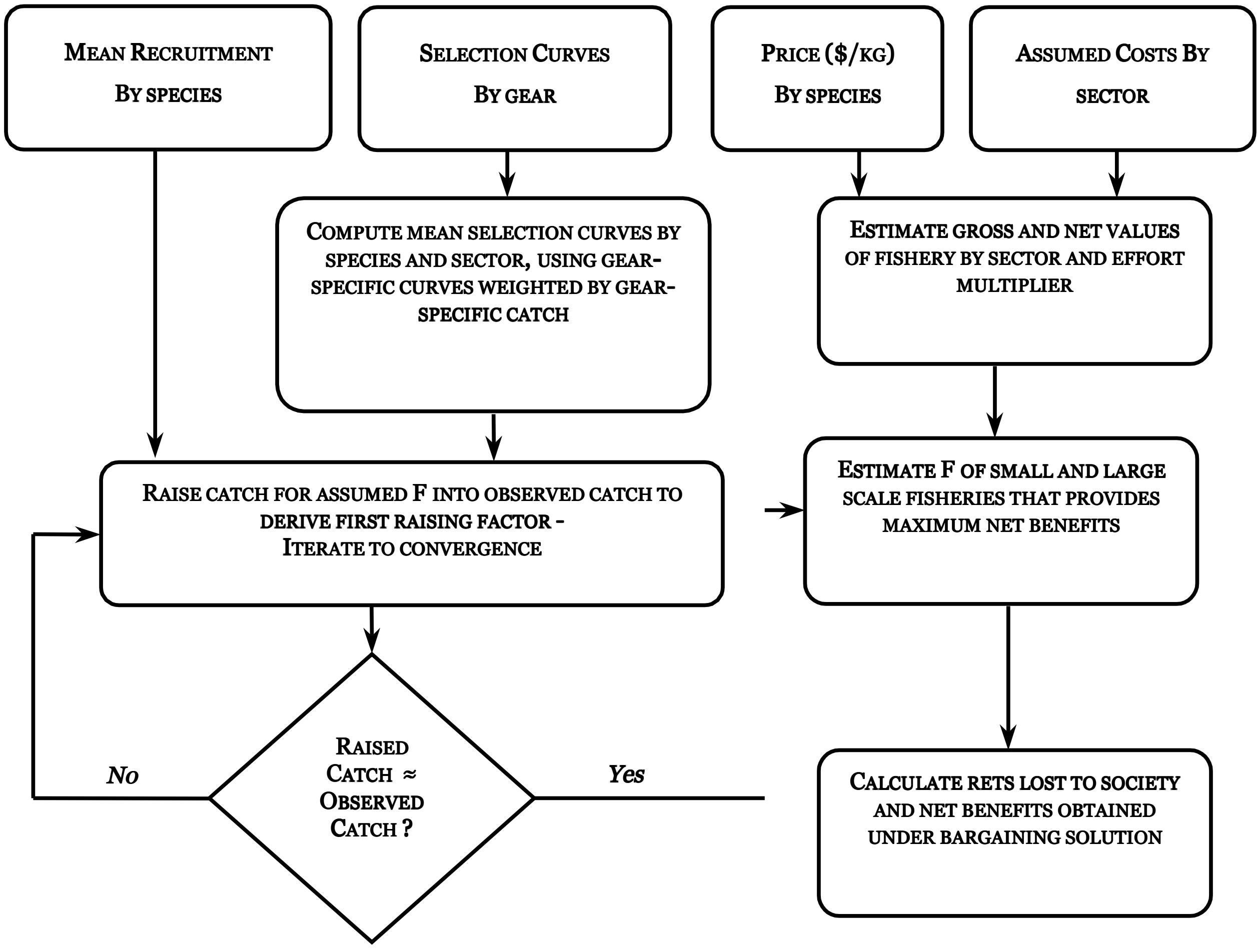


Figure 3. Procedures for calculating net returns for different fleet configurations (see text and Appendix)

The result is a matrix containing the aggregate gross value of the fishery for all combinations of effort on the part of small and large-scale fisheries. From this matrix we identify the combination(s) of effort leading to highest gross value(s). Net values are then calculated by subtracting costs from the gross values across the full range of f-factors of both sectors. In the absence of accurate data on fishing costs, we estimate them by assuming that the fishery is currently at its bio-economic equilibrium, i.e., that rents have been driven to zero (see below). If this is the case, the aggregate gross value (GV) of the current fishery is equal to the aggregate costs. By definition, the current gross value of the fishery is found at the point where the small and large scale f-factors are equal to one. Then, assuming that fishing costs scale linearly with fishing effort (see below), the net value (NV) for every combination of effort is estimated for the small scale sector as follows (where is l = large f-factor, and s = small f-factor):

Small NV(l,s) = Aggregate GV(l,s) –
SmallGV(1,1)* (s) ...1)

Similar calculations can be made to determine the aggregate and large-scale net revenues. From the matrix of net values we identify the combination of f-factors yielding the highest aggregate net returns.

Cost Sensitivity Analysis

Because accurate data on the costs of fishing effort are difficult to obtain, we have made simplifying assumptions: that rents are currently equal to zero and that costs scale linearly with effort. These assumptions are especially problematic if: a) rents deviate from zero in opposite directions in each sector and/or b) costs scale differently in each sector. We perform a cost sensitivity analysis to analyze what would happen to the optimal levels of effort if either of these two scenarios were the case. Four variants of our cost assumptions are analyzed. In the first variant, we ask what would happen if current small-scale rents are actually positive while large scale ones are negative. Specifically, we let the current small

NV = 5% of the small GV and the current large NV = -5% of the current large GV. The formula for determining net revenues in the case of the small sector is:

Small NV(l,s) = Aggregate GV(l,s) –
(SmallGV(1,1) – SmallNV(1,1))* (s) ...2)

The second variant is simply the reverse of the first; small net revenue is less than zero while large net is greater than zero. In the third variant, we let small scale costs scale at 95% of the f-factor while large scale costs scale at 105% of the f-factor. The formula for the small sector is:

Small NV(l,s) = Aggregate GV(l,s) –
(SmallGV(1,1)* (s)*0.95) ...3)

The fourth case is the reverse of the third case.

Nash Bargaining Solution

Two particularly interesting pieces of information can be drawn from the results. First, the value of rents foregone through non-cooperation by the large and small-scale fleets is equal to the value of the aggregate maximum since, by definition, the current rent is equal to zero. Second, Nash equilibria can be identified from among these points. In game theoretic terminology, the current state of the fishery serves as a threat point of a cooperative game (Nash 1953, in Munro 1979). The threat point gives us the payoff that each player can expect to take home if they do not cooperate. Assuming that the aggregate rents generated at our optimum are higher than at present (and thus Pareto optimal) and there is a single optimum point of highest aggregate net values, then, if the two sectors do choose to cooperate and co-ordinate their effort, and if side-payments are an acceptable solution, there is a Pareto frontier constituting the set of possible profits after various levels of side-payments have been made (imagine each axis of a graph as representing each sector's profits given that particular allocation of the 'extra' rent generated from cooperation). In this set of points, if the large [small] sector obtains a profit of x [y] without cooperating but obtains a proft of a [b] if they cooperate and side-payments are made, then the Nash bargaining solution for such a cooperative game is determined by choosing point (a,b) on the Pareto frontier so as to maximize the product of the difference between the payoffs received under cooperation and those received at the threat point:

$$\max(a-x)(b-y) \qquad \text{...4)}$$

Since by definition, x = y = 0 in our case, it can be shown that the values of x and y that maximize the product of a*b occurs when the extra benefits above the sum of threat point payoffs are shared equally between the participants, i.e. a = b (see also Luce and Raiffa 1967 in Munro 1979).

RESULTS

Table 4 presents the results of the yield per recruit analysis. Levels of the f-factor for each sector, and the gross and net revenues obtained in aggregate and by sector are given for the three points; the current scenario, the point at which gross revenues are maximized and the point at which net revenues are maximized. In no case are there multiple optima. Present effort is by definition at the point where f-factor large = f-factor small = 1, and again, by definition, net values are equal to zero.

Table 4. Results of yield per recruit analysis. Values are US$ (10^6).

	Aggregate	Small	Large
Current f-factor		1.00	1.00
Gross Value	235.37	62.68	172.69
Net Value	0.00	0.00	0.00
Gross Max f-factor		2.50	0.92
Gross Value	251.52	111.55	139.97
Net Value	-64.06	-45.16	-18.90
Net Max f-factor		0.11	0.35
Gross Value	174.89	18.27	156.60
Net Value	107.56	11.38	96.18
Payments		42.40	-42.40
After bargaining NV		53.78	53.78

We see that when only gross revenues are considered, the highest revenues are achieved when small-scale effort is at least 2.5 times the current effort and when large-scale effort is 0.92 times its current effort. This is not surprising given that the small-scale sector catches most of the highly valued invertebrates, e.g. lobster, sea urchins. When cost, and thus net values, are considered the optimal levels of effort change considerably; small-scale effort drops to 0.11 of its current value and large-scale effort drops to 0.35 of its current level. At this point the aggregate rents lost to society from over-fishing are equal to $107.56 million dollars. By coordinating effort levels, the small scale sector as a whole would stand to gain $11.38 million and the large scale sector $96.18 million. By co-ordinating effort and then in addition bargaining to share the proceeds from effort coordination, each sector could gain $53.78 million as a whole.

In table 5, we present the results of the cost sensitivity analysis. As described earlier, there are four different cases that we analyze. In the first two, we essentially change the intercept of the f-factor (x) versus cost (y) function. In variant 1 the intercept is lowered for the small-scale sector and increased for the large-scale one while the reverse is true for variant 2. In variant 3, the slope of the of the same function is decreased for the small–scale sector and increased for the large-scale sector. Variant 4 is the reverse of 3.

Table 5. Cost sensitivity analysis results. Values are US$ (10^6).

Variant		Small	Large
1	f-factor	0.18	0.32
	Net Value	18.55	86.57
2	f-factor	0.05	0.38
	Net Value	5.12	105.34
3	f-factor	0.19	0.32
	Net Value	18.7	88.6
4	f-factor	0.05	0.38
	Net Value	5.28	102.06

DISCUSSION

The results indicate that given our inputs and assumptions, there is a single set of effort levels on the part of the large and small-scale fleets that maximizes aggregate rents, i.e., provides a Pareto efficient solution. A single Nash bargaining solution can thus be identified as occurring when the benefits from cooperation are shared equally through side payments. In this case, the flow of payments is from the large-scale fleet to the small-scale fleet.

One of the most intriguing findings from this analysis results from a comparison of the levels of effort needed to produce maximum gross as opposed to net returns. In the former case, a very large increase in small-scale effort above current levels is called for and there is a sizable decline in net revenues generated by the fishery as a whole ($-64 million). In contrast, a sizable reduction in small-scale effort is required for maximum net revenues to be generated but the result is an increase in rents to society of over $100 million dollars. Each sector is also better off than currently. These results highlight the need for fisheries managers to attend to net returns to fishing and not simply gross returns.

An analysis of the sensitivity of these results to our assumptions about costs supports our overall conclusion that current levels of effort need to be substantially reduced, in both sectors. The relative levels of effort reduction do vary, however, depending on the specific assumptions. What is especially notable is that the optimal effort level of the small-scale sector is much more sensitive to changes in cost estimates than is the large-scale sector. Very modest changes in the slope and intercept of the cost function were reported here (5%). When changes of 10% were examined for variant 1, the recommended effort level of the small-scale sector jumped up to 0.26 while the optimal large-scale effort level declined to only 0.29.

Two notes of caution should be taken regarding this analysis. First, the different behavior of the two sectors is driven entirely by differences in gear selectivity. We do not include any other differences between the two sectors, e.g. discount rates, harvesting costs, selling price differences, etc. Munro (1979) has considered a number of these in the context of a bio-economic model based on a surplus-production (Schaefer) model. He finds that each of these factors can greatly influence the equilibrium outcome. Although we consider the optimal behavior of two sectors of a fishery rather than the decisions made by two countries it might be suspected that these variables do differ between sectors, in particular discount rates. We justify our lack of such a detailed economic analysis by noting that the aim of this particular model is simply to demonstrate the amount of rent that is lost from a non-optimal allocation of effort. We hope that others use these results as the basis for a more sophisticated economic analysis of the entry and exit decisions of small and large-scale fishers.

This brings us to a second note of caution. With respect to the fact that our results pertain to two sectors rather than two nations, we have presumed a willingness on the part of fishers to make side-payments. While this has been an effective solution in the case of transnational resources where the two players, countries, act effectively as individuals (see Munro 1979 for examples), it is not clear to what extent side-payments would be an acceptable solution to the many individuals who comprise the small and large-scale sectors. One cannot treat them as individual players as easily as one would two countries. Yet in this particular case both sectors actually benefit from cooperation without needing to bargain. However, the small-scale sector is composed of many more individual fishers and thus, per capita increases in returns may be insignificant if there is no bargaining and no transfer of revenues from the large to the small-scale sector.

Overall, we find that both gross and net returns can be increased if the two sectors of the fishery co-ordinate their levels of effort. However, different levels of effort are required to increase net as opposed to gross revenue and furthermore, the direction of change is opposite for the small-scale sector. When net returns are considered, a sizable reduction in total fishing effort can generate sizable increases in revenues

ACKNOWLEDGEMENTS

We would like to thank Anthony Charles, Trond Bjorndal, Tony Pitcher, Carl Walters, Reg Watson and Dirk Zeller for their assistance and constructive critiques of the work. In addition, we are grateful to the Pew Charitable Trusts for their generous support of his work.

REFERENCES

Baranov, F. I. (1918) [On the question of the biological basis of fisheries.] Nauchn. Issled. Ikhtiol. *Inst. Izv.* 1, 81–128.

Bertalanffy, L. von (1934) Untersuchungen über die Gesetzlichkeiten des Wachstums. 1. Allgemeine Grundlagen der Theorie. Roux' Arch. Entwicklungsmech. *Org.* 131, 613–53.

Beverton, R. J. H. and Holt, S. J. (1957) On the dynamics of exploited fish populations. Fish. Invest. Ser. II 19, 533 pp.

Beyer, J. (1987) On length-weight relationships. Part I: computing the mean weight of fish in a given length class. Fishbyte *(ICLARM)* 5,12-14.

Bigelow, H. B. and Schroeder, W. C. (1953) Fishes of the Gulf of Maine. Bulletin of the Fish and Wildlife Service 53. US Government printing office, Washington D.C.

Caddy, J. F. (1975) Spatial model for an exploited shellfish population, and its application to the Georges Bank scallop fishery. *Journal of the Fisheries Research Board of Canada* 32, 1305-1328.

Caddy, J. F. (1989) A perspective on the population dynamics and assessment of scallop fisheries, with special reference to the sea scallop, *Placopecten magellanicus* Gmelin. In: Caddy, J. F. (ed.) Marine Invertebrate Fisheries: Their Assessment and Management. John Wiley & Sons, New York, 559-589.

Campbell, A. (1986) Implications of size and sex regulations for the lobster fishery of the bay of Fundy and Southwestern Nova Scotia. *Canadian Special Publications in Fisheries and Aquatic Science* 92, 126-132.

Charles, A. (1989) Bio-socio-economic dynamics and multidisciplinary models in small-scale fisheries research. In: Proceedings of a symposium on Research and Small-scale Fisheries Montpellier, France 1989, 983-992.

Christensen, V. (1995). A multispecies virtual population analysis incorporating information of size and age. ICES C.M. 1995/D:8. 14 pp.

Estrella, B. T. and McKiernan, D. J. (1989) Catch-per-unit-effort and biological parameters from the Massachusetts coastal lobster (*Homarus americanus*) resource: description and trends. NOAA Technical Report NMFS 81, 1-21.

Fournier, D. A., Sibert, J. R. and Terceiro, M. (1991) Analysis of the length frequency samples with relative abundance data for the Gulf of Maine Northern Shrimp (Pandalus borealis) by the MULTIFAN method. *Canadian Journal of Fisheries and Aquatic Science* 48, 591-598.

Haynes, E. B. and Wigley, R. L. (1969) Biology of the northern shrimp, Pandalus borealis, in the Gulf of Maine. *Transactions of the American Fisheries Society* 98, 60-76.

Hoydal, K., Rørvik, C. J. and Sparre, P. (1982) Estimations of effective mesh size and their utilization in assessment. *Dana* 2, 69–95.

Gordon, H. S. (1954) Economic theory of a common property resource: The fishery. *Journal of Political Economy*, 62, 124-142.

Gulland, J. A. (1965) Estimation of mortality rates. Annex to Arctic fisheries working group report ICES C.M. 1965/D:3. (Reprinted as pp. 231–41 in Cushing, P. H. (ed), Key Papers on Fish Populations. IRL Press, Oxford, 1983.)

Jones, R. (1984) Assessing the effects of changes in exploitation pattern using length composition data (with notes on VPA and cohort analysis). FAO Fisheries Technical Paper 256, 118 pp.

Kimura, L. (1977) Statistical assessment of the age–length key. *Journal of the Fisheries. Research Board Canada* 34, 317–24.

Lange, A. M. T. and Johnson, K. L. (1981) Dorsal mantle length - total weight relationships of squids Loligo pealei and Illex illecebrosus from the Atlantic coast of the United States. NOAA Technical Report NMFS SSRF 745.

Longhurst, A. R. and Pauly, D. (1987) Ecology of Tropical Oceans. Academic Press Inc., San Diego CA.

Luce, R. D. and Raiffa, H. (1967) Games and Decisions. Wiley, New York.

Maurstad, A. (2000) To fish or not to fish: small-scale fishing and changing regulations of the cod fishery in Northern Norway. Human Organization 59, 37-47.

Mesnil, B. (1980) Théorie et pratique de l'analyse des cohortes. *Rev. Trav. Inst. Pêches Marit.* 44, 119–55.

Milazzo, M. (1998) Subsidies in world fisheries. World Bank Technical Paper 406, 86 pp.

Munro, G. R. (1979) The optimal management of transboundary renewable resources. *Canadian Journal of Economics* 12, 355-376.

Munro, J.L. (1980) Stock assessment models: applicability and utility in tropical small-scale fisheries. In: Roedel, P.M. and Saila, S.B. (eds.) Stock Assessment for Small-scale Fisheries. International Center for Marine Resources Development, University of Rhode Island, Kingston USA, 35-47.

Nash, J. F. (1953) Two-person cooperative games. *Econometrica* 21, 128-40.

NOAA (n.d.) Status of the Fishery Resources off the Northeastern United States. NOAA Technical Memorandum NMFS-NE-115.

Paloheimo, J. E. and Cadima, E. (1964) On statistics of mesh selection. ICNAF Ser. Doc. 1394/ 98. Pag. var.

Parsons, D. G. and Fréchette, J. (1989) Fisheries for Northern Shrimp (*Pandalus borealis*) in the Northwest Atlantic from Greenland to the Gulf of Maine. pp. 63-86. In: Caddy, J. F. (ed) Marine Invertebrate Fisheries: Their Assessment and Management. John Wiley & Sons, New York.

Pauly, D. (1980) On the interrelationships between natural mortality, growth parameters and mean environmental temperature in 175 fish stocks. *J. Cons., Cons. Int. Explor. Mer* 39(3), 175-192.

Pauly, D. (1984) Fish population dynamics in tropical waters: a manual for use with programmable calculators. ICLARM Studies Review 8, 325 pp.

Pauly, D. (1985) Population dynamics of short-lived species, with emphasis on squids. *NAFO Science Council Studies* 9, 143-154.

Pauly, D. and Pitcher, T. J. (2000) Assessment and mitigation of fisheries impacts on marine ecosystems: A multidisciplinary approach for basin-scale inferences, applied to the North Atlantic. In: Pauly, D. and Pitcher, T.J. (eds) Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. *Fisheries Centre Research Reports* 8(2), 1-12.

Pope, J. G. (1972) An investigation of the accuracy of virtual population analysis using cohort analysis. *ICNAF Research Bulletin* 9, 65–74.

Russell, M. P., Ebert, T. A. and Petraitis, P. S. (1998) Field estimates of growth and mortality of the green sea urchin, *Stongylocentrotus droebachiensis*. *Ophelia* 48, 137-153.

Shumway, S. E., Perkins, H. C., Schick, D. F. and Stickney, A. P. (1985) Synopsis of biological data on the pink shrimp, *Pandalus borealis* (Kroyer, 1838). NOAA Technical Report NMFS 30, 57 pp.

Sissenwine, M. P., Cohen, E. B. and Grosslein, M. D. (1984) Structure of the George's Bank ecosystem. Rapp. P.-v. Réun. Cons. *Int. Explor. Mer* 183, 243-254.

Steneck, R. S. (1997) Fisheries-induced biological changes to the structure and function of the Gulf of Maine Ecosystem. Plenary Paper. In: Wallace, G. T. and Braasch, E. F. (eds). Proceedings of the Gulf of Maine Ecosystem Dynamics Scientific Symposium and Workshop. RARGOM Report, 91 - 1. Regional Association for Research on the Gulf of Maine, Hanover, NH, 151-165.

Sumaila, U. R. (1997) Cooperative and non-cooperative exploitation of the Arcto-Norwegian cod stock in the Barents Sea. Environmental and Resource *Economics* 10, 147-165.

Swan, E. F. (1958) Growth and variation in sea urchins of York, Maine. *Journal of Marine Research* 17, 507-522.

Thompson, W. F. and Bell, F. H. (1934) Biological statistics of the Pacific halibut fishery. 2. Effect of changes in intensity upon total yield and yield per unit of gear. Rep. I. 8 49 pp.

Townsend, R. E. (1986) A critique of models of the American lobster fishery. *Journal of Environmental Economics and Management* 13, 277-291.

Tyedmers, P. (2000) Quantifying the energy consumed by North Atlantic fisheries. In: Pauly, D. and Pitcher, T.J. (eds) Methods for assessing the impact of fisheries on marine ecosystems of the North Atlantic. Fisheries Centre Research Reports 8(2), (in prep) 123-135.

Watson, R., Guénette, S., Fanning, P. and Pitcher, T. J. (2000) The basis for change 1: reconstructing fisheries catch and effort data. In: Pauly, D. and Pitcher, T.J. (eds) Methods for assessing the impact of fisheries on marine ecosystems of the North Atlantic. Fisheries Centre Research Reports 8(2), (in prep), 23-39.

Wilen, J. E. and Homans, F. R. (1997) Unraveling rent losses in modern fisheries: production, market, or regulatory inefficiencies? In: Global Trends: Fisheries Management. (Proceedings of the Symposium Global Trends: Fisheries Management, American Fisheries Society Symposium 20, Seattle Washington, 14-16 June 1994, Pikitch, E. K., Huppert, D. D. and Sissenwine, M. P. (eds) Duke, M. assoc. ed., American Fisheries Society, Bethesda Maryland, 256-263.

APPENDIX

Key Equations and Assumptions

Beverton and Holt (1957), following up on Baranov (1918) showed that the catch (C_i) from a population during a unit time period, i, is equal to the product of the population size at the beginning of the time period (N_i) times the fraction of the deaths caused by fishing, times the fraction of total deaths, which can be written

$$C_i / N_{i+1} = (F_i / Z_i) \cdot (e^{Z_i} - 1) \qquad ...5)$$

where F/Z expresses the fraction of the mortality caused by fishing. This is the equation for the virtual population analysis (VPA) of Gulland (1965).

Given values of C_i and an estimate of M, Equation (5) can be used to estimate (retroactively) the size of past cohorts (i.e. of groups of fish born at the same time and exposed to the same mortalities throughout their lives), given an estimate of N_{i+1}, from which to start the computation (Mesnil 1980). An approximation to (5) is given by

$$N_i = (N_{i+1} \cdot e^{-M/2} + C_i) \cdot e^{M/2} \qquad ...6)$$

wherein fishing mortality, for which Equation (5) cannot be solved directly, does not occur as an explicit parameter.

If we work backwards in time, estimating a new population size (N) at each step, fishing mortality estimates can be then obtained from the successive N values, using:

$$F_{i+1} = \ln(N_i / N_{i+1}) - M \qquad ...7)$$

When recruitment and new F for each length group (i) is given, the process can then be used to predict the catch. This predictive method is commonly called, 'Thompson and Bell' (1934) method.

While the same procedural flow is followed as with the age-structured Thompson and Bell model, some equations need to be altered to account for the conversion of length to age (or to relative age when t_o is not known). Converting length to age requires the use of a mathematical expression of fish growth, here the VBGF (von Bertalanffy growth function; Bertalanffy 1934):

$$L_t = L_\infty \left[1 - e^{-K(t - t_o)}\right] \qquad ... 8)$$

where

- L_∞ is the asymptotic length, that is the mean length the fish of a given stock would reach if they were to grow indefinitely;
- K is the rate (of dimension time^{-1}) at which L_∞ is approached; and
- t_o is the 'age of the fish at zero length' if they had always grown in the manner described by the equation (note that t_o is generally negative).

Thus, any age t_i pertaining to a length L_i can be obtained from

$$t_i = (1/K) \cdot \ln\left[1 - (L_i / L_\infty)\right] + t_o \qquad ...9)$$

and similarly for age t_{i+1}, pertaining to L_{i+1}. From the length–age relationships for L_i and L_{i+1}, Δt_i is obtained as the difference between t_{i+1} and t_i, or after some rearrangement

$$\Delta t_i = (1/K) \cdot \ln\left[(L_\infty - L_i)/(L_\infty - L_{i+1})\right] \qquad ... 10)$$

Recursively applying Equations (6) and (7), the catches can be computed for a change in the F-array (Jones 1984).

Given the parameters (a, b) of a length–weight relationship and the computed catches per length group (C_i), the corresponding yield (Y_i) then can be estimated (Beyer 1987) from

$$Y_i = \overline{w}_i \cdot C_i \qquad ...11)$$

where

$$\overline{w}_i = \left(\frac{1}{L_{i+1} - L_i}\right)\left(\frac{a}{b+1}\right) \cdot \left({L_{i+1}}^{b+1} - {L_i}^{b+1}\right) \qquad ...12)$$

Similarly, multiplying the yield estimates to a mean value (e.g. commodity price) will provide an overview of the expected change in the total value of the return.

Multiplying an F-array (see below) by a factor (the f-factor) simulate a change in effort (f). Thus, it is straightforward to estimate the amount of effort that should be added to or removed from a fleet.

The method presented can be used straightforwardly in multi-species situations if two crucial assumptions are met:

(i) The fishing pattern has no influence on recruitment;

(ii) Biological interactions among species can be neglected.

Assumption (i) implies here not only that over a wide range, recruitment is not affected by changes in the effort level — as is also assumed for single-species Y/R analyses — but also that the relative strength of recruitment between species remains unaffected by fishing. Thus, it is assumed that if three species A, B and C recruit to the fishing ground with relative strengths of 0.1, 0.6 and 0.3 respectively, species B will remain dominant even if its adults are targeted by the fishery.

This assumption is not likely to be met in reality — at least not strictly. However, radical changes of the relative species composition of a multi-species stock take a while to manifest themselves, even when they are induced by a fishery. Also, there are configurations that are more stable than others, with certain species remaining dominant over decades. Finally, it must be recalled that yield per recruit analyses usually lead to advice that, when implemented, may be conducive to *stabilizing* recruitment to the stock, especially when these analyses consider spawning biomass per recruit.

Assumption (ii), that species do not interact biologically means, in terms of the multi-species version of the approach presented; that the species-specific M values do not change as a function of fishing mortality. Thus, it is assumed among other things that the natural mortality of small fish remains constant irrespective of the biomass of large fish, i.e., of actual and/or potential predators.

This assumption is evidently not likely to be met in any real stock. Models exist (e.g. multi-species VPA) in which M is explicitly made to vary with predator biomass and size (age) structure (Christensen 1995). However, even without variable natural mortalities, the multi-species version of the Thompson and Bell model represents an improvement over the single-species approach. Further, there is always the possibility of running the model several times, with different values of M such as to be able to assess the effects of changes of M on yields.

Estimating an F-array from Selection Data

The model presented above requires estimates of mean size at first capture, i.e. ,the length at which 50 percent of the fish encountering a gear are retained if (L_{50}, or L_c). A common method to estimate L_c is to fit selection data with a logistic curve of the form

$$P_i = 1 \Big/ \left[1 + e^{-r(L_i - L_c)}\right] \qquad \text{...13)}$$

where P_i is the probability of capture at the midpoint of a length class i and r a constant whose value increases with the steepness of the selection curve; assuming the observed selection pattern to be symmetrical (or nearly so). Equation (13) may also be rewritten

$$P_i = 1 \Big/ \left[1 + e^{(S_1 - S_2 \cdot L_i)}\right] \qquad \text{...14)}$$

and L_i is the length interval midpoint, S_1 and S_2 being constant (Paloheimo and Cadima 1964, Kimura 1977 and Hoydal, Rørvik and Sparre 1982). Equation (25) can be re-expressed as

$$\ln[(1/P_i) - 1] = S_1 - S_2 \cdot L_i \qquad \text{...15)}$$

which can be identified with a regression line, where S_1 = a and S_2 = b (note that Equation 15 is not defined for P_i = 0 or P_i = 1).

There is a one-to-one correspondence between S_1 and S_2 and L_{25}, L_{50} and L_{75}, the lengths at which respectively 25, 50 and 75 percent of the fish are retained. The length range from L_{25} to L_{75}, which is symmetrical around L_{50}, is called the *selection range*.

The formulae for calculating L_{25}, L_{50} and L_{75} are

$$L_{25} = [S_1 - \ln(3)]/S_2 \qquad \text{...16)}$$

$$L_{50} = S_1/S_2 \qquad \text{...17)}$$

$$L_{75} = [\ln(3) + S_1]/S_2 \qquad \text{...18)}$$

S_1 and S_2 can be derived from L_{75} and L_{50} using:

$$S_1 = L_{50} \cdot \ln(3)/(L_{75} - L_{50}) \qquad \text{...19)}$$

$$S_2 = \ln(3)/(L_{75} - L_{50}) = S_1/L_{50} \qquad \text{...20)}$$

Similarly, de-selection (ability of the fish to escape or avoid the gear) can also be evaluated from

$D_{25} = [D_1 - \ln(3)]/D_2$ ...21)

$D_{50} = D_1/D_2$ ...22)

$D_{75} = [\ln(3) + D_1]/D_2$ ...23)

D_1 and D_2 can be derived from D_{75} and D_{50} using:

$D_1 = L_{50} \cdot \ln(3)/(D_{75} - D_{50})$ ...24)

$D_2 = \ln(3)/(D_{75} - D_{50}) = D_1/D_{50}$ ...25)

The cumulative effects of selection and de-selection effects can be computed as

$$P_i = 1 \Big/ \left[1 + e^{(S_1 - S_2 \cdot L_i)}\right] \cdot 1 \Big/ \left[1 + e^{(D_1 - D_2 \cdot L_i)}\right] \quad \text{...26)}$$

where P_i is the probability of capture for length group (i). When more than one gear (g) is used to exploit the given stock, the total probability of capture can be computed from

$$P_{i,g} = \sum \left(1 \Big/ \left[1 + e^{(S_{1,g} - S_{2,g} \cdot L_i)}\right] \cdot 1 \Big/ \left[1 + e^{(D_{,g1} - D_{2,g} \cdot L_i)}\right]\right)$$

...27)

The relative fishing mortality per length group (F-array) can be derived from Equation 27 by multiplying it by the catch (in number)

$$F'_i = \sum \left(C_i \Big/ \left[1 + e^{(S_{1,g} - S_{2,g} \cdot L_i)}\right] \cdot 1 \Big/ \left[1 + e^{(D_{,g1} - D_{2,g} \cdot L_i)}\right]\right)$$

...28)

The approximation of the F-array can then be computed by recursively applying Equation, (28) until the difference between the estimated total catch and the recorded total catch for a given gear is minimized.

Ecopath with Ecosim: Methods, Capabilities and Limitations

Villy Christensen and Carl Walters

University of British Columbia, Fisheries Centre, 2204 Main Mall, Vancouver, BC, Canada V6T 1Z4

Abstract

The *'Sea Around Us'* project uses ecosystem modeling based on the Ecopath with Ecosim (EwE) approach as an important component to characterize the status of North Atlantic ecosystems. For this, ecosystem models will be constructed covering the North Atlantic region, with a minimum of one model for each large marine ecosystem of the North Atlantic. For many of the areas the modeling will include construction of additional models representing the time period before industrialized fisheries had a major impact.

The EwE modeling approach combines software for ecosystem trophic mass balance (biomass and flow) analysis (Ecopath), with a dynamic modeling capability (Ecosim) for exploring past and future impacts of fishing and environmental disturbances. Ecosim models can be replicated over a spatial map grid (Ecospace) to allow exploration of policies such as marine protected areas, while accounting for spatial dispersal/advection effects.

The Ecopath approach and software has been under continuous development since 1990, with Ecosim emerging in 1995, and Ecospace in 1998, leading to an integrated package now called 'Ecopath with Ecosim'. We present an overview of the computational aspects of the Ecopath, Ecosim and Ecospace modules as they are implemented in the most recent software version. The paper summarizes the capabilities of the modelling system with respect to evaluating how fisheries and the environment impact ecosystems. We conclude by a warning about pitfalls in the use of the software for policy exploration.

Introduction

The *'Sea Around Us'* project relies extensively on ecosystem modeling to characterize the status of North Atlantic ecosystems (Pauly and Pitcher 2000). As part of this considerable effort is allocated to the construction of ecosystem models throughout the North Atlantic region, with a minimum of one ecosystem model being constructed for each large marine ecosystem of the North Atlantic (Pauly et al. 2000a). For many of the ecosystems the modeling will include construction of additional models representing the time period before major impact of industrialized fisheries, typically from around 1950, the start of the fishery catch database series of the *Sea Around Us Project* (Watson et al. 2000a).

The modeling approach that will be applied by the 'Sea Around Us' project relies heavily on the computer package known as 'Ecopath with Ecosim' (EwE, www.ecopath.org). EwE is coming to be widely used as a tool for analysis of exploited aquatic ecosystems, having reached 1750 registered users in 118 countries. EwE combines software for ecosystem trophic mass balance (biomass and flow) analysis (Ecopath) with a dynamic modeling capability (Ecosim) for exploring past and future impacts of fishing and environmental disturbances. It has a very elaborate user interface that eases a variety of data management chores and calculations that are a cumbersome but necessary part of any endeavor to systematically examine an ecosystem.

Recent versions of the software have integrated Ecosim with traditional single-species stock assessment, by allowing age-structured representation of particular, important populations and by allowing users to 'fit' the model to data. Ecosim models can be replicated over a spatial map grid (Ecospace) to allow exploration of policies such as marine protected areas, while accounting for spatial dispersal/advection effects.

The Ecopath approach and software have been under continuous development since the late 1980s (Christensen and Pauly 1992), with Ecosim emerging in 1995 (Walters et al. 1997, 2000), and Ecospace in 1998 (Walters et al. 1999), leading to an integrated package now called Ecopath with Ecosim. We give an overview of the computational aspects and capabilities of the Ecopath, Ecosim and Ecospace modules as they are implemented in the most recent software version (EwE Version 4 Beta), along with some reflections of potential pitfalls related to application of the software.

Mass-balance modeling: Ecopath

The core routine of Ecopath is derived from the Ecopath program of Polovina (1984), since modified to make superfluous its original assumption of steady state. Ecopath no longer assumes steady state but instead bases the parameterization on an assumption of mass balance over an arbitrary period, usually a year (but also see discussion below about seasonal modeling). In its present implementation Ecopath parameterizes models based on two master equations, one to describe the production term and one for the energy balance for each group.

Mortality for a prey is consumption for a predator

The first Ecopath equation describes how the production term for each group (i) can be split in components. This is implemented with the equation,

$$P_i = Y_i + M2_i + E_i + BA_i + M0_i \quad \text{...1)}$$

where P_i is the total production rate of (i), Y_i is the total fishery catch rate of (i), $M2_i$ is the total predation rate for group (i), E_i the net migration rate (emigration – immigration), BA_i is the biomass accumulation rate for (i), while $M0_i$ is the 'other mortality' rate for (i).

Of the terms in the equation above, the production rate, P_i, is calculated as the product of B_i, the biomass of (i) and P_i/B_i, the production/biomass ratio for group (i). The P_i/B_i rate under most conditions corresponds to the total mortality rate, Z, (see Allen 1971), commonly estimated as part of fishery stock assessments. The 'other mortality' is a catch-all term including all mortality not elsewhere included, e.g., mortality due to diseases or old age, and is internally computed from,

$$M0_i = P_i \cdot (1 - EE_i) \quad \text{...2)}$$

where EE_i is called the 'ecotrophic efficiency' of (i), and can be described as the proportion of the production that is utilized in the system (see Equation 6). The production term, M2, in Equation 1 serves to link predators and prey as,

$$M2_i = \sum_{j=1}^{n} Q_j \cdot DC_{ji} \quad \text{... 3)}$$

where the summation is over all (n) predator groups (j) feeding on group (i), Q_j is the total consumption rate for group (j), and DC_{ji} is the fraction of predator (j)'s diet contributed by prey (i). Q_j is calculated as the product of B_j, the biomass of group (j) and Q_j/B_j, the consumption/biomass ratio for group (j).

An important implication of the equation above is that information about predator consumption rates and diets concerning a given prey can be used to estimate the predation mortality term for the group, or, alternatively, that if the predation mortality for a given prey is known the equation can be used to estimate the consumption rates for one or more predators instead.

For parameterization Ecopath sets up a system with (at least in principle) as many linear equations as there are groups in a system, and it solves the set for <u>one</u> of the following parameters for each group:

- biomass;
- production/biomass ratio;
- consumption/biomass ratio; or
- ecotrophic efficiency

while the other three parameters along with following parameters must be entered for all groups:

- catch rate;
- net migration rate;
- biomass accumulation rate;
- assimilation rate; and
- diet compositions.

It was indicated above that Ecopath does not rely on solving a full set of linear equations, i.e., there may be fewer equations than there are groups in the system. This is due to a number of algorithms included in the parameterization routine that will try to estimate iteratively as many 'missing' parameters as possible before setting up the set of linear equations. The following loop is carried out until no additional parameters can be estimated,

1. The gross food conversion efficiency, g_i, is estimated using

$$g_i = (P_i/B_i) / (Q_i/B_i) \quad \text{...4)}$$

while P_i/B_i and Q_i/B_i are potentially solved by inverting the same equation;

2. The P/B ratio is then estimated (if possible) from

$$\frac{P_i}{B_i} = \frac{Y_i + E_i + BA_i + \sum_j Q_j \cdot DC_{ji}}{B_i \cdot EE_i} \quad \text{...5)}$$

This expression can be solved if both the catch, biomass and ecotrophic efficiency of group i, and the biomasses and consumption rates of all predators on group i are known (including group i if a zero order cycle, i.e., 'cannibalism' exists). The catch, net migration and biomass accumulation rates are required input, and hence always known;

3. The EE is estimated from

$$EE_i = \frac{Y_i + E_i + BA_i + M2_i \cdot B_i}{P_i}, \quad \text{...6)}$$

where the predation mortality M2 is estimated from Equation 3;

4. In cases where all input parameters have been estimated for all prey for a given predator group it is possible to estimate both the biomass and consumption/biomass ratio for such a predator. The details of this are described in the EwE Help System, Appendix 4, Algorithm 3 (available at www.Ecopath.org and distributed with EwE);

5. If for a group the total predation can be estimated it is possible to calculate the biomass for the group as described in detail in the EwE Help System, Appendix 4, Algorithm 4;

6. In cases where for a given predator j the P/B, B, and EE are known for all prey, and where all predation on these prey apart from that caused by predator j is known, the B or Q/B for the predator may be estimated directly.

7. In cases where for a given prey the P/B, B, EE are known and where the only unknown predation is due to one predator whose B or Q/B is unknown, it may be possible to estimate the B or Q/B of the prey in question.

After the loop no longer results in estimation of any 'missing' parameters a set of linear equations is set up including the groups for which parameters are still 'missing'. The set of linear equations is then solved using a generalized inverse method for matrix inversion described by Mackay (1981). It is usually possible to estimate P/B and EE values for groups without resorting to including such groups in the set of linear equations.

The loop above serves to minimize the computations associated with establishing mass-balance in Ecopath. The desired situation is, however, that the biomasses, production/biomass and consumption/biomass ratios are entered for all groups and that only the ecotrophic efficiency is estimated, given that no procedure exists for its field estimation.

The mass balance constraint implemented in the two master equations of Ecopath (Equation 1 and Equation 7) should not be seen as questionable assumptions but rather as filters for mutually incompatible estimates of flow. One gathers all possible information about the components of an ecosystem, of their exploitation and interaction and passes them through the 'mass balance filter' of Ecopath. The result is a possible picture of the energetic flows, the biomasses and their utilization. The more information used in the process and the more reliable the information, the more constrained the outcome will be.

The energy balance of a group

After the 'missing' parameters have been estimated so as to ensure mass balance between groups, energy balance is ensured within each group using the equation

$$\text{Consumption} = \text{production} + \text{respiration} + \text{unassimilated food} \quad \text{...7)}$$

This equation is in line with Winberg (1956) who defined consumption as the sum of somatic and gonadal growth, metabolic costs and waste products. The main differences are that Winberg focused on measuring growth, where we focus on estimating losses, and that the Ecopath formulation does not explicitly include gonadal growth. The Ecopath equation treats this as included in the predation term (where nearly all gonadal products end up in any case). This may be a shortcoming, but it is one that can be remedied fairly easily, and actually is in Ecosim (see section on page 88).

We have chosen to perform the energy balance so as to estimate respiration from the difference between consumption and the production and unassimilated food terms. This mainly reflects our focus on application for fisheries analysis, where respiration rarely is measured while the other terms are more readily available. To facilitate computations we have, however, included a routine ('alternative input') where the energy balance can be estimated using any given combination (including ratios) of the terms in the equation above.

Ecopath can work with energy as well as with nutrient related currencies (while Ecosim and Ecospace only work with energy related currencies). If a nutrient based currency is used in Ecopath the respiration term is excluded from the above equation, and the unassimilated food term is estimated as the difference between consumption and production.

Addressing uncertainty

Most, if not all, Ecopath models constructed so far have initially been based on a single set of input parameters representing the mean for the model period, typically for a given year. The model constructor typically modifies the input parameters so as to obtain mass balance, and the outcome is a possible representation of the trophic interactions in the system during a given year. 'Possible' in this context means that basic physiological and thermodynamic constraints are considered, but also that it is just one of many possible representations of the flows in the ecosystem.

The procedure described above has heuristic value as the model constructor may gain knowledge of how ecosystem resources interact and also of the implications of changes in input parameters (something made explicit in Ecopath through a formal sensitivity routine quantifying the impact on all estimated parameters of changes in any of the input parameters). The procedure does, however, ignore the inherent uncertainty of input parameters. To account for this a resampling routine, Ecoranger, has been designed to accept input probability distributions for the biomasses, consumption and production rates, ecotrophic efficiencies, catch rates, and diet compositions.

Using a Monte-Carlo approach, a set of random input variables is drawn from user-selected frequency distributions and the resulting model is evaluated based on user-defined criteria, and physiological and mass balance constraints. The results include probability distributions for the estimated parameters along with distributions of parameters in the accepted model realizations.

The Ecoranger routine can be viewed as providing probability distributions for transformation of the input variables. The derived probability distributions are likely to be narrower than the original distributions indicating that we have gained information in the process of checking for mass balance constraints, and eliminating parameter combinations that violate thermodynamic constraints. The information that is gained comes from evaluation of structural relationships as implemented in the Ecopath model, contrary to standard Bayesian approaches, which relies on data sampling. Combining such structural information from Ecopath with prior probabilities (the original probability distributions) corresponds to combining data with priors to derive the posterior distributions in the Bayesian sense. A procedure implementing such an approach using a 'sampling-importance-resampling' scheme (McAllister et al. 1994) is included in the Ecoranger module of EwE making it straightforward to derive what may be called 'Bayes marginal posterior distributions' (Walters 1996).

Categorizing data sources

The Ecoranger module has been available for several years but only a few examples of its use have been published, and so far none has fully exploited its Bayesian capabilities. A major reason for this is that it is a very data intensive task to describe the probability distributions for all input parameters (including the diet compositions matrices). To facilitate this task and to make the process more transparent we have implemented a 'pedigree' (Funtowicz and Ravetz 1990) routine that serves a dual purpose by describing data origin, and by assigning

Table 1. Options included in EwE for definition of 'pedigree' for consumer production/biomass and consumption/biomass ratios in ECOPATH. Similar option tables are implemented for biomasses, catches, and diets. For each group in an ecosystem one of these options is used to define the pedigree of the input parameter. The Index value is used for calculation of a pedigree index. The confidence intervals (C.I.) are used to describe parameter uncertainty in the balanced ecosystem model using the Ecoranger module. Index values and confidence intervals are defaults that can be changed by users.

Option	Index	C.I. (%)
Estimated by ECOPATH (other model)	0.0	±80
Guesstimate	0.1	±70
From other model	0.2	±60
Empirical relationship	0.5	±50
Similar group/species, similar system	0.6	±40
Similar group/species, same system	0.7	±30
Same group/species, similar system	0.8	±20
Same group/species, same system	1.0	±10

confidence intervals to data based on their origin (Pauly et al. 2000b).

The pedigree routine allows the user to mark the data origin using a pre-defined table for each type of input parameters. An example pertaining to both production/biomass and consumption/biomass ratios is given in Table 1. The Ecoranger module can subsequently pick up the confidence intervals from the pedigree tables and use these as prior probability distributions for all input data.

The pedigree index values in Table 1 are also used to calculate an overall pedigree index for a given model. The index values for input data scale from 0 for data that is not rooted in local data up to a value of 1 for data that are fully rooted in local data. Based on the individual index value an overall 'pedigree index' P is calculated based on

$$P = \sum_{i=1}^{n} \frac{I_{ij}}{n}$$

...8)

where I_{ij} is the pedigree index value for group i and parameter j for each of the n living groups in the ecosystem; j can represent either B, P/B, Q/B, Y or the diet. To scale based on the number of living groups in the system, an overall measure of fit, t^* is calculated as,

$$t^* = P \cdot \frac{\sqrt{(n-2)}}{\sqrt{1-P^2}}$$

...9)

This measure of fit is seen to describe how well rooted a given model is in local data. It addresses an often-aired concern regarding the degree to which 'models feed on models', i.e., whether models are based on data from other models, which again are based on data from other models, etc. We are presently in the process of describing the pedigree indices for all published Ecopath models where we have access to the model descriptions (in excess of 100 cases).

Particle size distributions

Based on growth and mortality information (see input data) the particle size distribution, (PSD, Sheldon et al. 1972) for a model can be calculated. A routine for this is included in EwE, where for each living group the following steps are conducted:

The time spent in each of a user-defined number of weight class is calculated starting at time 0, using

$$t = \ln(1-(\frac{W_t}{W_\infty})^{b^{-1}})/(-K) + t_0$$

...10)

where W_t is the lower limit of the weight interval, W_∞ is the asymptotic weight, b the exponent in the length-weight relationship, K the curvature parameter of the von Bertalanffy Growth Function (VBGF), and t_0 is the usually negative 'age' at which the weight is estimated to be zero in the VBGF. Once the time spent to reach each weight class limit is calculated, the time spent in each weight class is calculated by subtraction;

The survival is calculated as

$$N_t = N_{t-\Delta t} \cdot e^{-Z \cdot \Delta t} \quad \text{...11)}$$

where N_t is the number alive at time t, $N_{t-\Delta t}$ the number alive at the previous time step, Δt before, and Z is the total mortality rate, equivalent to the production/biomass ratio for the group;

The biomass contribution for the group to each weight class is calculated as

$$B_t = N_t \cdot W_t \cdot \Delta t \quad \text{...12)}$$

where B_t is the biomass contribution, Δt is the time the groups spends to grow through the given weight class (t), and the rest as explained above. B_t is scaled over all weight classes so as to sum up to the total biomass of the group;

The system PSD is calculated, finally, by summing up over all groups within each weight class. We anticipate that size distributions based on ecosystem models of past and present states of LME will be one of the pillars for describing ecosystem health (see below).

Ecosystem 'health'

The health status of a patient can often be captured with a single parameter, the temperature. Many have tried to find an index with similar ability to describe the health of an ecosystem to avoid the insurmountable task associated with bottom-up approach summing up the health of all ecosystem components, but a clear candidate has not appeared. The effort has led to development and description of a variety of system indicators, typically though with a given researcher exploring only one or a few of the potential indicators and on one or a few systems only.

We have sought to include a selection of ecosystem indicators in EwE using the criteria that the indicators can be estimated based on information included or potentially includable in EwE, typically based on quantified descriptions of food webs. In doing so we have facilitated straightforward calculation of the indices, leading to comparison of their properties through application to a variety of the models described using Ecopath.

One area of research where we have used this approach relates to ecosystem maturity, a potential descriptor of ecosystem health. Odum (1969, 1971) described how ecosystems develop over time in a non-deterministic way. We can assume an undisturbed ecosystem to be mature *sensu* Odum. Implications of this include that in a more mature system all niches should tend to be filled; that a larger part of the energy flows should be through detritus-based food webs; that primary production should be more efficiently utilized; that the total system biomass/energy throughput ratio should be higher; etc.

When ecosystems are disturbed, notably by fishing, we expect their maturity to decrease. This was indicated by the findings of Christensen (1995b), who used a series of indicators to rank a large number of ecosystem representations after maturity, and concluded that the ranking obtained was in agreement with the expect state of maturity. The study included several ecosystems for which the maturity state could be compared before and after a disturbance, and the findings were in all cases in agreement with disturbances leading to a reduction in maturity. Christensen and Pauly (1998) tried to model the present and the unfished state for two marine ecosystems, and for both systems concluded that the indices of ecosystem maturity for the fished and unfished states in all cases were in agreements with Odum's theory.

While these studies are inconclusive, they do indicate that it is feasible to use a composite of ecosystem indices to describe the state of a given system and how it may have changed over time. We intend to explore this further, and to include a number of additional measures of ecosystem health in EwE.

The selection of ecosystem indicators referred to above is included in EwE as part of a series of network analyses. In overview form (see references below and the EwE Help system for more detailed descriptions) the following routines are among those included,

- Cycling index: fraction of an ecosystem's throughput that is recycled, Finn (1976);
- Predatory cycling index: corresponds to the cycling index but computed with cycles involving detritus groups excluded;
- Cycles and pathways: based on an approach suggested by Ulanowicz (1986) a routine has been implemented to describe the numerous cycles and pathways that are implied by the food web representing an ecosystem;
- Connectance index: defined for a given food web as the ratio of the number of actual links to the number of possible links. Feeding on detritus (by detritivores) is included in the count, but the opposite links (i.e., detritus 'feeding' on other groups) are disregarded.
- System omnivory index: defined as the average omnivory index of all consumers weighted by the logarithm of each consumer's food intake. The logarithms are used as weighting factors because it can be expected that the intake rates are approximately log normally distributed. The system omnivory index is a measure of how the feeding interactions are distributed between trophic levels. An omnivory index is also calculated for each consumer group as a measure of the variance of the trophic level estimate for the group.
- Trophic level decomposition: aggregates the system into discrete trophic levels *sensu* Lindeman based on an approach suggested by Ulanowicz (1995). The routine reverses the routine for calculation of fractional trophic levels;
- Trophic transfer efficiencies: calculated for a given trophic level as the ratio between the sum of the exports plus the flow that is transferred from one trophic level to the next, and the throughput on the trophic level. The transfer efficiencies are used for construction of trophic pyramids;
- Primary production required (PPR): to estimate the primary production required (Christensen and Pauly, 1993) to sustain the catches and the consumption by the trophic groups in an ecosystem the following procedure has been implemented: first, all cycles are removed from the diet compositions, and all paths in the flow network are identified using the method suggested by Ulanowicz (1995). For each path the flows are then raised to primary production equivalents using the product of the catch, the consumption/production ratio of each path element times the proportion the next element of the path contributes to the diet of the given path element.
- Mixed trophic impact (MTI): Leontief (1951) developed a method for input-output analysis to assess the direct and indirect interactions in the economy of the USA, using what has since been called the Leontief matrix. A modified input-output analysis based on the procedure described by Ulanowicz and Puccia (1990) is implemented in EwE. The MTI describes how any group (including fishing fleets) impacts all other groups in an ecosystem trophically. It includes both direct and indirect impact, i.e. both predatory and competitive interactions.

  The MTI for living groups is calculated by constructing an n x n matrix, where the i,j^{th} element representing the interaction between the impacting group i and the impacted group j is

  $$MTI_{i,j} = DC_{i,j} - FC_{j,i}\ , \qquad \text{...13)}$$

  where $DC_{i,j}$ is the diet composition term expressing how much j contributes to the diet of i, and $FC_{j,i}$ is a host composition term giving the proportion of the predation on j that is due to i as a predator. When calculating the host compositions the fishing fleets are included as 'predators'.

  For detritus groups the $DC_{i,j}$ terms in Equation 13 above are set to 0 . For each fishing fleet a 'diet composition' is calculated representing how much each group contributes to the catches, while the host composition term as mentioned above includes both predation and catches.

The diagonal elements of the MTI are further increased by 1, i.e.

$$MTI_{i,i} = 1 + MTI_{i,i} \quad \text{...14)}$$

The matrix is inverted using a standard matrix inversion routine.

- Ascendency: EwE includes a number of indices related to the ascendency measure described in detail by Ulanowicz (1986). Ascendency is seen as a measure of ecosystem growth and development. The method for calculation of ascendency has since been changed by Ulanowicz (pers. comm.), and we have not yet incorporated the new version of ascendency.

Time-dynamic Simulation: Ecosim

The basics of Ecosim consist of biomass dynamics expressed through a series of coupled differential equations. The equations are derived from the Ecopath master equation (Equation 1), and take the form

$$dB_i/dt = g_i \sum_j Q_{ji} - \sum_j Q_{ij} + I_i - (M_i + F_i + e_i)B_i \quad \text{...15)}$$

where dB_i/dt represents the growth rate during the time interval dt of group (i) in terms of its biomass, B_i, g_i is the net growth efficiency (Equation 4), M_i the non-predation ('other') natural mortality rate, F_i is fishing mortality rate, e_i is emigration rate, I_i is immigration rate, (and $e_i \cdot B_i - I_i$ is the net migration rate of Equation 1). The two summations estimate consumption rates, the first expressing the total consumption by group (i), and the second the predation by all predators on the same group (i). The consumption rates, Q_{ji}, are calculated based on the 'foraging arena' concept, where B_i's are divided into vulnerable and invulnerable components (Walters et al. 1997, Figure 1), and it is the transfer rate (v_{ij}) between these two components that determines if control is top-down (i.e., Lotka-Volterra), bottom-up (i.e., donor-driven), or of an intermediate type.

The set of differential equations is solved in Ecosim using (by default) an Adams-Basforth integration routine or (if selected) a Runge-Kutta 4th order routine.

Predicting consumption

Ecosim bases the crucial assumption for prediction of consumption rates on a simple Lotka-Volterra or 'mass action' assumption, modified to consider 'foraging arena' properties. Following this, prey can be states that are or are not vulnerable to predation, for instance by hiding, (e.g., in crevices of coral reefs or inside a school) when not feeding, and only being subject to predation when having left their shelter to feed (Figure 1). In the original Ecosim formulations (Walters et al. 1997, 2000) the

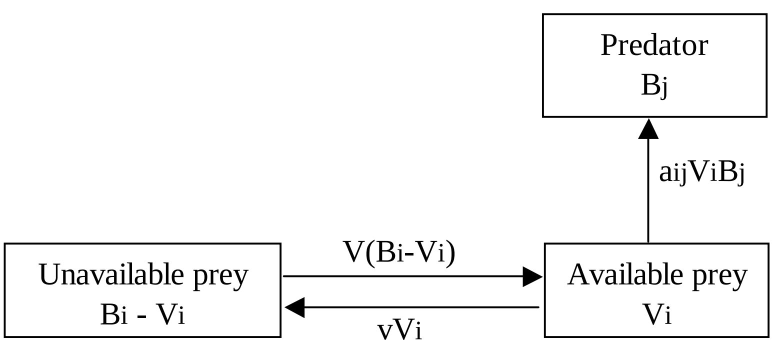


Figure 1. Simulation of flow between available (V_i) and unavailable (B_i - V_i) prey biomass in Ecosim. a_{ij} is the predator search rate for prey i, v is the exchange rate between the vulnerable and not-vulnerable state. Fast equilibrium between the two prey states implies $V_i = vB_i / (2v + aB_j)$. Based on Walters et al. (1997).

consumption rate for a given predator i feeding on a prey j was predicted from,

$$Q_{ij} = \frac{a_{ij} \cdot v_{ij} \cdot B_i \cdot P_j}{2v_{ij} + a_{ij} \cdot P_j}$$

...16)

where, a_{ij} is the effective search rate for predator j feeding on a prey i, v_{ij} base vulnerability expressing the rate with which prey move between being vulnerable and not vulnerable, B_i prey biomass, P_j predator abundance (N_j for split pool groups discussed later, and B_j for other groups).

The model as implemented argues that 'top-down vs. bottom-up' control is in fact a continuum, where low v's implies bottom-up and high v's top-down control. (Note that the input vulnerability rates in EwE are scaled to range from 0 to 1, with 0.3 serving as default for mixed control, and 0 implying bottom-up, 1 top-down control. The actual v's used in the computations are rescaled).

Early experience with Ecosim has led to a more elaborate expression to describe the consumption:

$$Q_{ij} = \frac{a_{ij} \cdot v_{ij} \cdot B_i \cdot P_j \cdot T_i \cdot T_j \cdot S_{ij} \cdot M_{ij} / D_j}{v_{ij} + v_{ij} \cdot T_i \cdot M_{ij} + a_{ij} \cdot M_{ij} \cdot P_j \cdot S_{ij} \cdot T_j / D_j}$$

...17)

where, T_i represents prey relative feeding time, T_j predator relative feeding time, S_{ij} user-defined seasonal or long term forcing effects, M_{ij} mediation forcing effects, and D_j effects of handling time as a limit to consumption rate,

$$D_j = \frac{h_j \cdot T_j}{1 + \sum_k a_{kj} \cdot B_k \cdot T_k \cdot M_{kj}}$$

...18)

where h_j is the predator handling time. The feeding time factors are discussed further below, (see Equation 29). A vulnerability setting of 0 will result in consumption being estimated using bottom-up conditions only through

$$Q_{ij} = a_{ij} \cdot B_i \cdot T_j \cdot S_{ij}$$

...19)

Life history handling

To better represent ontogenetic shifts in Ecosim, groups can be split into juvenile and adult components, and Ecosim then applies a Deriso-Schnute delay-difference model (Deriso 1980, Schnute 1987) to keep track of the number that recruits from juvenile to adult stages, and the number at age/size in the adult groups (Walters et al 2000).

Denoting the pool index for adults as A, and for juveniles as J, the basic model structure is,

$$\mathrm{B}_{\mathrm{A,t+1}} = e^{-\mathrm{Z}_{\mathrm{A,t}}}[\alpha_{\mathrm{A,t}}(\mathrm{Q}_{\mathrm{A,t}})\mathrm{N}_{\mathrm{A,t}} + \rho_A \mathrm{B}_{\mathrm{A,t}}] + \mathrm{I}_{\mathrm{A}} + \mathrm{w}_{\mathrm{J,k,t}}\mathrm{N}_{\mathrm{J,k,t}}$$

... 20)

$$\mathrm{N}_{\mathrm{A,t+1}} = \mathrm{N}_{\mathrm{A,t}} e^{-\mathrm{Z}_{\mathrm{A,t}}} + \mathrm{N}_{\mathrm{J,k,t}}$$

...21)

$$\mathrm{N}_{\mathrm{J,1,t+1}} = \mathrm{R}(\mathrm{B}_{\mathrm{A,t}}, \mathrm{N}_{\mathrm{A,t}}, \mathrm{Q}_{\mathrm{A,t}})$$

...22)

$$\mathrm{N}_{\mathrm{j,a,t+1}} = e^{-\mathrm{Z}_{\mathrm{J,t}}}\,\mathrm{N}_{\mathrm{j,a-1,t}} \quad a=1,\ldots,k$$

...23)

$$\mathrm{w}_{\mathrm{J,a,t+1}} = \mathrm{w}_{\mathrm{J,a-1,t}} + \mathrm{g}_j{}'\mathrm{Q}_{\mathrm{J,t}}/\mathrm{N}_{\mathrm{J,t}}$$

...24)

where t = time, (in months to increase flexibility to represent seasonality, short lives);

$Z_{A,t}$ = adult pool total mortality rate $M_{A,o}+e_{A,o}+F_{A,t}+\Sigma C_{Aj}/B_{A,t}$;

$Z_{J,t}$ = juvenile pool total mortality rate $M_{J,o}+e_{J,o}+F_{J,t}+\Sigma Q_{Jj}/B_{J,t}$;

$N_{J,a,t}$ = Number of age (months) a juveniles at time t;

$R(B_{A,t},N_{A,t},Q_{A,t})$ = recruitment function predicting the number of age (month) 0 juveniles produced in month t, from adult numbers, biomass, and food consumption $Q_{A,t}$;

k = age (months) at recruitment to the adult pool;

$w_{J,a,t}$ = body weight of an age a juvenile at time t;

$Q_{J,t}$ = total food consumption by juveniles J in month t;

$N_{J,t}$ = total number of juveniles at start of month t (summed over ages a);

g_j' = juvenile growth efficiency;

$\alpha_{A,t}(Q_{A,t})$ = Ford-Brody growth model intercept, assumed to depend on adult food consumption $Q_{A,t}$ in month t;

ρ = Ford-Brody growth model slope, representing metabolism.

The delay-difference representation of population age and size structure permits explicit representation of changes in growth, mortality, and recruitment processes with changing feeding conditions. It also makes it straightforward to include (1) changes in how food intake is allocated between growth and reproduction as food conditions varies; (2) changes in vulnerability to predation associated with changes in feeding behavior as prey densities vary; and (3) recruitment constraints related to juvenile size and fecundity. These aspects will be described further in the next sections.

Food allocation between growth and reproduction

The net (of assimilation and SDA losses) food intake $g'C_{A,t}$ (see Equation 15) by an adult pool can be distributed between food used for growth,

$$F_g = P_g \cdot g_A{}' \cdot Q_{A,t} \qquad \text{...25)}$$

where P_g is the proportion of net intake allocated to growth, g_A' is the growth efficiency for adults, and food used for reproduction,

$$F_r = g_A^{'} \cdot Q_{A,t} - F_g \qquad \text{...26)}$$

Assume that $Q_{A,t}$ is allowed to vary with feeding opportunities as predicted from Equation 17. In order for the growth curve α to remain constant at some value α_o, where $\alpha = F_g / N_{A,t}$ (the per capita allocation of food to growth), the allocation proportion P_g must vary as,

$$P_g^{constant\ growth} = \frac{\alpha_0}{g_A^{'} \cdot Q_{A,t} / N_{A,t}} \qquad \text{...27)}$$

(subject to $P_g^{constant\ growth} \leq 1.0$). The opposite extreme from this assumption is that a constant proportion P_o of the net food intake is used for growth, which implies that both α and fecundity will be proportional to food intake.

In Ecosim the user can move between these two extreme hypotheses ($\alpha = \alpha_o$ versus α proportional to Q/N) by specifying a life history 'weighting factor' W_g:

$$P_g^{realized} = W_g \cdot P_g^{constant\ growth} + \left(1 - W_g\right) \cdot P_0 \qquad \text{...28}$$

That is, at every model time step Ecosim calculates the net food intake per animal $g' Q_{A,t} / N_{A,t}$, and uses this to calculate $P_g^{constant\ growth}$. The $P_g^{realized}$ for that time step is then given by the equation above. W_g can be varied freely to generate a range of physiological allocation scenarios.

Foraging time and predation risk

The food consumption prediction relationship in Equation 17 contains two parameters that directly influence the time spent feeding and the predation risk that feeding may entail: a_{ij} and v'_{ij}. To model possible linked changes in these parameters with changes in food availability as measured by per biomass food intake rate $q_{it} = Q_{it} / B_{it}$ (i=juvenile index J or adult index A), we need to specify how changes in q_{it} will influence at least relative time spent foraging.

Denoting the relative time spent foraging as T_{it}, measured such that the rate of effective search during any model time step can be predicted as $a_{jit} = T_{it}\, a_{ji}$ for each prey type j that i eats. Further, we assume that time spent vulnerable to predation, as measured by v'_{ij} for all predators j on i, is inversely related to T_{it}, i.e., $v'_{ijt} = v'_{ij} / T_{it}$. An alternative structure that gives similar results is to leave the a_{ij} constant, while varying the v_{ij} by setting $v_{ijt} = T_{jt} \cdot v_{ij}$ in the numerator of Equation 17, and $v_{ijt} = T_{it} \cdot v_{ij}$ in the denominator.

For convenience in estimating the a_{ij} and v'_{ij} parameters, we scale T_{it} so that $T_{io}=1$, and $v'_{ij}=v_{ij}$. Using these scaling conventions, the key issue then becomes how to functionally relate T_{it} to food intake rate q_{it} so as to represent the hypothesis that animals with lots of food available will simply spend less time foraging, rather than increase food intake rates.

In Ecosim a simple functional form for T_{it} is implemented that will result in near constant feeding rates, but changing time at risk to predation, in situations where rate of effective search a_{ji} is the main factor limiting food consumption rather than prey behavior as measured by v_{ji}. This is implemented in form of the relationship:

$$T_{i,t} = T_{i,t-1} \cdot \left(1 - a + \frac{a \cdot q_{i,opt}}{q_{i,t-1}}\right) \qquad \text{...29}$$

where, a is a user-defined feeding time adjustment factor [0, 1], $q_{i,opt}$ is the (internally computed) feeding rate that optimizes feeding rate versus mortality risk for (i), $q_{i,t-1}$ is the feeding rate in the previous time step for the group. The time spent feeding is constrained by a user-defined value (default of two times the feeding rate in the Ecopath base model).

Grow fast, die young?

Ecosim predictions, especially for recruitment dynamics in split-pool cases, are generally quite sensitive to assumptions about how organisms adjust time spent foraging in relation to changes in feeding rates. Foraging time adjustments have two opposing (tradeoff) effects: more time means higher food intake (or less reduction in intake rate during a period of decline in prey abundance), but also possibly higher predation loss rate. There is much interest in evolutionary ecology in how organisms balance these effects, i.e., in how natural selection has 'optimized' foraging time.

One type of check of whether the behavioral parameters/assumptions in Ecosim are reasonable is to see if a model would 'evolve' toward very different foraging time patterns if the effects of the tradeoff are explicitly recognized. A routine for checking this is included in Ecosim, applying this routine the normal functions linking foraging time to Q/B are disabled. Instead, Ecosim calls a routine that evaluates the derivative of a fitness measure f with respect to foraging time for each pool, and at each simulated time step Ecosim then adjusts foraging times in a direction that will increase this fitness measure. That is, Ecosim tries to 'evolve' foraging times toward values that will maximize fitness. The fitness measure used is simply the per capita biomass derivative, f = dB/Bdt, i.e., the biomass derivative divided by biomass.

Stock-recruitment considerations

For split biomass pools, Ecosim estimates a baseline recruitment rate R_o to each adult pool by assuming equilibrium in Equation 22 as noted above. Recruitment takes place at age k months, and we can calculate baseline recruitment R^*_o at age 0 months from $R^*_o = R_o \exp(Z_{J,o}\, k)$. A basic problem is then to predict how R* will vary over time with changes in adult abundance, feeding rate, and/or body size (i.e., to define a reasonable recruitment function $R(B_{A,t}, N_{A,t}, Q_{A,t})$). We assume that R* is limited by recent feeding rate rather than accumulated food intake as reflected in body size, and use the following function to represent this limitation

$$R(B_{A,t}, N_{A,t}, Q_{A,t}) = R_0^* \cdot \left(\frac{N_{A,t}}{N_{A,0}}\right) \cdot \left[\frac{\left(1 - P_g^{realized}\right) \cdot Q_{A,t}}{\left(1 - P_0\right) \cdot Q_{A,0}}\right]^r \qquad \text{...30}$$

This relationship scales R relative to R^*_o by the ratio of adult abundance at time t to initial ($N_{A,t}/N_{A,o}$), and by the ratio of per biomass food consumption allocated to reproduction ($(1-P_g^{realized})Q_{A,t}$ to the baseline food allocated. The power parameter r can be used to generate nonlinear effects of the food consumption rate on recruitment per adult individual.

If there is immigration of juveniles, this is added to the equation above. Further, egg production is allowed to vary seasonally or over long-term through a user-defined forcing function. If an egg production curve is defined, the egg production term is multiplied on the equation above for the first age group of juveniles.

Primary production

For primary producers, the production is estimated as a function of the producers' biomass, B_i, from a simple saturating relationship

$$f(B_i) = \frac{r_i \cdot B_i}{1 + B_i \cdot h_i} \qquad \text{...31)}$$

where, r_i is the maximum production/biomass ratio that can be realized (for low B_i's), and r_i/h_i is the maximum net primary production when the biomass is not limiting to production (high B_i's). For parameterization it is only necessary to provide an estimate of r_i / (P_i/B_i), i.e., a factor expressing how much primary production can be increased compared to the base model state.

Fitting Ecosim to time series data

Based on time series 'reference' data, relative and absolute biomasses, and on total mortality of various pools over a particular historical period, along with estimates of changes in fishing impacts over that period, Ecosim estimates a statistical measure of goodness of fit to these data each time Ecosim is run. This goodness of fit measure is a weighted sum of squared deviations (SS) of log biomasses from log predicted biomasses, scaled in the case of relative abundance data (y) by the maximum likelihood estimate of the relative abundance scaling factor (q) in the equation $y = q \cdot B$, where B is the absolute abundance. The reference data series can be assigned a relative weight expressing how variable or reliable that type of data is compared to the other reference time series. Based on the time series, three types of analysis with the SS measure are available:

1. determine sensitivity of SS to the critical Ecosim vulnerability parameters by changing each one slightly then re-running the model to see how much SS is changed;
2. search for vulnerability estimates that give better 'fits' of Ecosim to the time series data;
3. search for time series values of forcing functions, e.g., annual relative primary productivity that may represent historical productivity 'regime shifts' impacting biomasses throughout the ecosystem.

The searches include a SS minimization procedure based on a Marquardt nonlinear search algorithm with trust region modification of the Marquardt steps. For users familiar with the nonlinear estimation procedures used in single-species stock assessment, e.g., for fitting production models to time series CPUE data, the procedure implemented in Ecosim should be quite familiar. In essence, the Ecosim search procedure for vulnerabilities is an 'observation error' fitting procedure where vulnerability changes usually have effects quite similar to changes in population 'r' parameters in single-species models. Allowing the search to also include historical primary production 'anomalies' corresponds to searching also for 'nuisance parameter' estimates of what is usually called the 'process errors' in single-species assessment.

Compensatory mechanisms

Sustaining fisheries yield when fishing reduces stock size depends on the existence of compensatory improvements in per capita recruitment, growth, and/or natural mortality rates. Ecosim allows users to represent a variety of specific hypotheses about compensatory mechanisms. Broadly, these mechanisms fall in two categories:

1. *direct* - changes caused over short time scales (order one year) by changes in behavior of organisms, whether or not there is an ecosystem-scale change due to fishing; and
2. *indirect* - changes over longer time scales due to ecosystem-scale responses such as increased prey densities and/or

reduced predator densities. Usually we find the direct effects to be most important in explaining historical response data. Here we describe how to generate alternative models or hypotheses about direct compensatory responses; these hypotheses fall in three obvious categories: recruitment, growth, and natural mortality.

Compensatory recruitment (models with split pools only)

Compensatory recruitment effects are usually expressed as a flat or dome-shaped relationship between numbers of juveniles recruiting to the adult pool versus parental abundance (stock recruit relation). There are two main ways to create such effects in Ecosim:

a. non-zero feeding time adjustment for the juvenile pool combined with fixed time in juvenile stage and high EE, or high proportion of the 'other' mortality (the mortality not accounted for) being sensitive to changes in predator feeding time; and

b. zero feeding time adjustment combined with variable time in juvenile stage.

Mechanism (a) represents density-dependent changes in juvenile mortality rate associated with changes in feeding time and predation risk, while (b) represents density-dependent changes in juvenile growth rate and hence total time spent exposed to high predation rates over the juvenile life stage. Other, generally weaker compensatory responses can also be caused by changes in adult energy allocation to reproduction. For mechanisms (a) and (b), it is usually also important that the vulnerabilities of prey to the juvenile group (Flow control tab) also be relatively low.

Compensatory growth

Compensatory growth rate responses are modeled by setting the feeding time adjustment rate to zero, so that simulated Q/B is allowed to vary with pool biomass (nonzero feeding time adjustment results in simulated organisms trying to maintain Ecopath base Q/B by varying relative feeding time). Net production is assumed proportional (growth efficiency) to Q/B, whether or not this production is due to recruitment (split pools) or growth. The Q/B increase with decreasing pool biomass is increased by decreasing vulnerability of prey to the pool. In the extreme as vulnerability approaches zero (donor or bottom up control), total food consumption rate Q approaches a constant (Ecopath base consumption), so Q/B becomes inversely proportional to B.

Compensatory natural mortality

Compensatory changes in natural mortality rate (M) can be simulated by combining two effects: nonzero feeding time adjustment, and either high EE from Ecopath or high proportion of 'other' mortality being sensitive to changes in predator feeding time. With these settings, especially when vulnerabilities of prey to a group are low, decreases in biomass lead to reduced feeding time, which leads to proportional reduction in natural mortality rate.

Compensation in recruitment

The 'split pool' representation of juvenile and adult biomasses was originally included in Ecosim to allow representation of trophic ontogeny (differential diets for juveniles and adults). To implement this representation it was necessary to include population numbers and age structure, at least for juveniles, so as to prevent 'impossible' dynamics such as elimination of juvenile biomass by competition/predation or fishing without attendant impact on adult abundance (graduation from juvenile to adult pools cannot be well represented just as a biomass 'flow').

When we elected to include age structure dynamics, we in effect created a requirement for model users to think carefully about the dynamics of compensatory processes that have traditionally been studied in terms of the 'stock-recruitment' concept and relationships. To credibly describe the dynamics of split-pool populations, Ecosim parameters for split pools usually need to be set so as to produce an 'emergent' stock-recruitment relationship that is at least qualitatively similar to the many, many relationships for which we now have empirical data (see data summary in www.mscs.dal.ca/~myers/data.html). In most cases, these relationships are 'flat' over a wide range of spawning stock size, implying there must generally be strong compensatory increase in juvenile survival rate as spawning stock declines (otherwise less eggs would mean less recruits on average, no matter how variable the survival rate might be).

When creating split pool dynamics care must be exerted in setting model parameters that define or create compensatory effects. This begins with the Ecopath input parameters; in order for the juvenile dynamics to exhibit compensatory mortality changes, at least two conditions are needed or helpful:

1. the juvenile group must have relatively high total mortality rate;
2. the juvenile group must have most mortality accounted for as predation or fishery effects within the model, or the user must specify that the 'other' mortality (the mortality not accounted for) is very sensitive to changes in predator feeding time.

Given these Ecopath conditions, Ecosim can then generate direct (as opposed to just predator-prey) compensatory changes in juvenile recruitment via at least three alternative mechanisms or hypotheses:

1. simple density-dependence in juvenile production rate by adults, due to changes in adult feeding rates and fecundity (not a likely mechanism);
2. changes in duration of the juvenile stage and hence in total time exposed to relatively high predation risk;
3. changes in juvenile foraging time (and hence exposure to predation risk) with changes in juvenile feeding rates.

For all of these mechanisms, compensatory effects are increased (recruitment relationship flat over a wider range of adult stock size, steeper slope of recruitment curve near the origin) by

1. limiting availability of prey to juveniles by forcing juveniles to use small 'foraging arenas' for feeding;
2. make effective time exposed to predation while feeding drop directly with decreasing juvenile abundance (simulates possibility that when juveniles are less abundant, remaining ones may be able to forage 'safely' only in refuge sites without exposing themselves to predation risk). This option should be used only if field natural history observation indicates that the juveniles do in fact restrict their distribution to safe habitats when at very low abundance.

Parameter sensitivity

Ecosim does not include any formal sensitivity analysis. Experience shows, however, that of the extra parameters added to those required by the typical Ecopath models, the most sensitive parameter is the vulnerability setting. This parameter expresses the exchange rate between the prey being in vulnerable and non-vulnerable states (Figure 1).

The vulnerability parameter is in general not subject to direct measurement. There are however other ways of estimating it, and Ecosim includes three independent methods of estimation:

1. build Ecopath models for a system covering two different time periods, and use a routine included in Ecosim to search for vulnerability parameter settings that with the given exploitation rates will make it possible to move from the first to the second model state;
2. using the evolutionary optimization model discussed above, seeks the evolutionary optimum between spending more time feeding and growing faster but at higher mortality risk;
3. through fitting to time series data (see above).

It is possible and indeed recommended to use all of these methods to obtain estimates for the vulnerability parameters.

Spatial simulation: Ecospace

Ecospace is a dynamic, spatial version of Ecopath, incorporating all key elements of Ecosim (Walters et al. 1999). It works by dynamically allocating biomass across a user-defined grid map while accounting for:

1. symmetrical movements from a cell to its four adjacent cells modified by whether a cell is defined as 'preferred habitat' or not;
2. user defined increased predation risk and reduced feeding rate in non-preferred habitat;
3. a level of fishing effort that is proportional, in each cell, to the overall profitability of fishing in that cell, and whose distribution is sensitive to spatial fishing costs.

Prediction of mixing rates

The instantaneous emigration rates from a given cell in Ecospace are assumed to vary based on the pool type, the groups preference for the habitat type represented by the cell, and a 'risk ratio' representing how the organisms in the cell respond to predation risk. Base dispersal rates are calculated based on this, but weighted based on a habitat gradient function increasing the probability of organisms moving towards favorable habitats. The mechanisms involved in this procedure are explained in more detail by Walters et al. (1999).

Predicting spatial fishing patterns

EwE works with multiple fishing fleets, with fishing mortality rates (F) initially distributed between fleets based on the distribution in the underlying Ecopath base model. In Ecospace the F's are distributed using a simple 'gravity model' where the proportion of the total effort allocated to each cell is assumed proportional to the sum over groups of the product of the biomass, the catchability, and the profitability of fishing the target groups (Caddy 1975, Hilborn and Walters 1987). This profitability of fishing includes factors such as the cell-specific cost of fishing.

Assuming that there are N cells representing water areas, each fleet k can cause a total fishing mortality rate $N \cdot F_k$. For each step in the simulation this rate is distributed among cells, c, in proportion to the weights G_{kc} based on:

$$G_{kc} = O_{kc} \cdot U_{kc} \cdot \frac{\sum_i p_{ki} \cdot q_{ki} \cdot B_{ic}}{C_{kc}}$$

...32)

where C_{kc} is 1 if cell c is open to fishing by fleet k, and 0 if not; U_{kc} is 1 if the user has allowed fleet k to work in the habitat type to which cell c belongs, and 0 if not; p_{ki} is the relative price fleet k receives for group i fish, q_{ki} is the catchability of group i by fleet k (equal to the F_{ki} in the Ecopath model); B_{ic} is the biomass of group i in cell c; and C_{kc} is the cost for fleet k to operate in cell c. Based on the weights in Equation 32 the total mortality rate is distributed over cells according to

$$F_{kc} = \frac{N \cdot F_k \cdot G_{kc}}{\sum_c G_{kc}}$$

...33)

while each group in the cell is subject to the total fishing mortality

$$F_{ic} = \sum_k F_{kc} \cdot q_{ki}$$

...34)

Numerical solutions

Ecospace is based on the same set of differential equations as used in Ecosim, and in essence performs a complete set of Ecosim calculations for each cell for each time step. This represents a formidable amount of computations, but fortunately it has been possible to take a number of shortcuts to speed the processing up to an acceptable rate. Briefly explained the background for this takes its starting point in Equation 15, which expresses the rate of change for each biomass pool over time. If the rate constants were constant over time (they are not, but if!) the biomass would change as a linear dynamical system, and would move exponentially towards an equilibrium given by (omitting indices for cell and biomass pools)

$$B_e = \frac{1 + gC}{Z + E}$$

...35)

while following the time trajectory

$$B_{t+\Delta t} = B_e + (B_t - B_e) \cdot e^{-(Z+e)\Delta t}$$

...36)

Denoting the exponential weight term above W_t this can be re-expressed as,

$$B_{t+\Delta t} = W_t \cdot B_t + (1 - W_t) \cdot B_e$$

...37)

Hence, if input and output rates were constant, the time solutions would behave as weighted averages of past values and equilibrium values with weights depending on the mortality and migration rates. Using expressions of the type in Equation 37 the Ecospace computations can be greatly increased by using a variable time splitting where moving equilibria are calculated for groups with high turnover rates, (e.g., phytoplankton), while the integrations for groups with slower turnover rates, (e.g., fish and marine mammals) are based on a Runge-Kutta method. Comparisons indicate that this does not change the resulting time patterns for solutions in any noticeable way – hence, the 'wrong' assumption of time rate constancy introduced above is useful for speeding up the computations without noticeable detraction of the final results. The resulting computations are carried out orders of magnitude faster than if the time splitting was not included.

Advection in Ecospace

Advection processes are critical for productivity in most ocean areas. Currents deliver planktonic production to reef areas at much higher rates than would be predicted from simple turbulent mixing processes. Upwelling associated with movement of water away from coastlines delivers nutrients to surface waters, but the movement of nutrient rich water away from upwelling locations means that production and biomass may be highest well away from the actual upwelling locations. Convergence (down-welling) zones represent places where planktonic production from surrounding areas is concentrated, creating special opportunities for production of higher trophic levels.

Ecospace provides a user interface for sketching general current patterns or wind/geostrophic forcing patterns for surface currents. Based on these patterns Ecospace calculates equilibrium horizontal flow and upwelling/down-welling velocity fields that maintain continuity (water mass balance) and effects of Coriolis force. That is, the advection field is calculated by solving the linearized pressure field and velocity equations $dh/dt = 0$, $dv_u/dt = 0$, $dv_v/dt = 0$ across the faces of each Ecospace grid (u,v) cell, where f is sea surface anomaly, the v's are horizontal and velocity components (u, v directions) and the rate equations at each cell face satisfy (omitting grid size scaling factors for clarity):

$$\frac{dh}{dt} = \frac{v_{uh}}{u} + \frac{v_{vh}}{v} - D_h \quad \text{...38)}$$

$$\frac{dv_u}{dt} = k \cdot W_u - k \cdot v_u - f \cdot v_v - \frac{g \cdot h}{u} \quad \text{...39)}$$

$$\frac{dv_v}{dt} = k \cdot W_v - k \cdot v_v - f \cdot v_u - \frac{g \cdot h}{v} \quad \text{...40)}$$

Here, the W's represent the user sketched forcing or general circulation field, h represents sea surface anomaly, k represents bottom friction force, f the Coriolis force, D represents downwelling/upwelling rate, and g acceleration due to sea surface slope.

Solving these equations for equilibrium is not meant to be a replacement for more elaborate advection models; generally the W_u and W_v need to be provided either by such models or by direct analysis of surface current data, so the Ecospace solution scheme is only used to assure mass balance and correct for 'local' features caused by bottom topography and Coriolis forces. That is, absent shoreline, bottom, and sea surface anomaly (h) effects, the equilibrium velocities are just $v_u = W_u$, $v_v = W_v$ up to corrections for Coriolis force. We could just allow users to input the W fields and then calculate upwelling/downwelling rates needed to satisfy these, but solving the equations using general forcing sketches of W patterns allows us to internally correct for factors such as topographic steering of currents near shorelines, without demanding that the user enter W fields that precisely maintain mass balance (and/or correct upwelling/downwelling velocities) absent any correction scheme.

Once an advection pattern has been defined, the user can specify which biomass pools are subject to the advection velocities (v_u,v_v field) in addition to movement caused by swimming and/or turbulent mixing. This allows examination of whether some apparent 'migration' and concentration patterns of actively swimming organisms, (e.g., tuna aggregations at convergence zones) might in fact be due mainly to random swimming combined with advective drift.

Capabilities and limitations

EwE has been developed largely through case studies, where users have challenged us to add various capabilities and as we have seen inadequacies through comparison to data; see as a good example the discussions in the proceedings from an FAO workshop on the application of EwE (Pauly 1998). Various capabilities have been added to EwE in response to these challenges, and there has inevitably been some uncertainty about what the approach and software presently can and cannot do, and about how it should be used in the design of sustainable fisheries policies. Such uncertainty may be expressed through too simplistic interpretations of what mass balance and biomass dynamics models are capable of representing, through to unwarranted optimism about how it should be used to replace or complement existing assessment tools. Here we review the capabilities and limitations through a series of 'frequently asked questions', followed by explanations of what we think EwE is actually capable of doing.

Note that many of the questions discussed below have their root in an assumption that EwE is somehow intended to supplant or replace single-species assessment methods. Our primary goal when developing EwE has been to develop a capability for asking policy questions that simply cannot be addressed with single-species assessment. Examples are questions about impacts of fishing on nontarget species, and the efficacy of policy interventions aimed at limiting unintended side effects of fishing. Also, as is shown through examples below, EwE can now incorporate time series data from single-species assessment as input and use these for parameter fitting. We indeed advocate an iterative process where information is passed between single-species analysis and EwE to check and improve estimates in the process, addressing questions about the degree to which ecosystem events can and cannot be attributed to impact of fisheries, climate change, etc.

Does Ecopath assume steady state or equilibrium conditions?

Ecopath provides an 'instantaneous' estimate of biomasses, trophic flows, and instantaneous mortality rates, for some reference year or multi-year averaging window. Biomasses need not be at equilibrium for the reference year, provided the Ecopath user can provide an estimate of the rate of biomass 'accumulation' (or depletion) for each biomass for that reference year. In fact, in a number of cases, e.g., Christensen (1995a) it was necessary to recognize that biomasses were in fact changing over the period for which Ecopath reference data (B, P/B, Q/B, diet composition) were provided. In these cases, assuming equilibrium for the reference year led to overly optimistic estimates of sustainable fishing mortality rates.

Should Ecopath be used even if there is insufficient local information to construct models, or should more sampling go first?

It is a fairly common conception that since we do not know enough to make perfect models at the individual or species level there is no way we can have enough information at hand to embark on modeling at the ecosystem level. This may hold if we try to construct models bottom-up – we cannot account for all the actions and processes involving all the individuals of the world. This is, however, not what Ecopath models do, instead they place piecemeal information in a framework that enable evaluation of the compatibility of the information at hand, gaining insights in the process. Adding to this is that there is much more information of living marine resources available than most will anticipate. The best demonstration of this can be obtained by searching the FishBase database on finfish (Froese and Pauly 2000, www.fishbase.org) for Ecopath-relevant information using the semi-automated search routine available for the specific purpose at the website.

Another aspect is that ecosystem models can help direct research by pinpointing critical information and gaps in the present knowledge. As more information becomes available it is straightforwardly included in the model, improving estimates and reducing uncertainty (see above on '**Addressing uncertainty**').

Does EwE ignore inherent uncertainty in assembling complex and usually fragmentary trophic data?

Ecopath has a number of routines that encourage users to explore the effects of uncertainty in input information on the mass balance estimates. In particular, the 'Ecoranger' routine allows users to calculate probability distributions for the estimates when they specify probability distributions for the input data components. Similarly, Ecosim has a graphical interface that encourages policy 'gaming' and sensitivity testing.

Lack of historical data and difficulty in measuring some ecosystem components and processes will likely always plague efforts to understand trophic structure and interactions. This is not just a problem with Ecopath, but rather with aquatic ecology in general (ludwig et al. 1993). We need to respond to it not by complaining about the incompleteness of our data, but rather by using models like EwE to direct attention toward components that are most uncertain and also make the most difference to policy predictions. We also need to use the models to search for robust policy options and management approaches that will allow us to cope with the uncertainty, rather than pretending that it will just go away.

When EwE is used for policy comparison, it is important to recognize that incorrect comparisons (EwE leading user to favor a wrong policy) are not due to uncertainty in general about the model parameters, but rather to errors in specific input data to which the particular policy comparison is sensitive. In other words, EwE can give correct answers for some policy comparisons but wildly incorrect ones for others, so it is meaningless to claim that it should not be used because of uncertainty in general. For example, EwE predictions of the impact of increasing fishing rates for a particular species are most sensitive to assumptions about vulnerability of prey to that species, since the vulnerability parameters largely determine the strength of the compensatory response by the species to increased mortality rate. But even if EwE predicts the strength of the compensatory response to fishing correctly, it may still fail to predict the response of that same species to a policy aimed at increasing its productivity by reducing abundance of one or more of its predators: EwE may have a good estimate of total mortality rate for the species, but a very poor estimate of how that mortality rate is distributed among (or generated by) predators included in the model.

Can Ecopath mass balance assessments provide information directly usable for policy analysis?

Instantaneous snapshots of biomass, flows, and rates of biomass change have sometimes been used to draw inferences about issues such as ecosystem health as measured by mean trophic level or other indices of fishing impact, (e.g., Christensen 1995b, Pauly and Christensen 1995, Pauly et al. 1998). But the snapshots cannot be used directly to assess effects of policy changes that would result in changes in rates, (e.g., reduction in fishing rates) since the cumulative effects of such changes cannot be anticipated from the system state at one point in time. In fact the Ecosim part of EwE was initially developed specifically to provide a method for predicting cumulative changes, while recognizing that all rate processes in an ecosystem may change over time, as biomasses change. For example, one might conclude from the Ecopath mortality rate estimates or mixed trophic impact analysis (see above) that reducing the abundance of some particularly important predator might result in lower mortality rates of its prey, and hence growth in abundance of these prey. This prediction may hold for a short time, but might be reversed entirely over longer time scales due to increases in abundance of other predators or on an intermediate time scale due to predator prey switching in response to the initial responses in prey density.

Can Ecopath provide a reliable way of estimating potential production by incorporating knowledge of ecosystem support capabilities and limits?

Ecologists have long sought simple ways of predicting productive potential of aquatic ecosystems from 'bottom up' arguments about efficiency of conversion of primary production into production of higher trophic levels, (e.g., Polovina and Marten 1982). While Ecopath inputs can be organized so as to provide such predictions, we do not recommend using EwE for management this way. There are simply too many ways that simple efficiency predictions can go wrong, particularly in relation to 'shunting' of production into food web components that are not of direct interest or value in management, (e.g., ungrazeable algae, fish species that are not harvested). Ecopath can help provide broad bounds for potential abundances and production in an exploratory research mode, but these bounds are unlikely to be tight enough to be useful for management planning related to fishery development or recovery potential.

Can Ecopath predict biomasses of groups for which no information is available?

In most EwE applications today, we try very hard to avoid using the Ecopath biomass estimation capability for more biomass components than absolutely necessary. Estimation of biomass with Ecopath usually requires making explicit assumption about the ecotrophic efficiency, i.e., about the proportion of the total mortality rate of a group that we account for by the predation, migration, biomass accumulation and fishing rates included explicitly in the Ecopath data. There is rarely a sound empirical basis for using any particular value of EE, except perhaps for top predators in situations where total mortality rate (Z=P/B) is well estimated and EE represents a 'known' ratio of fishing rate (F) to total Z (and the rest of Z, e.g., the natural mortality (M) is known not to be due to other predators included in the model nor to other factors not considered).

Where biomasses really are unavailable or are known to be biased, e.g., if the only biomass estimates for pelagics are from swept-area analysis based on demersal trawling, it may still be better to use assumed EE's than to stop short of constructing an ecosystem model pending, e.g., funding and development of capabilities to conduct acoustic surveys. In such cases one can assume reasonable EE values for groups where biomasses are missing – an example: small pelagics do not die of old age in exploited ecosystems, most are either eaten or caught, hence EE is likely to be in the range 0.90 to 0.99. As confidence intervals can be assigned to all input parameters and can be estimated for the output parameters using the Ecoranger module of EwE (where a range for acceptable output parameters is also incorporated as part of the model evaluation process), the mass balance constraints of the model can be used to predict potential ranges for biomasses of the species in the system.

Should Ecopath mass balance modeling be used only in situations where data are inadequate to use more detailed and realistic methods like MSVPA?

Multispecies virtual population analysis (MSVPA) has been used to reconstruct age-size and time dependent estimates of trophic flows and mortality rate components, using the VPA assumption that historical abundances can be inferred by back-calculating how many organisms must have been present in order to account for measured and estimated removals from those organisms over time (Sparre 1991, Magnússon 1995). In a sense, Ecopath does this as well, but generally does not account for size-age dependency and temporal variation (biomasses are constrained to be large enough to account for assumed removals estimated from biomasses, consumption/biomasses, and diet composition of predators, just as in MSVPA).

But the really big difference between Ecopath and MSVPA is not in the detail of calculations; constructing an Ecopath model that details age, size and time components is tedious but feasible. The more important difference is in the use of total mortality rate as input data by Ecopath, in the form of the P/B ratio that Ecopath users must provide. Ecopath biomass and mortality estimates are 'constrained' to fit the total mortality rates entered as P/B data. In contrast, MSVPA (like single-species VPA) can produce cohort abundance patterns (die-off patterns over age-size and time) that do not agree in any way with apparent cohort decay patterns evident from direct examination of the size-age composition data. In effect, the MSVPA (and VPA) user must reject or ignore any direct evidence about total mortality rate Z that might be present in age-size composition data, and must treat discrepancies between apparent Z from the cohort reconstructions versus apparent Z from composition data as being due to size-age dependent changes in vulnerability to the composition sampling method. This can be unwise, just as it has been unwise to ignore information about Z in single-species VPA, (e.g., Newfoundland cod VPA's resulted in much lower estimates of Z than would be estimated from catch-curve analysis of the age composition data, and in this case it turned out that VPA tuning resulted in underestimates of fishing mortality rate, see, e.g., Walters and Maguire, 1996).

It is obviously comforting to us as biologists to be able to provide more detailed accounting of predation interactions, which are almost always size and age dependent. But in assessments of ecosystem-scale impacts of changes in trophic conditions, it is not automatically true that the best aggregate estimate is the sum of component estimates, any more than it is automatically true in single-species assessment that more detailed models and data always provide better assessments than simpler models. For statistical and logical reasons, the 'more is better' argument is no more valid in dynamic modeling than it is in multiple regression analysis, where we are familiar with how adding more independent variables is often an invitation to better fits but poorer predictions.

As noted in the following two points, Ecopath and Ecosim do not 'ignore' the fact that trophic interactions are strongly size-age and seasonally structured. Rather, we assume that initial (Ecopath base or reference period) structuring has been adequately captured in preparing average/total rate input data, and that changes in structural composition over time are not large enough to drastically and persistently alter interaction rates/parameters. This is very similar to the assumption in single-species biomass dynamics and delay-difference modeling that stock composition changes produce regular or predictable changes in overall (stock-scale) production parameters, not that there is no composition effect in the first place.

Do EwE models ignore seasonality in production, mortality, and diet composition?

In most applications, Ecopath calculates components of biomass change over a one-year accounting step. There is no explicit assumption about how mortality rates, consumption rates, and diet composition may have varied within this step, except that the Ecopath user is assumed to have calculated a correct, weighted average of the rates over whatever seasonality may have been present in the data. Such averages can be difficult to calculate in practice, and a program interface component has been developed to help users with this chore (Martell 1999).

In Ecosim, model users can define seasonal 'forcing shapes' or functions that can be applied as seasonal multipliers to the modeled production and consumption rate functions. Generally, including seasonal variation in this way results in graphics displays that are hard to follow visually (strong seasonal oscillations in ecosystem 'fast' variables like phytoplankton concentration), but very little impact on predicted interannual (cumulative, long term) patterns of system change.

Do biomass dynamics models like Ecosim treat ecosystems as consisting of homogeneous biomass pools of identical organisms, hence ignoring, e.g., size-selectivity of predation?

The biomass rate equations in Ecosim (sums of consumption rates less predation and fishing rates) can be viewed as 'sums of sums', where each trophic flow rate for an overall biomass pool is the sum of rates that apply to biomass components within that pool. In this view, doing a single overall rate calculation for a pool amounts to assuming that the proportional contributions of the biomass components within the pool remain stable, i.e., the size-age-species composition of the pool remains stable over changes in predicted overall food consumption and predation rates. In fact, the assumption is even weaker: pool composition may indeed change over time provided that high and low rate components change so as to balance one another, or proportional contribution of major components is stable enough so that total rates per overall biomass are not strongly affected.

We know of at least one condition under which the compositional stability assumption may be violated – when ratios of juvenile to adult abundance can change greatly, (e.g., under changes in fishing mortality) for a species that has strong trophic ontogeny (very different habitat use and trophic interactions by juveniles). To deal with such situations, Ecosim allows model users to 'split' biomass pools representing single-species with strong trophic ontogeny, into 'juvenile' and 'adult' pools. If so, the Ecosim biomass dynamics equations are replaced with an explicit age structured model for monthly age cohorts in the juvenile pool, and a delay-difference model for the adult pool. That is, for 'split pool' species Ecosim replaces the biomass dynamics model with a much more detailed and realistic population model, (see 'Life history handling' on page 87). This allows Ecosim users to not only represent compositional effects, but also to examine the emergent stock-recruitment relationship caused by density-dependent changes in adult fecundity and juvenile growth and foraging time behavior.

Do ecosystem biomass models ignore behavioral mechanisms by treating species interactions as random encounters?

Historically, trophic interaction rates in biomass dynamics models have been predicted by treating predator-prey encounter patterns as analogous to 'mass-action' encounters between chemical species in chemical reaction vat processes, where reaction (encounter, 'predation') rates are proportional to the product of predator and prey densities. Such 'Lotka-Volterra' models generally predict much more violent dynamic changes, and considerably simpler ecosystem organization, than we see in field data.

Ecosim was constructed around the proposition that this mass-action principle is incorrect for ecological interactions, and instead interactions take place largely in spatially and temporally restricted 'foraging arenas' where prey make themselves available to predation through activities such as foraging and dispersal. To represent this within-pool heterogeneity, we treat each biomass pool as consisting at any instant of two biomass components with respect to any predator, one sub-pool of individuals vulnerable to the predator and another sub-pool 'safe' from the predator. In this view, predation rate is limited jointly by search efficiency of the predator for vulnerable prey individuals, and exchange rate of prey between the invulnerable and vulnerable states. When Ecosim users set the vulnerability exchange rates to high values, the model moves toward 'top down' or mass-action control of predation rates. When users set the vulnerability rates to low values, the model moves toward 'bottom up' control where predation rates are limited by how fast prey move (or grow, or disperse) into the vulnerable state.

Obviously the two-state (vulnerable / invulnerable) representation of prey biomass composition is a first approximation to the much more complex distribution of vulnerabilities among prey individuals that is likely to be present in most field situations. But it goes a remarkable way toward explaining dynamic patterns (lack of predator -prey cycles, persistence of apparent competitors and high biodiversity) that we have been unable to explain with simpler Lotka-Volterra mass-action models.

Do Ecosim models account for changes in trophic interactions associated with changes in predator diet compositions and limits to predation such as satiation?

In nature, diet compositions and feeding rates can change due to five broad factors:

1. changes in 'habitat factors' such as water clarity, temperature, and escape cover for prey;
2. changes in prey abundance and activity, and hence encounter rates with predators;
3. changes in predator abundance, and hence interference/exploitation competition for localized available prey;
4. changes in predator search tactics (search images, microhabitat used for foraging);
5. handling time or satiation limitations to predator feeding rates.

Ecosim allows (or requires) representation of four of these factors, namely all but predator search tactic changes (4). Type (1) factors can be optionally introduced by including 'time forcing' functions representing temporal habitat change, and/or 'trophic mediation' functions where other biomasses modify predation interaction rates for any predator-prey pair(s). Types (2), (3), and (5) are built into the calculations by default (though some effects can be disabled by particular parameter choices).

In Ecosim, changes in prey abundance (factor (2) above) lead to proportional changes in predator diet composition only when prey feeding times are deliberately held constant by 'turning off' Ecosim foraging time adjustment parameters. When prey foraging time is allowed to vary (default assumption), declines in prey density generally result in apparent sigmoid (type III) decreases in predator consumptions of that prey type: as the prey declines, it generally spends less time feeding (reduced intraspecific competition for its own prey) and hence reduced encounter rates with its predators. The user can exaggerate this sigmoid effect by turning on parameters that cause the prey to spend less time feeding when predation risk is high (i.e., direct response to perceived predation risk).

Predator satiation effects are represented in Ecosim by foraging time adjustments such that predators 'try' to maintain constant food consumption rates (unless foraging time adjustments are deliberately disabled), by spending more time feeding when feeding rates begin to decrease due to decreasing densities of one or more prey types. Likewise, handling time limits to feeding rate (lower attack rate on any one prey type as abundance of another increases, due to predator spending more time pursuing/handling individuals of the other type) are represented by a 'multispecies disc equation' (generalization of Holling's type II functional response model).

Our philosophy in developing Ecosim predation rate predictions has been to look first at the fine-scale (space, time) behavioral ecology of prey and predators, and in particular at how they vary and 'manage' their time. Overall predation response patterns, such as Type II sigmoid effects of reduced prey density, then 'emerge' as effects of the time management representation rather than being 'hardwired' into the model by particular overall equations for predation rates and diet composition.

Are the population models embedded in Ecosim better than single-species models since they explain the ecosystem trophic basis for production?

In a number of case studies, Ecosim users have treated the model as though it were a single-species assessment tool, varying its parameters so as to fit time series data for a particular species, (e.g., yellowfin tuna in the Eastern Pacific, herring in southern British Columbia). In such cases, it generally turns out that the biomass dynamics or delay-difference 'submodel' for the target species behaves quite similarly when 'embedded' in Ecosim (with explicit accounting for production and mortality rate as function of food resources and predators) to the corresponding single-species assessment model where competition effects are represented as implicit functions of stock size, (e.g., stock recruitment model) and predation mortality rates are assumed constant.

So if one has an Ecosim model whose 'production' parameters have been estimated by fitting the model to single-species data, and a corresponding single-species model also fitted to the data, one should not be surprised that the two approaches usually give about the same answers to policy questions related to changing fishing mortality rate for the species, (e.g., fishing rates for MSY). Ecosim models may diverge from the single-species predictions at very low stock sizes (Ecosim may predict 'delayed depensation' effects due to changes in predation rates on juveniles), but otherwise do not generally lead us to interpret the single-species data any differently with respect to single-species assessment issues, (e.g., MSY) than if we just used the single-species model.

Thus, it would be wrong when applying Ecosim for single-species harvest policy analysis to contend that Ecosim is 'better' than a single-species model, when both give the same answer. It may comfort us as biologists to know that the Ecosim representation has somehow explained production in terms of ecosystem relationships rather than implicit relationships on stock size, but making biologists 'feel better' should not be a criterion for judging the effectiveness of a policy tool. When fitting Ecosim to the data we encounter the same risks as in single-species assessment of incorrect biomass estimation, misinterpretation of trend data, (e.g., hyperstability of catch per effort data), and failure to account for persistent effects such as environmental regime changes or confounding of these effects with the effects of fishing.

Does Ecosim population models provide more accurate stock assessments than single-species models by accounting for changes in recruitment and natural mortality rates due to changes in predation rates?

As noted above, using Ecosim for single-species assessments usually results in similar fits to historical data as would be obtained with traditional surplus production or delay-difference models. In principle Ecosim should be able to improve a bit on models that assume stationary stock-recruitment relationships and constant natural mortality rates, at least for mid-trophic level species that may be subject to highly variable predation risk. But in practice we have so far not obtained substantial improvements in fit to data, which could be due to poor data or to stability in mortality rates of the sort predicted when Ecosim vulnerability parameters are set to mimic 'bottom up' control of predation rates.

In one case (the Strait of Georgia, British Columbia) where we have fit Ecosim to multiple time series data on major species (herring, salmon, hake, ling cod, seals) by estimating 'shared production anomalies' attributed in the fitting to changes in primary productivity, we were able to show that about half the total variance around single-species model fits to changes in relative abundance over time could be explained by ecosystem-scale effects. That is, we were able to 'improve' on the single-species fitting, but this improvement was due to assuming changes in ecosystem scale 'forcing' rather than to accounting for temporal variation in predation mortality rates associated with impacts of fishing on predators. In another case (French Frigate Shoals, Hawaii) we were again able to fit time series data (rock lobsters, monk seals) better by including effects of an ecosystem-scale regime shift (decreased primary production in the Central North Pacific after 1990), and were not able to explain deviations from single-species model fits through changes in trophic interactions alone.

These cases, along with experience that Ecosim generally does not behave much differently from single-species models when only fishing effects are considered, lead us to suspect that Ecosim (and perhaps other, more detailed trophic interaction assessments) will not lead to substantial improvements in stock size prediction just by accounting for predator-prey effects. However, there is a good chance that Ecosim will be very helpful in interpreting effects of large-scale, persistent regime changes that are likely to have caused ecosystem-scale changes in productivity. In such situations, Ecosim may be particularly helpful in finding some resolution for the so-called 'Thompson-Burkenroad' debates about the relative importance of fishing versus environmental changes in driving historical changes in abundance.

Rather than pretending that Ecosim and single-species methods are competitors, a useful assessment tactic may be to work back and forth between Ecosim and single-species assessment methods, using each to check and improve the other. For example, we have used ordinary VPA and stock synthesis results for Pacific herring as reference 'data' (summary of raw age composition, harvest, and spawn survey data) for fitting Ecosim models of the Georgia Strait. The Ecosim herring model predicts somewhat lower abundances than VPA during periods of low stock size, and somewhat higher abundances than VPA during high stock periods. Ecosim also estimates lower natural mortality rates (M) for herring during the low abundance periods. If Ecosim is correct in estimating that M has been (weakly) density-dependent, then VPA has probably overestimated abundance (used too high an M in the VPA backcalculation) during population lows, and is probably underestimating juvenile abundance now (due to using an M that is too low for the current high stock size).

Can one rely on the Ecosim search procedure time series fitting to produce better parameter estimates?

Ecosim users are cautioned that the search procedure in no way guarantees finding 'better' Ecosim parameter estimates. Better fits to data can easily be obtained for the wrong reasons (some time series, particularly catch/effort data, can be misleading in the first place, as can historical estimates of changes in fishing mortality rates; many parameter combinations may equally well 'explain' patterns in the data). Nonlinear search procedures can become lost or 'trapped' at local parameter combinations where there are local minima in the SS function far from the combinations that would actually fit the data best. The best way to insure against the technical problems of searching a complex SS function is to use 'multiple shooting': start the search from a variety of initial parameter combinations, and see if it keeps coming back to the same final estimates. Look very closely at the time series data for possible violations of the assumption that the relative abundance, y, is a product of a scaling factor and the total biomass, due to progressive changes in the methods of estimating y or nonlinearities caused by factors such as density-dependent catchability. If y is a biomass reconstruction from methods such as VPA that assume constant natural mortality rate M, spurious trends in y caused by the sort of changes in M that Ecosim predicts, particularly for younger animals, call for concern. Alternative combinations of Ecosim parameters may fit the data equally well but would imply quite different responses to policy changes such as increases in fishing rates.

Search procedures are most useful in diagnosing problems with both the model and data. That is, the greatest value of doing some formal estimation is while it seems not to be working, when it cannot find good fits to data. Poor fits can be informative about both the model and the data.

Does Ecosim ignore multispecies technical interactions (selectivity or lack of it by gear types) and dynamics created by bycatch discarding?

By separating groups into juveniles and adults, each with different biomasses and catches (and hence fishing mortalities), fundamental differences in selection can be accounted for. Moreover, Ecosim users can specify fishing mortality patterns over time either at the group level (fishing rate for each group over time) or the fleet level. Fleet level changes are specified as changes in relative fishing effort (relative to the Ecopath baseline model), and these changes impact fishing rates for the species caught by each gear in proportion to Ecopath base estimates for the species composition of the gear. That is, technical interactions (fishing rate effects on a variety of species caused by each gear type) are a basic part of the Ecopath data input and Ecosim simulations. However, Ecosim does not provide simple scenario development options for simulating tactics that might make each gear more or less selective in future.

Discarded bycatch can be treated as a biomass pool in Ecopath, i.e., as a diet component (and hence component of production) by species that consume discards (e.g. sharks, birds, shrimp). Ecopath input data on bycatch and discard rates are passed to Ecosim, and Ecosim does time accounting for changes in discard rates and biomass in relation to simulated changes in fishing fleet sizes. In scenarios where some species are heavily dependent on bycatch, Ecosim will then track impacts of bycatch management on food availability and feeding rates of such species. For instance, Ecosim has produced some very interesting scenarios for shrimp fishery development and how shrimp often appear to become more productive under fishing, by including effects of both reducing abundance of predatory fishes (when they are killed as bycatch) and providing biomass from those fishes as food for the shrimp.

Does Ecosim ignore depensatory changes in fishing mortality rates due to range collapse at low stock sizes?

Ecosim users have two options for specifying fishing mortality rate patterns: (1) direct entry of fishing rate (F) values over time; or (2) entry of relative fishing effort values over time, with fishing rate calculated as q(B)·(relative effort), where q(B) is a biomass-dependent catchability coefficient. Under the second option, q is modeled as a hyperbolic function of B ($q = q_{max} / (1 + kB)$), so that q can be increased dramatically with decreases in stock size. The concept in this formulation is to recognize that catchability q can be expressed as a ratio q = a / A, where a is the area swept by one unit of effort and A is the area over which fish are distributed. Increases in q with decreasing stock biomass are usually assumed to be caused by decreases in stock area A occupied with decreases in B.

Does Ecosim ignore the risk of depensatory recruitment changes at low stock sizes?

Depensatory recruitment changes are apparently not common (Myers et al. 1995, Liermann and Hilborn 1997), but should not be ignored in risk assessments for situations where a depensatory recruitment decline would have large economic or social consequences. Depensatory effects are usually assumed to be due to Type II predator feeding effects, where predators would exert an increasing mortality rate on juvenile fishes if they tend to eat a constant number of juveniles despite decreasing juvenile density. There are relatively few field situations where we would expect such type II predator feeding effects (like migrating pink salmon fry being eaten by resident trout in a small stream).

Ecosim has helped identify another possible depensation mechanism that may be more common, which we call the 'delayed depensation' or 'cultivation-depensation' effect (Walters and Kitchell in press). When a large, dominant species is fished down in Ecosim models, the model often predicts a substantial increase in smaller-sized predators that have been kept down in abundance by a combination of direct predation and competition effects with the large dominant species. These predators then cause an increase in predation mortality rate on (or compete for food with) juveniles of the large dominant. This causes a depensatory decrease in the recruitment rate per spawner for the large dominant, slowing or preventing population recovery even if the fishing effects are removed.

So far from ignoring depensatory recruitment effects, Ecosim warns us to be more careful about the risk of these effects. It warns us to be especially wary in the management of the most common, large, and dominant fish species that are the most valuable components of most fisheries.

Major Pitfalls in the Application of EwE

EwE can produce misleading predictions about even the direction of impacts of policy proposals. Erroneous predictions usually result from bad estimates or errors of omission for a few key parameters, rather than 'diffuse' effects of uncertainties in all the input information. We warn EwE users to be particularly careful about the following problems that we have seen in various case studies.

Incorrect assessments of predation impacts for prey that are rare in predator diets

It is easy to overlook a minor diet item in specifying diet composition for some predator. Unfortunately, while that prey type may not be important for the predator, it may represent a very large component of total mortality for the prey type. This is a particularly important problem in representation of mortality factors for juvenile fishes, which usually suffer high predation mortality rates but are often not major components of any particular predator's diet and are notoriously difficult to measure in diet studies (fast digestion rates, highly erratic and usually seasonal occurrence in predator diets).

Another way that 'minor' diet items can come to assume considerable importance is through 'cultivation-depensation' effects (Walters and Kitchell in press). Suppose for example that some small predatory fish is kept at low densities by another, larger predator, but the number of predation events needed to exert this control is small compared to the total prey consumption by the larger predator. It would be easy to miss this linkage entirely in formulating the initial Ecopath model. But then suppose the larger predator is fished down, 'releasing' the smaller predator to increase greatly in abundance. The smaller predator may then cause substantial decrease in juvenile survival rates of the larger predator, creating a 'delayed depensation' effect on the larger predator's recruitment. Possibly the larger predator was abundant in the first place at least partly because it was able to exert such control effects on predators/competitors of its own juveniles. Even if such 'perverse' trophic interactions are rare, they are certainly worth worrying about because they imply a risk that overfishing will result in delayed recovery or a persistent low equilibrium abundance for larger predators.

Trophic mediation effects (indirect trophic effects)

We use the term 'mediation effect' for situations where the predation interaction between two biomass pools is impacted positively or negatively by abundance of a third biomass type. For example, predation rates on juvenile fishes by large piscivores may be much lower in situations where benthic algae, corals, or macroinvertebrates provide cover for the juveniles. Pelagic birds like albatrosses that feed on small fishes may depend on large piscivores to drive these small fishes to the surface where they are accessible to the birds. Some large piscivores may create enough predation risk for others to prevent those others from foraging on some prey types in some habitats.

When a mediation effect is in fact present but is not recognized in the Ecosim model development, it is not unlikely for the model to predict responses that are qualitatively incorrect. For example, fishing down tunas in a pelagic model is likely to result in predicted increases in abundance of forage fishes, and hence to predicted increases in abundance of pelagic birds. But in fact, reducing tuna abundance may have exactly the opposite effect, resulting in bird declines due to the baitfish spending less time at the surface when tuna are less abundant.

Underestimates of predation vulnerabilities

Predation impacts can be limited in Ecosim by assuming low values of the exchange parameters (v's) between behaviorally invulnerable and vulnerable prey 'states'. We call these exchange parameters 'vulnerabilities', and they are estimated by assuming ratios of maximum to Ecopath base estimates of prey mortality rates for each predator-prey linkage. That is, if $M(0)_{ij} = Q(0)_{ij} / B(0)_i$ is the base instantaneous natural mortality rate for prey type i caused by predator j base (Ecopath estimate) consumption rate $Q(0)_{ij}$ on prey base biomass $B(0)_i$, we assume that the maximum possible rate for very high predator j abundance would be $v_{ij}B_i$ where $v_{ij} = KM(0)_{ij}$, K>1, represents the rate at which prey become vulnerable to predator j. By using a K near 1, i.e. v_{ij} only a little larger than $M(0)_{ij}$, Ecosim users can simulate the 'bottom up' control possibility that changes in predator abundances do not cause much change in prey mortality rates because these rates are limited by physiological or behavioral factors of the prey. The assumption that there are such limitations is supported by scattered observations where total mortality rates (Z) were poorly correlated with changes in predator abundances.

Another way of saying that vulnerabilities of prey to predators are very limited is to say that predators are already eating almost every prey that does become vulnerable. If this is indeed true, then there is likely intense exploitation competition among predators for the prey that do become vulnerable, i.e. the number of vulnerable prey seen by each predator is severely limited by the number of other predators competing for those prey. This has potentially large implications for the dynamics of the predator: reductions in predator abundance may be accompanied by large increases in the densities of vulnerable prey available to each remaining predator. In such cases, Ecosim will predict a strong compensatory effect on the predator of reduced predator abundance (strong increases in food consumption rate and growth, or large decreases in predator foraging time with attendant decreases in mortality risk faced by the predator).

So the net effect of assuming low prey vulnerabilities is also to assume that predators should exhibit strong compensatory responses to reduced abundance of conspecifics, which in simulations of increased fishing pressure means strong compensatory responses and hence lower risk of overfishing. An enthusiastic proponent of 'bottom up' control of trophic processes must therefore also be a strong proponent of the idea that it is hard to overfish. This is a very risky assumption.

Non-additivity in predation rates due to shared foraging arenas

The default assumption in Ecosim is to treat each predation rate linkage as occurring in a unique 'foraging arena' defined by the behaviors of the specific prey and predator. In this formulation, elimination of one predator will result in a decrease in total prey mortality rate equal (at least initially) to the Ecopath base estimate of that predator's component of the prey total mortality rate. This may be partly compensated by increases in mortality rate due to other predators if the prey increases in abundance and spends more time foraging in response to increased intraspecific competition, but in general this compensatory effect will not completely replace the initial mortality rate reduction.

But suppose this formulation is wrong, and in fact the mortality rate of the prey represents movement of the prey into behavioral or physiological states (e.g. parasite loads) for which it is vulnerable to predators in general. In this case, removal of any one predator may simply result in the vulnerable prey individuals being taken just as fast, but by other predators. In this case, the total mortality rate of the prey will change much less than predicted by Ecosim.

For example, we recently used Ecosim to evaluate whether control of predatory sharks might help improve juvenile survival rates of monk seals off Hawaii. Sharks appear to be the proximate cause of many juvenile deaths, and it appears that juveniles are exposing themselves to much higher predation risk than normal due to decreases in prey abundance caused by a combination of lobster fishery and ocean productivity ('regime shift') effects. In this case, Ecosim predicts that shark control will at least temporarily improve monk seal juvenile survival rates. But if the real problem is not sharks, but rather that juvenile seals are spending more time exposed to predators in general, the Ecosim prediction about efficacy of control may be grossly optimistic: other predators may just take up the 'slack' after shark removal.

Temporal variation in species-specific habitat factors

Attempting to fit Ecosim models to time series data has revealed some cases where an important species or biomass pool shows dramatic change that cannot be attributed to any known change in trophic relationships or harvesting. Then this dramatic but 'unpredictable' change appears to result in major trophic impact on the rest of the ecosystem. An example would be a planktivorous fish species that is important to piscivores in the system (so piscivores respond strongly to changes in its abundance), which shows high recruitment variation and occasional very strong year classes that support temporary piscivore increases. It is quite possible for such recruitment 'events' to be linked to very localized habitat factors that affect juvenile survival of the planktivore, so that each event results in a persistent cascade of abundance changes throughout the food web. Another example would be loss of specific spawning sites or habitat for one species, that causes it to decline despite favorable trophic conditions in terms of food supply and predation risk.

Ecosim can help us detect possible habitat problems, by revealing prediction 'anomalies' from biomass patterns expected under trophic and fishing effects alone. But there is also a risk of producing 'spurious' good fits to Ecosim, when Ecosim parameters are varied so as to explain as much of the biomass change as possible; that is, Ecosim may explain patterns as trophic/fishing effects that in fact have been due to habitat changes. This is a particular risk in situations where habitat change involves some fairly regular 'regime shifts' or cycles in habitat variables; Ecosim may well attribute cyclic biomass changes in such situations to predator-prey instabilities rather than environmental forcing.

ACKNOWLEDGEMENT

We thank the Pew Charitable Trust for funding the 'Sea Around Us' project at the Fisheries Centre, University of British Columbia, and making the further development of the Ecopath with Ecosim approach and software possible. The paper has benefited from discussions with and comments by Daniel Pauly.

REFERENCES

Allen, R.R. (1971) Relation between production and biomass. *J. Fish. Res. Board Can.* 28, 1573-1581.

Caddy, J.F. (1975) Spatial model for an exploited shellfish population, and its application to the Georges Bank scallop fishery. *J. Fish. Res. Board.* Canada 32, 1305-1328.

Christensen, V. (1995a) A model of trophic interactions in the North Sea in 1981, the year of the stomach. *Dana* 11, 1-28

Christensen, V. (1995b) Ecosystem maturity - towards quantification. E*cological Modelling*, 77, 3-32.

Christensen, V. (1998) Fishery-induced changes in a marine ecosystem: insight from models of the Gulf of Thailand. *J. of Fish Biology* 53 (Supplement A), 128-142.

Christensen, V. and Pauly, D. (1992) ECOPATH II - a software for balancing steady-state ecosystem models and calculating network characteristics. *Ecological Modelling* 61, 169-185.

Christensen, V. and Pauly, D. (1993) Flow characteristics of aquatic ecosystems. In: Christensen, V. and Pauly, D. (eds) Trophic models of aquatic ecosystems. ICLARM Conference Proceedings No. 26, 338-352.

Christensen, V. and Pauly, D. (1998) Changes in models of aquatic ecosystems approaching carrying capacity. *Ecological Applications*, 8: S104-S109.

Deriso, R.B. (1980) Harvesting strategies and parameter estimation for an age-structured model. *Can. J. Fish. Aquat. Sci.* 37, 268-282.

Finn, J.T. (1976) Measures of ecosystem structure and function derived from analysis of flows. *J. Theor. Biol.* 56, 363-380.

Froese, R. and Pauly, D. (2000) FishBase 2000, a biological database on fish - concepts, design and data sources. ICLARM.

Funtowicz, S.O. and Ravetz, J.R. (1990) Uncertainty and Quality in Science for Policy. Kluwer, Dortrecht, 229 pp.

Hilborn, R. and Walters, C.J. (1987) A general model for simulation of stock and fleet dynamics in spatially heterogeneous environments. *Can. J. Fish . Aquat . Sci.* 44, 1366-1369

Hilborn, R., and Walters, C.J. (1992) Quantitative fisheries stock assessment. Choice, dynamics and uncertainty. Chapman and Hall, New York, 570 pp.

Leontief, W.W. (1951) The structure of the U.S. economy, 2nd ed. Oxford University Press, New York.

Liermann, M. and Hilborn, R. (1997) Depensation in fish stocks: a hierarchic Bayesian meta-analysis. *Can. J. Fish. Aquat. Sci.* 54, 1976-1984.

Ludwig, D., Hilborn, R. and Walters, C. (1993) Uncertainty, resource exploitation, and conservation: Lessons from history. *Science* 260, 17-36.

McAllister, M.K., Pikitch, E.K., Punt, A.E. and Hilborn, R. (1994) A Bayesian approach to stock assessment and harvest decisions using the sampling/importance resampling algorithm. *Can. J. Fish. Aquat. Sci.* 51, 2673-2687

Mackay, A. (1981) The generalized inverse. Practical Computing (September), 108-110.

Magnússon, K.G. (1995) An overview of the multispecies VPA - theory and applications. *Reviews in Fish Biology and Fisheries*, 5(2), 195-212.

Martell, S. (1999) Incorporating seasonality in Ecopath. FishByte 5(6).

McAllister, M.K., Pikitch, E.K., Punt, A.E. and Hilborn, R. (1994) A Bayesian approach to stock assessment and harvest decisions using the sampling/importance resampling algorithm. *Can. J. Fish. Aquat. Sci.* 51, 2673-2687

Myers, R.A., Barrowman, N.J., Hutchings, J.A. and Rosenberg, A.A. (1995) Population dynamics of exploited fish stocks at low population levels. *Science* 269, 1106-1108.

Odum, E.P. (1969) The strategy of ecosystem development. *Science* 104, 262-270

Odum, E.P. (1971) Fundamentals of ecology. W.B.Saunders Co., Philadelphia. 574p.

Pauly, D. (1995) Anecdotes and the shifting baseline syndrome of fisheries. *TREE* 10(10), 430.

Pauly, D. (ed) (1998) Use of Ecopath with Ecosim to Evaluate Strategies for Sustainable Exploitation of Multi-Species Resources. *Fisheries Centre Research Reports*, 6(2), 49 pp.

Pauly, D. and Pitcher, T. J. (2000) Assessment and mitigation of fisheries impacts on marine ecosystems: a multidisciplinary approach for basin-scale inferences, applied to the North Atlantic. In: Pauly, D. and Pitcher, T.J. (eds) Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. *Fisheries Centre Reseach Reports* 8(2), 1-12.

Pauly, D., Christensen, V., Dalsgaard, J., Froese, R. and Torres, F. Jr. (1998b) Fishing down marine food webs. *Science* 279, 860-863.

Pauly, D., Christensen, V., Froese, R., Longhurst, A., Platt, T., Sathyendranath, S., Sherman, K. and Watson, R. (2000b) Mapping fisheries onto marine ecosystems: proposal for a consensus architecture allowing inferences at local, basin and global scales. In: Pauly, D. and Pitcher, T.J. (eds) Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. *Fisheries Centre Research Reports* 8(2), 13-22.

Pauly, D., Christensen, V. and Walters, C.J. (2000) Ecopath, Ecosim and Ecospace as tools for evaluating ecosystem impact of fisheries. *ICES J. Mar. Sci.* 57, 697-706.

Polovina, J.J. (1984) Model of a coral reef ecosystems I. The ECOPATH model and its application to French Frigate Shoals. *Coral Reefs* 3(1), 1-11.

Polovina, J.J. and Marten, G.G. (1982) A comparative study of fish yields from various tropical ecosystems. In: Pauly, D. and Murphy, G.I. (eds) Theory and management of tropical fisheries. ICLARM Conference Proceedings 9, 255-286.

Schnute, J. (1987) A general fishery model for a size-structured fish population. *Can. J. Fish. Aquat. Sci.* 44, 924-940.

Sheldon, R.W., Prahask, A. and Sutcliffe, W.H. Jr. (1972) The size distribution of particles in the ocean. *Limnology and Oceanography*, 17, 327-340

Sparre, P. (1991) Introduction to multispecies virtual population analysis. *ICES mar. Sci. Symp.* 193, 12-21.

Ulanowicz, R.E. (1986) *Growth and development: ecosystem phenomenology.* Springer Verlag, New York. 203 pp.

Ulanowicz, R.E. (1995) The part-whole relation in ecosystems. In: Patten, B.C. and. Jørgensen, S.E. (eds) *Complex ecology.* Prentice-Hall, Englewood Cliffs, New Jersey, 549-560.

Ulanowicz, R.E. and Puccia, C.J. (1990) Mixed trophic impacts in ecosystems. *Coenoses* 5, 7-16.

Walters, C.J. (1996) Suggested improvements for Ecopath modeling. In: Pauly, D. and Christensen, V. (eds) Mass-Balance Models of Northeastern Pacific Ecosystems. *Fisheries Centre Research Reports*, 4(1), 82-86.

Walters, C.J. and Kitchell, J.F. (2000) Cultivation/depensation effects on juvenile survival and recruitment: implications for the theory of fishing. *Can. J. Fish. Aquat. Sci.* 57 (in press)

Walters, C.J. and Maguire, J.J. (1996) Lessons for stock assessment from the northern cod collapse. *Reviews in Fish Biology and Fisheries*, 6,125-137.

Walters, C.J., Christensen, V. and Pauly, D. (1997) Structuring dynamic models of exploited ecosystems from trophic mass-balance assessments. *Reviews in Fish Biology and Fisheries* 7, 139-172

Walters, C.J., Kitchell, J.F., Christensen, V. and Pauly, D. (2000) Representing density dependent consequences of life history strategies in aquatic ecosystems: Ecosim II. *Ecosystems* 3, 70-83.

Watson, R., Guénette, S., Fanning, P. and Pitcher, T. (2000) The basis for change: Part 1 – reconstructing fisheries catch and effort data. In: Pauly, D. and Pitcher, T.J. (eds) Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. *Fisheries Centre Reseach Reports* 8(2), 23-39.

Winberg, G.G. (1956) Rate of metabolism and food requirements of fishes. *Transl. Fish. Res. Board Can.* 253.

RESTORATION OF OVEREXPLOITED CAPTURE FISHERY RESOURCES: AN ECONOMIC/ECOSIM MODELING APPROACH

Gordon Munro and
Ussif Rashid Sumaila

ABSTRACT

Given the current sorry state of most of the fisheries in the world's marine ecosystems, it is appropriate to develop a restoration program for the world's ecosystems. We extend modern fisheries economic theory to make it applicable to ecosystem analysis. The extended theory is then applied to explore a number of questions, including (i) to what extent is it worth restoring current ecosystems to their past states? (ii) What is the optimal approach path to the past ecosystem? Is it optimal to invest rapidly in restoring the ecosystem, or should investment proceed more slowly? Ecopath and Ecosim models form the ecological basis for our analysis, while we rely on economic valuation techniques to help us determine the economically feasible restoration plans and paths. In undertaking the valuations, emphasis will be placed on both market and non-market values.

INTRODUCTION

The FAO and the National Research Council (U.S.) maintain that sustainable world capture fishery harvests are approaching the ceilings imposed by nature (FAO, 1999; NRC, 1999). The FAO also maintains that 70 per cent of world fishery resources are either fully, or over, exploited. Overexploitation in the past means that many capture fishery resources are now producing below their full potential.

The FAO concludes that there is some scope for increasing world capture fishery harvests, but that this will, in light of the foregoing, have to come largely through improved resource management, that is to say, through correcting the resource management errors and inadequacies in the past (FAO, 1999). Thus, it is entirely appropriate that the *Sea Around Us Project* should address itself to the issue of rebuilding those fishery resources, which had been excessively depleted in the past. What the project intends to do, however, is to go beyond the question of rebuilding individual fishery resource stocks to that of rebuilding the surrounding ecosystem, as well.

We proceed as follows. We discuss first the basic theory, or analytical framework, which will underlie the research. This will be found to involve the application, and extension of, modern fishery economics. We then proceed to discuss the basis of the empirical investigation, which are provided by Ecopath and Ecosim models. An Illustrative example will be provided.

THE ANALYTICAL FRAMEWORK

The analytical framework rests upon modern fisheries economics, surveys of which can be found, among other sources, in Bjorndal and Munro (1999) and Munro and Scott (1985). In modern fisheries economics, as in other branches of natural resource economics, it is commonplace to regard natural resources as a form of capital, 'natural' capital if you will. Capital, in turn, may be defined as any asset, tangible or intangible (e.g. human skills), which is capable of yielding a stream of economic benefits to society through time. The 'economic benefits', it must be stressed, are not confined to those flowing through the market and thus having an explicit price attached to them. Non-market benefits, to which imputed values are to be attached, must be included as well.

Additions to the stocks of capital through time constitute positive investment. Investment can, of course, be negative (disinvestment), i.e. stocks of capital can be deliberately depleted.

Natural resources, certainly including fishery resources, clearly fall under the definition of capital. With respect to renewable natural resources, one can talk meaningfully about both positive and negative investment in the resources. This being the case, the theory of the optimal management of fishery resources, and other natural resources, is essentially an application of the economist's theory of capital and theory of investment. The fundamental questions addressed by these closely interlinked bodies of theory are as follows:

1. To what extent is it worth society's while to build up stock of capital? In other words, what is the optimal target stock of capital?

2. What is the optimal approach path to the target stock of capital? Is it optimal to invest (disinvest) rapidly in the capital stock, or should investment (disinvestment) proceed more slowly?

The economic theory of capital and theory of investment were originally designed with 'conventional', i.e. human made, capital, e.g. plants and machinery in mind. There is a fundamental difference between 'natural' capital and 'conventional' capital in that 'natural' capital assets come to us as an endowment from nature. Nonetheless, the economic theories of capital and investment still apply to 'natural' capital. It does mean, however, that the question of optimal negative investment, i.e. depletion, becomes relevant. Indeed, with regards to non-renewable resources, e.g. hydrocarbons, optimal management of such resources is concerned just with depletion.

A key element in the theories of capital investment is the 'rate of discount' or interest rate. We shall be concerned with the 'social rate of discount'. The social rate of discount with respect to a single investment project reflects two factors; society's rate of time preference, and the 'opportunity cost' associated with other investment projects. The rate of time preference refers to the possibility that society will give higher weight to present consumption, than it will to future consumption. The 'opportunity cost of capital', reflects the fact that the resources which society has for investment (savings, essentially) during any one time period is finite. Consequently, investment in one project may come at the expense of others. The anticipated rate of return on the forgone investment projects (expressed as a rate of return) constitutes the 'opportunity cost' of the investment project at hand. Thus, with respect to a particular project, even if one argues that, on ethical grounds, the appropriate rate of time preference should be zero, one cannot legitimately argue that the overall rate of discount applied to the project should be zero.

Anthony Scott, one of the pioneers in the field of natural resource economics, has argued vigorously that the concept of opportunity cost of capital applies to investments in 'natural' capital, as well as to 'conventional'. To pretend that this is not the case, he maintains, will have the consequence of harming, rather than assisting, future generations (Scott, 1973).

How then does all of this apply to capture fisheries? Rebuilding fishery resources, and the surrounding ecosystem, that have been depleted in the past is a program of investment in 'natural' resource capital. The questions are then:

1. What are the appropriate, or optimal, target levels for hitherto depleted resources; and

2. What are the appropriate resource investment programs? The questions imply, of course, that the extent of the depletion of the resources was 'excessive' from society's point of view.

The answers to these questions can best be illustrated by commencing with the simplest case, that of a single fishery resource. For a detailed discussion of the theory see Bjorndal and Munro (1999); Clark (1990); Munro and Scott (1985).

To simplify the exposition to follow, let x denote the biomass, let $F(x)$ denote the rate of growth of the resource, let $h(t)$ denote the harvest rate at time (t) and let δ denote the social rate of discount where $\delta \geq 0$. Finally, denote the economic benefits (non-market, as well as market) flowing from the fishery at time t as: $\pi(x,h)$.

The resource management objective can then be perceived as that of maximizing the present value of the stream of economic benefits from the resource through time. This can be expressed, putting questions of uncertainty to one side, as

$$\max PV = \int_0^\infty e^{-\delta t} \pi(x(t), h(t), dt, \qquad \text{...1)}$$

subject to various constraints, in particular those imposed by nature.

This will give rise to a decision rule for determining the target optimal biomass level, which in its simplest form, is expressed in Equation (2). The rule has come to be referred to by some textbooks as the Fundamental Rule of Renewable Resource Exploitation (e.g. Pearce and Turner, 1990).

$$F'(x) + \gamma(x) = \delta \qquad \text{...(2)}$$

Equation (2) essentially states that one should 'invest' in the resource up to the point where the yield, or rate of return, on the marginal investment in the resource (L.H.S. of Eq. (2)) is equal to the social rate of disc ount. This rate of return consists of two components, the first of which is the impact of the marginal resource investment on sustainable harvests, as indicated by $F'(x)$.

The second component, referred to in the literature as the Marginal Stock Effect (Clark, 1990), reflects benefits arising from investing in the resource, other than sustainable harvests. In the model, which originally gave rise to the equation (Clark and Munro, 1975), the Marginal Stock Effect reflected the fact that in many fisheries the size of the biomass will have an impact on harvesting costs. The larger the size of the biomass, the lower will be harvesting costs, and the greater (other things being equal) will be economic benefits from the fishery based upon the resource.

The Marginal Stock Effect can, however, be easily adapted to capture other non-sustainable harvest benefits. Suppose, for example, that the members of society enjoy psychic benefits from knowing that the resource exists in a large and healthy state. The Marginal Stock Effect can be employed to capture these, 'existence' benefits, at the margin.

The effect of the Marginal Stock Effect is straightforward and obvious. The larger is $\gamma(x)$, the greater will be the incentive to invest in the resource.

As economists have stressed for almost 50 years, capture fisheries suffer from the fact that the resources providing the basis of the fisheries are 'common pool' resources. The consequence of this 'common pool characteristic of capture fisheries is that the fishers are given a powerful incentive to discount heavily future returns from the fishery. The fishers will act as if the appropriate rate of discount is $\rho. >>> \delta$. Indeed, it can be argued that ρ may approach ∞ (Clark, 1990). Be that as it may, if fishers, subject to these perverse incentives, are permitted to operate free of controls, then overexploitation of the resources – excessive disinvestment of natural capital - from society's perspective is all but guaranteed. In many capture fisheries, fishers, subject to the aforementioned perverse incentives have, in fact, been able to exploit the resources subject to inadequate controls (Christy 1977). This, the OECD argues, has been the fundamental cause of the overexploitation reported by the FAO (OECD, 1997).

In any event, there exists a powerful case to be made on economic, as well as biological, grounds that, from society's point of view, the extensive rebuilding of many of the world's capture fisheries is eminently desirable.

The existing economic theory also allows one to address the question of optimal investment in the resource. In the first instance, it allows us, and indeed forces us, to recognize that no investment comes without a cost. As in any investment program, one incurs costs and sacrifices today, in the hope of higher returns in the future.

The most rapid way in which to 'invest' in a fishery resource, subject to excessive depletion in the past, is to declare an outright harvest moratorium until the target biomass is achieved. Such a policy is feasible in certain fisheries. The Norwegian Spring Spawning (Atlanto–Scandian) herring fishery of the North Atlantic provides a case in point. The resource, which crashed in the late 1960s – early 1970s, was subject to a harvest moratorium, which lasted for over twenty years. The resource has now been rebuilt, i.e., the resource investment policy has been eminently successful (Munro, 1999).

In other fisheries, however, such a 'rapid investment' policy would wreak havoc on the fishing industry, and communities dependent upon the resource. Determining the optimal resource investment program (which in terms of practical policy would be referred to as 'the adjustment phase') in such cases can prove to be exceedingly complex. One is faced with an interaction between 'natural' capital, in the form of the resource, 'conventional' capital in the form of the fleet and processing capacity, and 'human' capital in the form of skilled fishers and processing plant workers (see Clark, Clarke and Munro, 1979; Gréboval and Munro, 1999).

Next, a comment about uncertainty is in order. In taking a capital theoretic approach to the economics of fisheries management, the question of uncertainty becomes inescapable. Modern fisheries economics does address the issue, both in terms of theory of optimal resource management, and in terms of fisheries management policy (Arnason, 1990; Bjorndal and Munro, 1999).

The discussion, to this point, has been in terms of a single resource. When we turn to consider an ecosystem approach, it can be said that, at a minimum, an ecosystem approach recognizes species interaction, with the implication that species should not be managed on an individual basis. In terms of the economics, management of a single resource can be thought of as a single 'natural' capital asset. When one moves beyond a single species analysis to examine multi-species, and the surrounding ecosystem, one moves from the management of a single natural resource asset to the management of a set, or portfolio, of natural resource assets. The focus is then on economic returns from, and investment in, the portfolio as a whole, rather than on the return from and investment in individual natural resource assets.

While there has been some work done on extending the economic analysis, both theoretical and empirical, beyond single species (see, for example: Arnason, 1998; Bjorndal and Munro, 1999; Flaaten, 1988; Munro and Scott; Sumaila, 1997), the work has been largely confined to simple predator-prey models. Even with these simple models, it has been difficult to deal rigorously with the issue of optimal resource investment programs.

It is the underlying biological models, which have held the economists back. It is hoped and expected that, in this project, Ecopath/Ecosim models will enable us to move well beyond the limitations of simple multi-species models, and to develop economic models of fisheries, both theoretical and empirical, which will reflect a true ecosystem approach to fisheries management.

With reference to the analytical framework, a commentary on resource management policy is required. The thrust of much policy in fisheries is directed towards overcoming the consequences of the 'common pool' characteristics of capture fisheries. This project demands that these policy issues be investigated. The relevance of these policy issues to the question of investment in capture fishery resources, and the surrounding ecosystem, is straightforward and stark. If the policy issues are not effectively addressed, then after the resources (and ecosystem) are rebuilt, the cycle of overexploitation will commence all over again. The sustainable return on the resource/ecosystem investment will be zero, or close to zero. The resource/ecosystem investment program will prove to have been an exercise in futility.

We turn now to the empirical component of this sub-project, which, as we have indicated, must be seen as its heart. In so doing, we shall only briefly discuss Ecopath and Ecosim models (Pitcher, 1998). The reader is strongly encouraged to examine, as well, other papers in this volume, which explore Ecopath and Ecosim models in considerable depth (see, in particular, Christensen and Walters, 2000).

THE EMPIRICAL ANALYSIS

This work applies the class of ecosystem models described in Christensen and Walters (2000), that is, the mass-balance trophic models known as ECOPATH and ECOSIM[1]. The quantitative results derived from these models are combined with economic valuation techniques based on our earlier theoretical discussion to assess the potential payoffs to be achieved in investing in rebuilding marine ecosystems. Ecopath, which is static, is used to capture the current state of the ecosystem, and the state of the system as it might have been sometime in the past (Pitcher, et al., 1999 and Pauly, Pitcher and Preikshot, 1998). Next, Ecosim, the dynamic version of Ecopath, is used to construct restoration models of the ecosystem to be analyzed. Since Ecosim is a dynamic model, it can tell us how long we need to wait in order to rebuild the ecosystem, given different restoration programs. The modeling exercise here will produce results such as (i) if catch rates are set equal to zero, it will take x years for the ecosystem to be restored to the state in which it was z years ago, (ii) if the catch rate is reduced to w per cent of its current rate, it will take y years to return to the ecosystem state v years ago. Clearly, such results are extremely important for us to undertake a full economic analysis of any restoration program. Different rebuilding plans are then assessed economically

[1] This modeling framework is chosen because it has a number of advantages over other existing ecosystem or multispecies modeling approaches, such as the Multispecies VPA (Sparre, 1991), the dynamic models of (Larkin and Gazey, 1982), and bio-energetic models. First, it includes all trophic levels in the analysis (from primary producers to top predators) as opposed to focusing only on the commercially important fish species. Second, its emphasis on ecological relationships makes it intuitively simple and transparent, without requiring a high degree of expertise from the modeler. Finally, as a dynamic version of the Ecopath mass-balance models, Ecosim is capable of answering what-if questions about policy and ecosystem changes that would cause shifts in the balance of trophic interactions (Walters et al., 1997).

to determine which plan gives us the most return for our investments.

Sumaila (1998) identifies two ways to go about evaluating benefits from ecosystems - the so-called *basic* and a*dvanced* approaches. The basic approach introduces economics into the Ecopath/Ecosim framework by taking the biological results generated by Ecopath/Ecosim under different scenarios, and applying appropriately determined prices for fish landed, non-market values of other ecosystem goods and services, the cost of exploiting the fish and the discount rate. On the other hand, the advanced approach seeks to incorporate the regulatory body's and/or fishers' behavior and motivations into the framework. This is done by defining and incorporating the objective functions, and decision variables of the participants into the analysis. Finally, this approach involves the optimization of the objective functions of the participants subject to the relevant constraints. For the purposes of the planned work in this paper, we apply the basic approach to evaluate the potential payoffs under different restoration scenarios. This involves undertaking cost-benefit-type analysis (see Angelsen and Sumaila, 1997) based on the ecological results generated by Ecopath/Ecosim, under different restoration scenarios. In this way, we are able to compute the net discounted economic benefits that can be achieved under the different scenarios, which in turn allows us to determine the rebuilding scenario that produces the best ecologically sustainable economic outcome.

The economic valuation explicitly takes into account both market (or commercial) and non-market (or non-commercial) benefits. The latter benefits are meant to incorporate in the valuation the fact that ecosystems and the resources they contain have value in 'themselves' above those bestowed on them due to their commercial value. A simple interpretation of this is that the remaining ecosystem and its resources, after we have taken the commercial catch, has both non-commercial and intrinsic values[2].

The literature on the valuation of ecosystem resources and services give wide, often controversial estimates of the value of these resources and services (see for instance, Costanza, 1997). Instead of attempting to impute a specific non-market value to ecosystem resources and services, we carry out a number of analyses assuming different values, usually taken from the literature, for the remaining biomass of all species of creatures in the ecosystem. In this way, we will be able to identify the cut off points at which one ecosystem alternative ceases to be optimal (that is, produces the best overall benefits) and the other becomes optimal.

An Example based on the Benguela Ecosystem of Namibia

For purely illustrative purposes we used an Ecopath with Ecosim model of the Benguela ecosystem off Namibia. The model was originally developed to analyze the ecological and economic impacts of the activities of distant water fleets (DWFs) in this ecosystem (see Sumaila and Vasconcelos, 2000). The results reported (in the cited paper) for the 'with' and 'without' DWFs scenarios are adopted for the purposes of this paper. Here, we define the 'with' DWF as the 'current' ecosystem and the 'without' DWF, the 'desired' ecosystem.

Results

Ecological results

The biomass (in million tonnes) of the main species in the current and desired ecosystems are given in Table 1. For the detail assumptions underlying the model from which these numbers are obtained see Sumaila and Vasconcellos (2000), and Jarre-Teichman and Christensen (1998). The table shows, among other things, that the total biomass of all species under the desired ecosystem alternative is 19% higher than in the current ecosystem. The difference between the two ecosystems is much larger if one compares the biomass of the three most important commercial species, namely, hake, sardine and horse mackerel. The difference between the desired and the current ecosystem are 51%, 68% and 61%, respectively, for hake, sardine and horse mackerel. Ecosim accounts for the indirect trophic effects of release from predation, and increase in competition for resources in the food web following fishing impact. Therefore, the balance of masses in the food web cause an increase in the biomass of preys and competitors (e.g. anchovy, mackerel, other pelagic and small demersal fish) with the depletion of hake, horse mackerel and sardine. The converse is predicted with the reduction in fishing pressure on these

[2] Non-commercial is used interchangeable with non-market. Intrinsic value of a natural resource has to do with the value of the natural resource 'in and of themselves'.

Table 1. Biomass and catches (in tonnes) from the hypothetical North Atlantic ecosystem

Species	Biomass in million t Desired ecosystem	Current ecosystem
1 Anchovy	0.380	0.703
2 Sardine	2.409	1.646
3 Mackerel	0.039	0.107
4 Horse mackerel	1.319	0.806
5 Snoek & Tuna	0.006	0.008
6 Other pelagics	0.439	1.242
7 Hake	2.094	1.077
8 Other demersals	0.191	0.210
Total	**6.876**	**5.80**
Ratio of total desired/total		**1.19**

three species in the "without" DWF (desired ecosystem) scenario. In fact, after the heavy exploitation by DWF during the 1960s and 1970s, the sardine fishery off Namibia declined and was partially replaced by a fishery for anchovy, which is now one of the most abundant pelagic species in the ecosystem.

The crucial question that the newest version of Ecosim is to help us answer, taking all the potential uncertainties into consideration is, how long will it take to restore the Benguela ecosystem from its current to the desired state? In the absence of real simulation results, let's assume, for illustrative purposes, that Ecosim predicts that if a harvest rate of 5% is exerted on the current biomass of hake, horse mackerel and sardines, and 15% exerted on the biomass of all the other species for the next 10 years, the current ecosystem will have an excellent chance of returning to the desired ecosystem state. Also, if the status quo is maintained, that is, if the same amount of fishing effort as employed by both the DWFs and the domestic Namibian fleet in the 'with' DWF (or current ecosystem alternatives) continue to be exerted, there will be a reduction in biomass (and hence harvest) of hake, sardine and mackerel to a third of their current quantities 5 years from now. On the other hand, after the ecosystem has been returned to the desired state, Ecosim shows that a harvest rate of 15% of the biomass of all the species in the ecosystem is sustainable through time.

Economic results

The above ecological results are valued economically to determine what a restoration program to return the ecosystem to what we call the desired state means economically. To carry out the economic valuation, we first undertake a pure market or commercial valuation. That is, we determine the discounted profit to be obtained from the commercial catches from the ecosystem in a time horizon of 30 years[3].

Figures 1 and 2 depict the profile of commercial and overall (that is, commercial and non-commercial) benefits. We see a similar picture of the profile of discounted benefits, both market and non-market in the two figures.

Next, we attempt a more complete analysis by incorporating non-market values. In this case we add to the discounted profit computed, an estimate of the non-market values associated with the remaining biomass of fish in the ecosystem. In general, it is estimated that the annual shadow value of biomass is between 30-70% of the market price (Arnason, pers. comm.). Based on this, we impute a unit value equivalent of 30% of the price of a given species to the standing biomass.

Initially, more benefits accrue under the current ecosystem alternative but this quickly changes a few years later when the three key commercial species get depleted, with the desired ecosystem taking over to produce more discounted benefits. The gap between the benefits from the desired and the current ecosystem increases in favor of the desired ecosystem in figure 2. That is, when non-market benefits are incorporated. This is as expected since overall, more biomass is left in the desired than in the current ecosystem. It should be noted that the sharp drops and increases we see in the diagrams are due to the assumption we made for rebuilding and depletion under the two ecosystem scenarios. These will likely be different when actual simulations are carried out.

Table 2 presents a more compact summary of the economic results. It displays both the total commercial and overall benefits from the two

[3] A realistic discount rate of 3% is used in the calculations.

ecosystems, and the percentage gains from the restoration effort. A total commercial benefit of N$12,175 million is obtainable over the 30 year time horizon from the desired ecosystem, as against N$ 8,957 for the current ecosystem. The figures in the case of the overall benefits are N$ 36,345 and N$20,106 from the desired and current ecosystems, respectively.

Table 2. Economic gains from restoration effort

Ecosystem alternative	Commercial	Overall
Desired	12175	36345
Current	8957	20106
Gain (%)	36	81

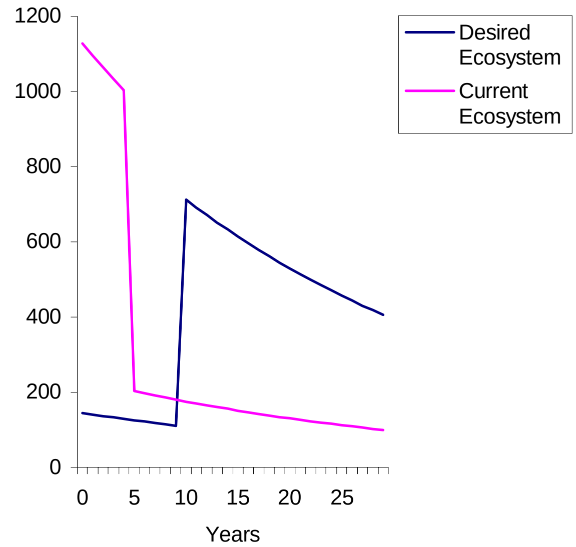


Figure 2. Profile of overall benefits in million Namibian $

CONCLUDING REMARKS

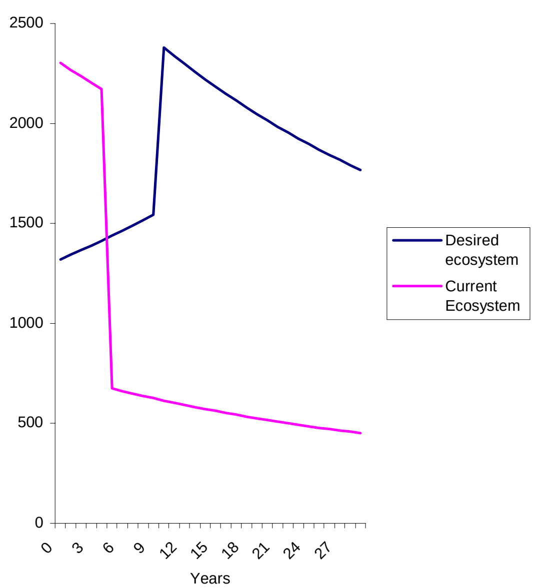


Figure 1. Profile of commercial benefits in million N$

We have presented the analytical framework, the empirical methodology, and an example to show the approach we will be applying to help us assess and evaluate the potential benefits from investing in the restoration of the marine ecosystems of the North Atlantic. In the coming months, we will construct ECOPATH and ECOSIM models of the major ecosystems of the North Atlantic; collect relevant costs and price data; and use the methodology outlined herein to explore restoration scenarios for the marine ecosystems in the North Atlantic region. Our example, which provides complementary analyses to those by Ruttan et al. (2000), gives an indication that there are potentially significant gains to be obtained from restoring ecosystems to their past states, the extent to which this is true of the North Atlantic will be demonstrated by this component of the *Sea Around Us Project.*

ACKNOWLEDGMENTS

The *Sea Around Us Project* is supported by the Environment Program of the Pew Charitable Trusts. We wish to thank all members of the SAUP, participants at the SAUP methodology workshop and three reviewers of the paper for helpful comments.

REFERENCES

Angelsen A. and Sumaila, U. R. (1997) Hard methods for soft policies: Environmental and social cost-cost benefit analysis. In: F. A. Wilson (ed) Towards sustainable project development. Edward Elgar, Cheltenham, 20-42.

Arnason, R. (1990) Minimum Information Management in Fisheries. *Canadian Journal of Economics*, 23, 630-653.

Arnason, R. (1998) Ecological Fisheries Management Using Individual Transferable Quotas, Ecological Applications, 5151-5159.

Bjorndal, T. and Munro, G.R (1999) The Economics of Fisheries Management: A Review. In: Tietenberg, T. and Folmer, M. (eds) The International Yearbook of Environmental and Resource Economics 1998/1999. Cheltenham, Edward Elgar, 153-188.

Christensen, V. and Walters, C. (2000) ECOPATH with ECOSIM: Working capabilities and limitations. In: Pauly, D. and Pitcher, T.J. (eds 2000) Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. *Fisheries Centre Research Reports* 8(2), 79-105.

Christy, F. (1997) Economic waste in fisheries: Impediments to change and conditions for improvement. In: Pikitch, E.L., Huppert, D.D. and Sissenwine, M.P. (eds) Global trends: Fisheries management. American Fisheries Society Symposium 20, Bethesda, Maryland, 28 – 39.

Clark, C. W. (1990) Mathematical Bioeconomics: The Optimal Management of Renewable Resources, 2nd edn. New York, Wiley-Interscience.

Clark, C. W. and Munro, G.R. (1975) The Economics of Fishing and Modern Capital Theory: A Simplified Approach. *Journal of Environmental Economics and Management*, 2, 92 - 106.

Clark, C. W., Clarke, F.H. and Munro, G. (1979) "The Optimal Management of Renewable Resource Stocks: Problems, of Irreversible Investment," *Econometri*ca, 47, 25-47.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P. and van den Belt, M. (1997) The value of the world's ecosystem services and natural capital. *Nature* 387, 253-260.

Flaaten, O. (1988) The Economics of Multispecies Harvesting: Theory and Applications to the Barents Sea Fisheries, Berlin, Springer-Verlag.

FAO (1999) The State of World Fisheries and Aquaculture, Rome, Italy.

FAO (1996) Yearbook of Fishery Statistics. Commodities, 83, 179 pp., Rome, Italy.

FAO (1954-55) Yearbook of Fishery Statistics. Production and craft, 5, pp. d-1 to d-30.

Gréboval, D. and Munro, G.R. (1999) Overcapitalization and Excess Capacity in World Fisheries: Underlying Economics and Methods of Control. In: Managing Fishing Capacity: Selected papers on Underlying Concepts and Issues, Food and Agriculture Organization of the U.N. FAO Fisheries Technical Paper No. 386, Rome, 1-48.

Jarre-Teichmann, A. and Christensen V.(1998) Comparative modelling of trophic flows in four large upwelling ecosystems: global vs. local effects. p. 423-443 in Durand, M.H., Cury P., Mendelssohn R., Roy C., Bakun A. and Pauly D. (eds.) Global vs. Local Changes in Upwelling Ecosystems. Proc. of the first CEOS symposium, 5-9 September 1994, Monterey, CA. Orstom, Paris

Larkin, P. A. and Gazey, W. (1982) Application of ecological simulation models to management of tropical multi-species fisheries. In: Pauly, D. and Murphy, G.I. (eds), Theory and Management of Tropical Fisheries, ICLARM Conference Proceedings, Manila, Philippines, 123-140.

Munro, G.R. (1999) The Management of High Seas Fisheries and the United Nations Conference on Straddling Fish Stocks and Highly Migratory Fish Stocks: A Second Review. Centre for Fisheries Economics, Norwegian School of Economics and Business Administration, Working Paper No. 36/1999.

Munro, G.R. and Scott, A.D. (1985) The Economics of Fisheries Management. In: A.V. Kneese and J.L. Sweeney (eds) Handbook in Natural Resource and Energy Economics, Vol. 2. Amsterdam, North-Holland, 623-676.

National Research Council (1999) Sustaining Marine Fisheries, Washington, National Academy Press. 164 pp.

OECD (1997) Towards Sustainable Fisheries: Economic Aspects of the Management of Living Marine Resources, Paris, 447-465.

Pauly, D., Pitcher, T.J. and Preikshot, D. (eds) Back to the Future: Reconstructing the Strait of Georgia ecosystem. University of British Columbia, Fisheries Centre Research Report No. 6(5), 100 pp.

Pearce, D.W. and Turner, R.K. (1990) Economics of Natural Resources and the Environment, Baltimore, The Johns Hopkins University Press.

Pitcher, T.J., Pauly, D., Haggan, N. and Preikshot, D. (1999) 'Back to the Future': a method employing ecosystem modelling to maximize the sustainable benefits from fisheries. In: Ecosystem Considerations in Fisheries Management. Alaska Sea Grant Anchorage, USA, pp. 447-465.

Pitcher, T.J. (1998) Back to the Future: A novel methodology and policy goals in fisheries. In: Pauly, D., Pitcher, T.J. and Preikshot, D. (eds) Back to the Future: Reconstructing the Strait of Georgia ecosystem. University of British Columbia, Fisheries Centre Research Report 6(5), 4 – 7.

Ruttan, L.M., Gayanilo, F.C. Jr. and Pauly, D. (2000) Small versus large scale fisheries: A multispecies, multifleet model for evaluating their potential to conserve fish and fisheries. In: Pauly, D. and Pitcher, T.J. (eds) Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. *Fisheries Centre Research Reports* 8(2), pp. 64-78.

Scott, A. (1973) Natural Resources: The Economics of Conservation, Toronto.

Sparre, P. (1991) Introduction to multispecies virtual population analysis. *ICES Mar. Sci. Symp.* 193, 12-21.

Sumaila, U.R. (1998) Bioeconomics and the ECOPATH/ECOSIM Framework. In: Pauly, D. (ed) Use of ECOPATH with ECOSIM to Evaluate Strategies for Sustainable Exploitation of Multi-Species Resources. Proceedings of a FAO/FC Workshop, Vancouver, Canada. *Fisheries Centre Research Reports*, Vol. 6 No 2, 19-20.

Sumaila, U.R. (1997) Strategic dynamic interaction: The case of Barents Sea fisheries. *Marine Resource Economics*, 12, 77-94.

Walters, C., Christensen, V. and Pauly, D. (1997) Structuring dynamic models from trophic mass-balance assessment. *Reviews in Fish Biology and Fisheries*, 7(2), 139 - 172.

TRACKING FISHERIES LANDINGS IN THE NORTH ATLANTIC*

Ussif Rashid Sumaila, Ratana Chuenpagdee and Gordon R. Munro

ABSTRACT

The aim of this paper is to develop a spreadsheet-based framework to help track the flow of fish landings within the North Atlantic region, and to identify profit margins to the various product sectors. We start by estimating the amount of domestic consumption for each country, which is the total fish landings, plus the amount of import, minus the amount of export. As we focus mainly on the trade in North Atlantic, we limit the import and export amounts to those within the region. Next, we determine the proportion of the major product forms under which they are marketed, e.g. fresh, frozen, salted, smoked, industrial, etc. For each product, we follow its marketing channel through various distributing sectors, i.e. wholesalers, retailers and restaurants and food services. In all cases, both the quantity and the value of traded products are recorded, as well as the operating costs of each sector. The results derived are used to identify the sectors or product forms that capture most of the economic benefits from the fishes of the North Atlantic. This knowledge, when used with other information and coupled with further investigation on ownership patterns in the fishing sector, will contribute to the development of management policies that are both ecologically and socio-economically viable.

INTRODUCTION

The objective is to develop an analytical system that will help us track the flow of fish landings within the North Atlantic region (NA). We therefore provide a diagram illustrating the flow of fish landings into their various product forms that go through different distributing sectors, such as wholesalers, retailers, restaurants and other food services. The process involves compiling information about landings, products, processing, trading, marketing and consumption of the major fish species of each NA country. This is based largely on the existing databases (international, regional and national), publications and reports. In addition to the quantities and values of each product, cost data are obtained. An Excel spreadsheet is used as a compilation tool to illustrate the marketing channels of North Atlantic fishery products. Ultimately, we hope to provide an insight into the contribution of each marketing sector to the whole NA fishery, and to highlight the role of each sector in the distribution of food products derived from fishery resources. This information could be used in conjunction with other knowledge to assist policy makers in the development of policies leading to sustainable and socio-economically viable management of the fishery resources of the North Atlantic.

Similar to Roy et al (1994) in the analysis of the U.S. market for Canadian Atlantic cod, we focus on the supply side of the fish products. Acknowledging the difficulty in obtaining the required data, as most fish species are reported as aggregated fish product, we do not attempt to use a sophisticated model to analyse the market structure of the fish trade. Rather, we aim to make use of available data and systematically follow each fish product, both fresh at landings and processed, through its distributing channel. Conceptually, the steps in the analysis include:

1. Starting with the total fish landings[1] from the waters of each major fishing nation within the North Atlantic region, we track how these landings flow into the different product forms under which they are marketed. That is, fresh, frozen, salted, smoked, industrial, etc. Appropriate conversion factors must be used to take into consideration the recoveries and yields of raw materials (see for example, Crapo et al. 1993).
2. Determine what portion of the product forms is consumed in the domestic market in each relevant country versus the export market.
3. Find out what portion of the product forms are exported to countries within and outside the North Atlantic region.
4. In the case of fish exported to countries outside the North Atlantic region, nothing more happens to them in our framework – they are assumed to flow into a “sink”. In similar fashion, imports from non-NA countries are ignored. This is important because the focus in this exercise is what happens to fish caught within North Atlantic waters, and how the catch may impact the sustainable use of these natural resources.[2]

[1] These amounts must be adjusted for the unreported catch based on Watson et al. (2000) to further investigate the discrepancy between amount traded and the amount consumed, if any.

[2] We acknowledge, of course, that by doing so, it might seem as if we ignore the actual trade pattern of fishery product around the world. The problem of the sink and the source will disappear once the analysis extends to the global level.

5. In the case of fish exported to countries within the NA, the amount exported is further split into the various countries to which there are exported to. It should be noted that the exports from country A to country B are also the imports to country B from country A.

6. The total portion of the landings used domestically is then added to all the imports from other countries within the NA to get the total fish from North Atlantic waters that are actually utilized in the particular country under consideration.

7. The landing in (6) above is then split into how much goes to (i) the restaurant/food services sector (direct to consumers), (ii) the processors and distributors' sector (both for human consumption and industrial use), and (iii) to retailers (to consumers-fresh seafood).

8. Prices per unit of fish for the different products are collected, and applied to the quantity of products marketed by a given sector to obtain the revenues accruing to each of the sectors mentioned above.

9. Cost of landing fish are collected and combined with the prices of the products marketed by the different sectors to calculate the average margin[3] received by each sector.

10. The information under points (8) and (9) are combined to determine the profitability of the different product sectors. This then helps us to isolate the sector(s) that captures most of the benefits from the fishes of the North Atlantic.

Construction of the Market Anatomy for North Atlantic

The construction of the market anatomy of fishery products within the NA relies almost entirely on secondary source of information, both published and unpublished data. Initial inputs, such as landings, values, imports, exports, etc., come from existing global and regional databases, such as FAO's GLOBEFISH, FAOSTAT, OECD Statistics, and the European Commission Fisheries Statistical Bulletin (see short descriptions in Box 1). The next set of data comes from national statistical agencies, some of which can be accessed via their website (see example in Box 1), and some are available as printed reports. Countries where substantial amount of information is available on-line include U.S.A., Canada, Iceland, Norway and to a lesser extent, Portugal. For the U.S. market, in particular, annual report on the seafood trade is published (see for example, Seafood Market Analyst 1997). The last source of information, concerning the development and the structure of the markets and other information relating to the marketing of the fishery products, is obtained through agency reports and publications. Additionally, it might be necessary to contact some of the major distributors and processors in each country to gain further information about their enterprises, particularly about the distribution of their products and the costs related to their operation.

[3] The margin is the difference between the price received by a sector and the price they paid per unit plus the cost of handling and processing by the sector.

BOX 1: EXAMPLES OF DATABASES AND INFORMATION ON TRADE AND MARKETING OF FISHERY PRODUCTS

FAO - The Food and Agriculture Organization of the United Nations (FAO) maintain several fishery statistical databases, two of which are used in the study, i.e., FAOSTAT and GLOBEFISH.

> *FAOSTAT* is an on-line and multilingual databases currently containing over 1 million time-series records covering international statistics in several areas such as production, trade, food balance sheets, fishery products, forestry products, etc. Specific information about fishery products, both primary and processed, can be obtained from FAOSTAT Fisheries Data. The database is available free of charge via the web (http://apps.fao.org) or in CD-ROM.
>
> *GLOBEFISH* is an integral part of the Fish INFOnetwork set up by FAO to provide regional marketing information. The core of GLOBEFISH is the databank containing fish price information, international trade statistics, catch and production data as well as news items of relevance to fisheries and fish trade. Information is available as publications or on-line service to subscribers.

OECD Statistics: OECD (Organisation for Economic Co-operation and Development) provides economic statistics on food, agriculture and fisheries, via their web page (http://www.oecd.org/statlist). Information on international trade in goods and services and foreign trade by commodity are available. (Note: no database is available via internet, some are free documents, but mostly one needs to subscribe or buy)

The European Commission - The European Commission web site contains important information on fisheries, under statistical bulletin (http://europa.eu.int/comm/dg14/bull/enbull.htm). The last update in March 1997 provides tables summarising fisheries data, including landings, external trade, processing industry, consumption, and markets. Most of these data come within the EU, such as Eurostat-Comext, and some data are obtained from FAO.

Fisheries and Oceans Canada (DFO) provides statistical services on a wide range of fisheries data including Canadian landings and Canada's international trade quarterly, fisheries products and stocks on an ad hoc basis, the Annual Statistical Review of Canadian Fisheries, Recreational Fisheries in Canada based on a five-year survey cycle and also provides, on request, customised reports covering currently available data. Information on imports, exports and trades are also provided. The web address for this database is http://www.ncr.dfo.ca/communic/statistics/.

The US National Marine Fisheries Services (NMFS) provides fisheries statistics and economics data via their web page (http://www.st.nmfs.gov). Extensive information is available on imports, exports and marketing of the fishery products. A review of processed fishery products is also available.

North Atlantic Solutions (NAS) - The NAS project is an umbrella organisation for Icelandic fish companies who export their products and services. The project is run by the Trade Council of Iceland. (http:/hubble.mmedia.is/intranet/nas/vefsidur.nsf/index/1?open). This web page provides links to The Icelandic Ministry of Fisheries, which also contains useful information on the disposition of catches and processing.

Statistics Norway (SN) is a central institution producing official statistics for Norway. Information provided in the yearbook includes catch by species (quantity and value) imports-exports of principal commodities, and operating results of fishing vessels. (http://www.ssb.no/www-open/english/yearbook)

Market channel

A spreadsheet is developed to systematically consolidate the above information to provide an overall picture of the market channel of fishery products in the North Atlantic (Figure 1). In general, we start with landings of a given species (e.g. cod, herring, etc.) in a given country in the region and follow the processing and distributing channels that it goes through, in various product forms, e.g. fresh, frozen, dried, etc., before it reaches consumers. Both the amount in tonne and the value in US $ are recorded. For each product form, including fresh fish, the total amount distributed in the market is the amount landed plus the imported amount from all NA countries under consideration. The amount of other products, though not available in the landing record, is approximated using either the proportion of domestic consumption or the proportion of imports of these products, or assumed if not known. The main product forms varies from species but generally include fresh, frozen (block, fillet), dried/salted, canned and fish meal. Again, appropriate conversion factor is used to proportion the amount landed as raw materials into the finished products.

The distributing sectors considered in the analysis include processors, wholesalers, retailers (e.g. fish and seafood stores, fish mongers, etc.), supermarkets and grocers, and restaurant and other food services. It should be noted that not all products go through all sectors in the channel. As well, we do not imply that the distributors at the end of the chain have no direct access to the products. On the contrary, some of the distributors in the middle of the channel might not play an important role in certain fisheries. Retailers, for example, could serve as intermediate between wholesalers and consumers at the end of the channel. In the case of big supermarkets buying directly from wholesalers or processors, however, retailers might not appear at all in the marketing chain. Consumers and exporters sit at the end of the channel as the final destination of the products. Only the amount exported to NA countries are included in the study and re-entered as imports. The exclusion of fish trade that involves countries other than those in NA may pose some problems in the analysis of the fish market, as a good proportion of fish consumed within the NA countries is imported from other region. The framework developed here can easily be applied to track the fisheries landings around the world, as we plan to do.

The cascading effect of the fishery products occurs at all levels of distribution. For example, fresh fish can be sold to processors, wholesalers, retailers, supermarket, restaurants, and in some cases, directly to consumers, and to exporters. The amount sold to processors and the imported amount to processing sector gets redistributed as other products to wholesalers, retailers, etc. For any one processed product, the amount wholesalers sell to the market is thus equivalent to the amount processors sell to wholesalers. However, the amount retailers sell to those that follow in the marketing channel include both the amount bought from processors and the amount bought from wholesalers. It is also assumed that supermarkets distribute their products to both restaurants and consumers, while the only outlet for restaurants is consumer. Exporters are assumed to receive the products either directly from fishers (if fresh) or from processors. Consumers, on the other hand, buy most of the products from fish and seafood stores, supermarkets and restaurants.

A similar approach is taken to incorporate values of fishery products in the analysis (Figure 2). Data mostly available include landing values, import and export prices. The prices of fishery products related to other sectors in the market channel are inferred from existing databases, national fishery statistic and country reports, publications, and personal communication. The analysis of the value of the fishery products allow us to investigate further into the importance of each marketing sector to the fishing industry in the North Atlantic and its varying degree of socio-economic impacts. When prices are used in the model, selling prices represent unit prices in all cases. For instance, unit price of fresh fish at landing is the ex-vessel price. The unit price of frozen fish from processing plants sold to wholesale, retail and other distributors are selling prices set by processors. As shown in Figure 2, the price of imports and exports vary depending on the origins and the destinations of the products.

The available data, particularly from existing databases and literature, provide a basis for the construction of the market channel. Nevertheless, many pieces are still missing and several assumptions must be made in order to obtain the complete structure. The model presented in this paper provides the basic framework to analyse the flow of fishery products in NA and could be easily fine-tuned once information becomes more available.

DETERMINING ECONOMIC BENEFITS TO MARKETING SECTORS

Two approaches will be employed in a complementary manner to help us calculate the gross economic benefit to each sector. First, we use the quantity and the price information of fishery products distributed through the various market sectors. This gross benefit is then split into three main components: (i) the total cost of acquiring the

Species - Atlantic Cod
Country - Portugal
Product form - Fresh (from landing to various sectors)

		FP		FW		FR		FS		FFS		FC		FX		
Country of origin*	Quantity	Processors		Wholesalers		Retailers		Supermarket		Rest/Food services		Consumers		Exporters		Destination
	(t)	prop.	(t)	prop.	(t)	prop.	(t)	prop.	(t)	prop.	(t)	prop.	(t)	prop.	(t)	
Total am. Traded	C = (A+B)	0.75	0.75 C	0.10	0.10 C	0.08	0.08 C	0.05	0.05 C	0.01	0.01 C	0.01	0.01 C	0.005	D	NA countries
Domestic landings	A															Canada
Imports from NA	B			Wholesale distribution		WR		WS		WFS		WC		WX		Denmark
Canada					(t)	prop.	(t)	prop.	(t)	prop.	(t)	prop.	(t)	prop.	(t)	Faeroe Is.
Denmark					E = 0.10C	0.40	0.40 E	0.30	0.30 E	0.20	0.20 E	0.100	0.10 E	0.00	0.00 E	France
Faeroe Is.																Germany
France					Retail distribution			RS		RFS		RC		RX		Ghana
Germany							(t)	prop.	(t)	prop.	(t)	prop.	(t)	prop.	(t)	Iceland
Ghana							F = 0.08 C + 0.40 E	0.35	0.35 F	0.35	0.35 F	0.300	0.30 F	0.00	0.00 F	Morocco
Iceland																Netherlands
Morocco								Supermarket distribution		SFS		SC		SX		Norway
Netherlands									(t)	prop.	(t)	prop.	(t)	prop.	(t)	Poland
Norway									G = 0.05 C+ 0.30 E+ 0.35 F	0.20	0.20 G	0.800	0.80 G	0.00	0.00 G	Portugal
Poland																Russian Fed.
Portugal										Food service distribution		FSC		FSX		Spain
Russian Fed.											(t)	prop.	(t)	prop.	(t)	Sweden
Spain											H = 0.01C+0.20 E+ 0.35 F+ 0.20G	1.000	1.00 H	0.00	0.00 H	UK
Sweden																USA
UK																Other EU
USA																Others
Oher EU																TOTAL
Others																
Total imports																

Legend for headings:
1) F = Fisher, P = Processor, W = Wholesaler, R = Retailer, S = Supermarket/Grocers, FS = Restaurant/Food Service, C = Consumer, X = Exporter
2) Thus, FP = Amount sold from Fisher to Processor, FW = Amount sold from Fisher to Wholesaler, etc.

Figure 1 Distribution of fishery products through various marketing sectors (hypothetical data)

\tlantic Cod
'ortugal
m - Fresh (from landing to various sectors)

						FW			FR			
rigin*	Quantity	Unit price	Value		Unit price	Wholesalers		Value	Retailers	Unit price	Value	*(continued*
	(t)	($/t)	($)	(t)	($/t)	prop.	(t)	($)	prop.	($/t)	($)	
raded	C = (A+B)		TV = VD + VI	0.75 C	FP	0.10	0.10 C	FW * 0.1C	0.08	FR	FR * 0.08C	
nding	A	PA	VD = PA * A									
n NA	B = B1 + B2 + ... + B17		VI =VI1 + VI2 +... + VI17	Wholesale distribution					WR			
nada	B1	PB1	VI1 = PB1 * B1				(t)		prop.	($/t)	$	
mark	B2	PB2	VI2 = PB2 * B2				E = 0.10C		0.40	WR	WR * 0.40E	
e Is.	B3	PB3	VI3 = PB3 * B3									
ance	.	.	.				Retail distribution					
nany	.	.	.									
hana	.	.	.									
eland	.	.	.									
occo	.	.	.									
ands	.	.	.									
rway	.	.	.									
oland	.	.	.									
tugal	.	.	.									
Fed.	.	.	.									
Spain	.	.	.									
eden	.	.	.									
UK	.	.	.									
USA	B17	PB17	VI17 = PB17*B17									
er EU												
thers												
ports												

						FC			FX			
et		Unit price	Value		Unit price	Consumers		Value	Exporters	Unit price	Value	Quantity
	(t)	($/t)	($)	(t)	($/t)	prop.	(t)	($)	prop.	($/t)	($)	(t)
0.05	0.05 C	FS	FS * 0.05C	0.01 C	Fres	0.01	0.01 C	FC * 0.01C	0.005		VE	2+... +D17
												D1
						WC			WX			D2
	(t)	($/t)	$	(t)	($/t)	prop.	(t)	$	prop.	($/t)	$	D3
0.30	0.30 E	WS	WS * 0.30 E	0.20 E	Wres	0.100	0.10 E	WC * 0.1 E	0.00		0	.
												.
						RC			RX			.
	(t)	($/t)	$	(t)	($/t)	prop.	(t)	$	prop.	($/t)	$	.
0.35	0.35 F	RS	RS * 0.35 F	0.35 F	Rres	0.300	0.30 F	RC * 0.3 F	0.00		0	.
												.
						SC			SX			.
	(t)			(t)	($/t)	prop.	(t)	$	prop.	($/t)	$	.
	G = 0.05 C+ 0.30 E+ 0.35 F			0.20 G	Sres	0.800	0.80 G	SC * 0.8 G	0.00		0	.
												.
	Food service distribution					FSC			FSX			.
				(t)		prop.	(t)	$	prop.	($/t)	$	.
		H = 0.01C+0.20 E+ 0.35 F+ 0.20G				1.000	1.00 H	esC * 1.0H	0.00		0	.
												D17

neadings:
r, P = Processor, W = Wholesaler, R = Retailer, S = Supermarket/Grocers, FS = Restaurant/Food Service, C = Consumer, X = Exporter
= Amount sold from Fisher to Processor, FW = Amount sold from Fisher to Wholesaler, etc.

Figure 2 Values of fishery products in each distributing sector (hypothetical data) sector's raw materials (the price paid to the sector's suppliers), (ii) the cost of handling and processing of the raw materials, and (iii) the margin received by the sector. In the case of the processing sector, for instance, the split can be expressed in a simple way as:

$$GB = c_r + m + c_h \qquad ...1)$$

where, GB denotes the gross economic benefits, c_r is the price paid to the supplier(s) of the sector's raw materials, c_h is the handling and processing costs, and m is the margin or value added by the sector. Since GB is the revenue from the sale of its products and c_h is the price paid by the sector to its suppliers, there are only two unknowns in the above equation, namely, m and c_h. c_h can be estimated using a combination of both secondary and survey data. Hence the margin, m, can be calculated. Secondly, we look at the profit and loss accounts of fishing companies for supplementary information on the profitability of the various sectors of the fishing sector.

Discussion and Extension of Current Work to the Entire North Atlantic

Figure 1 is an example of the market flow of one product, originated from one country. A similar model is needed for other products and for all NA countries before aggregation process can take place. This process would take place first at the product level, then at the country level. Using the model (both for quantities and values), we can arrive at an overall market anatomy for NA fisheries, as exemplified by Figure 3. Three main components incorporated in the anatomy are the quantity of fishery products traded in NA, the traded values and the economic benefits obtained by each distributing sector, as described in the section above.

A complete flow of the market channel involves the distribution of products from fishers to consumers through processors, wholesalers, retailers, supermarkets/groceries, and restaurants/food services. In addition to the domestic landings of the catch, there is a certain amount of imports within NA that again goes through the same channel. The total value of products traded in the market is therefore based on the quantity and the price at each sector, including the import amount. For example, fish processors in NA receive their products from three main sources: directly from fishers, from wholesalers and from importers. As processors pay different prices to each of these suppliers, the total value of fishery products that they trade (VP) is:

$$VP = \sum_i p_i q_i \qquad ...2)$$

where p and q are quantities and prices bought from and paid to i, where i = fishers, wholesalers and importers.

Similarly, the total value of fishery products that wholesalers trade (VW) is:

$$VW = \sum_j p_j q_j \qquad ...3)$$

where p and q are quantities and prices bought from and paid to j, where j = fishers, processors and importers.

The same calculation is carried out for other sectors in the market until the products reach consumers. At that level, the total value of fishery products that consumers receive (VC) is given by:

$$VC = \sum_k p_k q_k \qquad ...4)$$

where p and q are quantities and prices bought from and paid to k, where k = fishers, processors, wholesalers, retailers, supermarkets and restaurants.

It is implicit that the amount of imports from NA countries should balance with the amount of exports to NA countries, and to include both would result in double counting. It should also be noted that a reverse channel is possible, concerning the trading between processors and wholesalers, and this difference should be accounted for in the total amount traded.

Conclusion

The anatomy of the fishery products, as illustrated in Figure 3, is a good starting point for determining the sector(s) in the industry that capture(s) most of the economic benefits from the fishery. This information, coupled with knowledge about the ownership patterns in the fishing sector, can assist policy makers in designing sustainable ecosystem use policies, such as in the development of market intervention instruments, and in setting appropriate tax systems. It should be noted, however, that to arrive at such a simplistic diagram requires large aggregations of secondary data and to a lesser extent, information from personal contacts. In addition, information on marketing behaviour, and the approximation of costs and benefits associated with each product at each distributing sector are needed. Acknowledging that several assumptions had to be made to draw up the proposed framework, the methodology presented is useful as a tool for the compilation of different kinds of data from various sources on the processing and marketing of fishery products in North Atlantic.

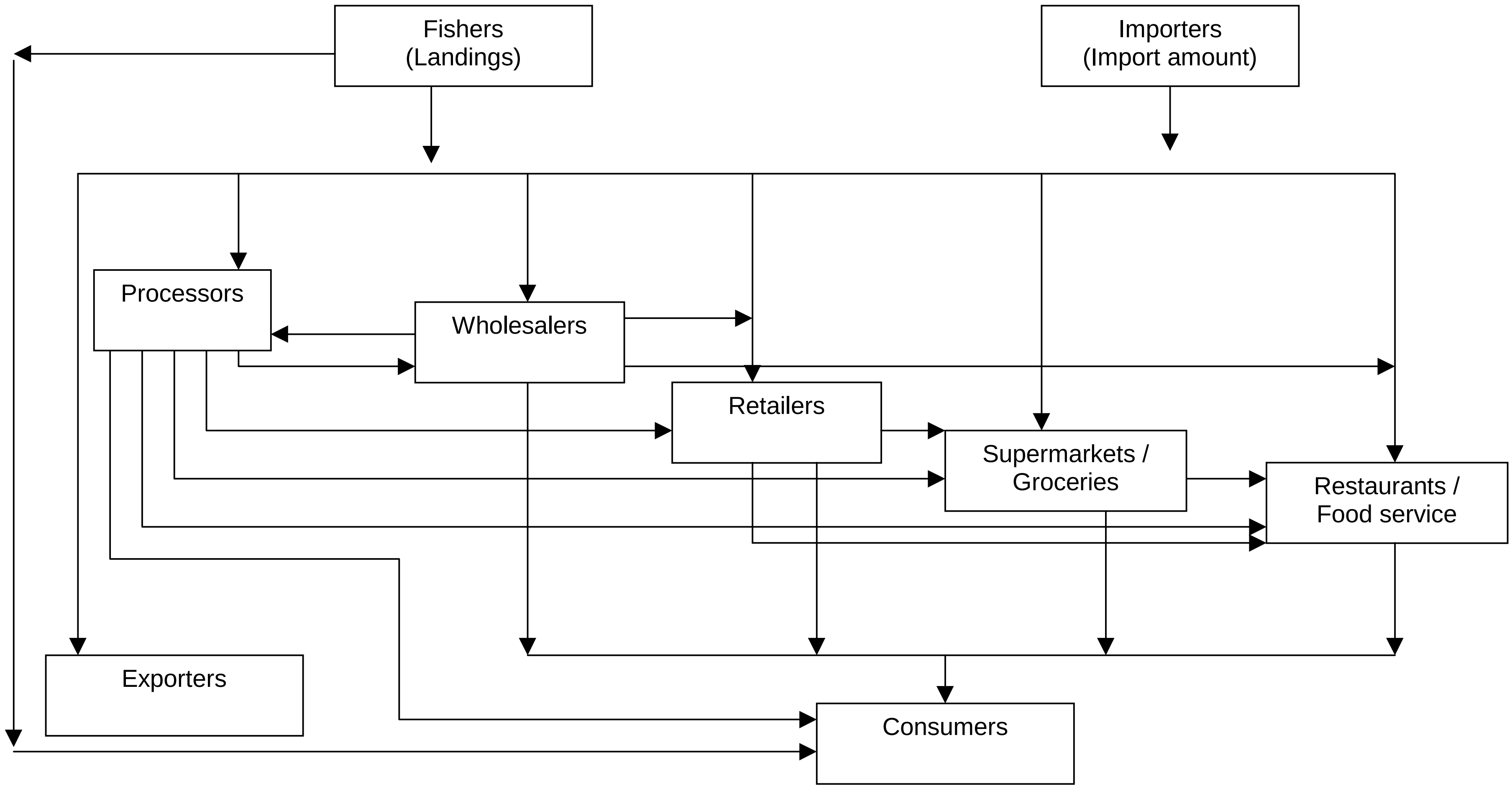


Figure 3 Schematic representation of the flow of products derived from North Atlantic fisheries.

ACKNOWLEDGEMENTS

The *Sea Around Us Project* is supported by the Environment Program of the Pew Charitable Trusts. We wish to thank the Food and Agriculture Organization of the United Nations (FAO), the Statistics Canada, and IPIMAR (Instituto de Investigação des Pescas e do Mar) for providing the data. We also thank Drs. Daniel Pauly and Tony Pitcher for detailed comments on the drafts of this contribution. Three reviewers, R. Quentin Grafton, Rognvaldur Hannesson, and Trond Bjorndal, provided interesting comments, for which we are thankful. Lastly, we extend our thanks to the participants of the *Sea Around Us Project* methodology review workshop, particularly the visiting scientists - Drs. Lee Alverson, Kevern Cochrane, Richard Grainger, Paul Fanning and Poul Degnbol, for their useful comments and suggestions.

REFERENCES

Crapo, C., Paust, B. and Babbitt, J. (1993) *Recoveries and yields from Pacific fish and shellfish.* Alaska Sea Grant College Program, University of Alaska Fairbanks, Fairbanks, Alaska. 32 pp.

Roy, N., Mazany, L. and Schrank, W.E. (1994) *A simultaneous equations model of the U.S. market for Canadian Atlantic cod.* Paper delivered at the Seventh International Conference of the International Institute of Fisheries Economics and Trade, Taipei, R.O.C., July 18-21, 1994.

Seafood Market Analyst (1997) 1996 *Year in Review* - U.S. Seafood Trade Report. Seafood Market Analyst, Narragansett, RI.

Watson, R., Guenette, S., Fanning, P. and Pitcher, T.J. (2000) The basis for change: Part 1 - Reconstructing fisheries catch and effort data. In: Pauly, D. and Pitcher, T. J. (eds) Methods for evaluating the impact of fisheries on North Atlantic ecosystems. *Fisheries Centre Research Reports* 8(2), 23-39.

QUANTIFYING THE ENERGY CONSUMED BY NORTH ATLANTIC FISHERIES

Peter Tyedmers[1]

Fisheries Centre,
The University of British Columbia,
Vancouver, Canada V6T 1Z4

ABSTRACT

As part of the *Sea Around Us Project* at the University of British Columbia, an analysis is being conducted of the fuel energy consumed by contemporary North Atlantic fisheries. Where possible, this will include evaluating changes in energy consumption over time for specific fisheries. The purpose of this paper is to describe the methods that will be used to achieve these ends. After reviewing the major findings of the published fisheries energy analysis literature, this paper introduces the two major thrusts of the planned research, and describes the techniques that will be used to address them. Specifically, there is a need to broadly apportion basin-wide catch data on a fishing gear- and vessel class-specific basis, along with a need to update fuel consumption estimates for a wide range of fisheries and fishing gear types. In support of the latter task, this paper summarizes my efforts to date acquiring detailed fuel consumption, catch, effort, and associated vessel description data for a variety of contemporary North Atlantic fisheries. Using two examples, I illustrate how data gathered in this way will be used directly to estimate the total fuel inputs to the associated fisheries. This is followed by a discussion and a further example of how generic fuel consumption rates based on vessel characteristics and fishing effort will be generated for specific gear sectors and applied to indirectly estimate the total fuel consumed in other fisheries. The paper ends by describing and providing examples of the ways in which fishery-specific and North Atlantic-wide fuel consumption estimates will ultimately be presented that facilitate comparison both amongst the fisheries evaluated and between fisheries and other food production sectors.

[1] Resource Management and Environmental Studies, 2206 East Mall, University of British Columbia, Vancouver, B.C. Canada V6T 1Z3 and *Sea Around Us Project*, Fisheries Centre, UBC (E-mail address: tyedmers@interchange.ubc.ca).

INTRODUCTION

The purpose of this paper is to describe the methods that will be used to quantify the major culturally mediated energy inputs to contemporary North Atlantic fisheries. In addition, examples will be provided that illustrate some of the techniques that will be used along with the forms in which the results will be expressed.

Key Terms Defined

Culturally mediated energy: fossil fuel and electrical energy dissipated in the process of human activities.

Energy intensity: the amount of direct and indirect culturally mediated energy required to provide a given quantity of a product or service of interest. In the current context, energy intensity is expressed in terms of the MJ of energy required to yield a round or live weight mass of fish or shellfish harvested.

Ecological Footprint: the area of land and/or water required to produce the resources consumed and to assimilate the wastes generated by a given population or activity on a continuous basis, wherever on Earth that land/water occurs.

Energy Return on Investment (EROI) ratio: a dimensionless ratio calculated by dividing the amount of useful energy produced by a given activity by the culturally mediated energy dissipated in providing it. In the case of food production systems, a common energy output used to calculate the EROI is the edible protein energy yield from the system being evaluated.

Fishing days: the number of complete or partial days in which a fishing vessel engages in fishing activities.

Sea days: the number of complete or partial days in which a fishing vessel is away from port on fishing related activities. Note, for a given fishing trip fishing days are always less than or equal to sea days.

As with all human activities, commercial fishing entails the consumption (or more accurately the dissipation) of matter and energy in support of their primary activity, the catching and killing of aquatic organisms. While these biophysical ‘costs’ are less obvious and consequently receive less attention than the direct impact that fishing has on targeted stocks and associated marine ecosystems, research indicates that they can be substantial. Moreover, they have real, if indirect, ecological impacts in and of themselves.

Following the oil price shocks of the 1970s a wave of research was undertaken to evaluate the energy intensity of a variety commercial fisheries (Wiviott and Mathews, 1975, Rochereau 1976, Leach 1976, Rawitscher 1978, Lorentzen 1978, Nomura 1980, Brown and Lugo, 1981, Hopper, 1981, Veal et al. 1981). The results of this and more recent research indicate that:

- Direct fuel energy inputs to fisheries typically account for between 75 and 90% of the total culturally mediated energy inputs. The remaining 10 to 25% of the total is comprised of direct and indirect energy inputs associated with vessel construction and maintenance, providing fish gear, and labour (Wiviott and Mathews, 1975, Rochereau 1976, Leach 1976, Edwardson, 1976, Rawitscher 1978, Lorentzen 1978, Tyedmers, forthcoming dissertation).
- Energy intensity can vary considerably between fishing gears used. In general, trawling tends to be more energy intensive than seining, purse seining or more passive techniques such as gillnetting, and trapping. (Wiviott and Mathews, 1975, Leach 1976, Edwardson, 1976, Lorentzen 1978, Rawitscher 1978, Nomura, 1980, Hopper, 1981)[2].
- In many cases, energy intensity was found to increase with vessel size within a given gear sector and fishery (Wiviott and Mathews, 1975, Rochereau 1976, Edwardson, 1976, Lorentzen 1978). However, exceptions to this have also been found (in particular, see Figure 1 in Edwardson, 1976).
- The energy intensity of a given fishery can increase dramatically over time as fisheries resources become scarcer, fleets expand, the average size of vessels increase, vessels travel further to fish, and become more technologically advanced (Brown and Lugo, 1981, Mitchell and Cleveland 1993)[3].

As part of the *Sea Around Us Project* at the University of British Columbia, I am undertaking an energy analysis of the fisheries of the North Atlantic. Ideally, such an analysis would encompass:

- direct fuel energy inputs;
- direct and indirect inputs to build and maintain fishing vessels;
- direct and indirect inputs to provide fishing gear 'consumed' in the process of fishing; and
- the energy required to sustain the fishing labour inputs.

However, because of the large number of fisheries to be considered, the heterogeneity that exists both between and within the fleets involved[4], and the general difficulty accessing reliable representative data the analysis will focus exclusively on estimating the direct fuel energy inputs to contemporary North Atlantic fisheries. Notwithstanding the above, in order to explore recent trends in energy use in commercial fisheries, fuel consumption time series estimates will be constructed for selected North Atlantic fisheries where necessary data are available.

METHODS TO BE USED

Estimating the total fuel energy inputs to as diverse a range of fisheries as currently occur in the North Atlantic presents two main challenges:

1. the catch must be broadly apportioned between fishing gears and sizes of vessels used; and
2. there is a need to update energy intensity estimates to better reflect contemporary North Atlantic fisheries.

Apportioning the Catch Amongst Fishing Gears and Vessel Classes

An important step in the process of estimating the total fuel energy consumed by contemporary North Atlantic fisheries will be to allocate the catch based on both the type of fishing gear used and the typical size class of vessel employed. This

---

[2] An exception to this relative energy intensity pattern occurs with respect to longlining, a passive fish harvesting technology which typically requires relatively large energy inputs relative to the amount of fish landed (Rawitscher 1978, Nomura, 1980).

[3] For example, Brown and Lugo (1981) estimated that between 1967 and 1975, while the fuel consumed by the entire U.S. fishing fleet (excluding vessels under 5 GRT) increased from 150 to 319 million gal/year the catch did not increase accordingly. As a result, the fossil energy input to edible protein energy output ratio for the U.S. fleet increased from 8:1 to almost 14:1 over the same period. Similarly, but on a smaller scale, Mitchell and Cleveland (1993) found between 1968 and 1988, the fuel energy input to edible protein output ratio of the New Bedford, Massachusetts fleet rose from ~6:1 to over 36:1.

[4] With respect to the types of gears used, the size of vessels within each fleet, and the complex material composition of fishing vessels and gears.

is because energy inputs vary with respect to both of these parameters.

In the case of some species, this will be a fairly straightforward process. For example, most contemporary Atlantic menhaden (*Brevoortia tyrannus*) landings are made using relatively large (over 500 gross ton) purse seiners. Similarly, trawling accounts for the majority of contemporary North Atlantic shrimp and prawn landings, while lobsters (*Homarus* spp.) are caught using traps deployed from relatively small vessels and scallops are harvested by dredging.

In other cases, however, where a given species of fish or shellfish is typically harvested using more than one type of fishing gear[5], or vessels of dramatically different sizes, the North Atlantic-wide landings of these species will have to be apportioned accordingly[6]. This will be done using detailed catch statistics that relate landings to fishing gear and size of vessel employed. To this end, *Sea Around Us Project* team members are assembling statistics of catch by gear and vessel size from a variety of sources (see Watson et al. 2000 for a description of this process). For example, unpublished data sets that relate landings to fishing gear used, vessel size, and horsepower, and total fleet effort (measured either in terms of fishing days or days at sea) have been received and are currently being processed by the team for both Canadian and foreign fishing vessels operating in Canada's Atlantic Exclusive Economic Zone. Similar data sets are currently on order for U.S. North Atlantic fisheries.

However, complete detailed coverage of all North Atlantic fisheries will not be possible given the limitations of the data available. It will therefore be necessary to apply the proportion of the gear- and vessel size-specific catch of each major species from the countries and regions of the North Atlantic for which data are available to the landings of the entire North Atlantic. In the final report of this project, the extent of this extrapolation from known to total catch of the major species will be presented and the uncertainties that result will be discussed.

Estimating the Fuel Energy Inputs to Contemporary Fisheries and Gear Sectors

While many of the published commercial fishery energy analyses were conducted on North Atlantic fisheries, virtually all are based on primary data collected during the 1970's (Appendix). Because of changes that have been likely to occur over the last 25 to 30 years[7], I am reluctant to directly apply energy intensity estimates from fisheries of the 1970s to those of the late 1990s unless absolutely necessary. Consequently, I am actively updating estimates of fuel energy inputs to a wide range of contemporary fisheries and gear types using two main approaches.

Direct Solicitation of Fuel Consumption Data

I have begun to solicit annual fuel consumption, landings and temporal fishing effort (both fishing days and sea days) data together with the physical characteristics of the associated vessels from companies and individuals currently engaged in North Atlantic fisheries. Table 1 summarizes the fisheries, gear types, and vessel characteristics represented by this data collection effort to date[8] and presents preliminary estimates of the resulting fuel consumption per live weight tonne of fish or shellfish harvested.

To illustrate how fuel consumption and landings data will be used to estimate total fuel energy inputs to specific contemporary North Atlantic fisheries, preliminary estimates were made of the total direct fuel consumption associated with the 1997 Atlantic menhaden fishery (below) and the 1997 North Atlantic-wide scallop fisheries (adjacent).

Although by the end of the project I intend to have acquired data representing many more vessels and fisheries than are outlined in Table 1,

---

[5] For example, Atlantic cod (*Gadus morhua*) are harvested using bottom trawls, seines, gillnets, traps, and longlines while bluefin tuna (*Thunnus thynnus*) are harvested using purse seines, seines, gillnets, traps, hook and line and harpoon.

[6] While apportioning the catch based on gear type used will be straightforward, because fishing vessel size, either measured in terms of vessel length, gross tonnage or horsepower varies over a continuum, for simplicity it will be necessary to establish some arbitrary size classes. For example, appropriate size classes that might be used are as follows: under 5 gross tonnes (GT), from 5 to 50 GT, from 50 to 150 GT, from 150 to 500 GT, from 500 to 1000 GT, from 1000 to 2000 GT, and over 2000 GT (Ruttan, et al. 2000).

[7] Particularly with respect to: stock abundance, fleet size, average fishing trip length, average vessel size and horsepower, engine fuel efficiency and types of fuel used, etc. (see Brown and Lugo (1981) and Mitchell and Cleveland (1993) for examples of how fuel energy inputs to fisheries can change over time).

[8] Other fisheries for which fuel consumption inquiries have been initiated include the English groundfish trawl and longline fishery in the English Channel, and the Icelandic capelin trawl and purse seine fisheries and groundfish trawl fisheries.

the extent of data coverage will vary widely between fisheries. In some cases, vessels for which I have data will represent a relatively large proportion (>50%) of the total annual basin-wide catch of a given species. In these cases, the extrapolated

Table 1. Fisheries, Gears and Vessels for Which Fuel Consumption Data has been Acquired.

Fishery	Gear Type	Vessel Size (Tonnage/HP)	Vessel(s) Represented	Annual Catch by Vessel(s) (round tonnes)	Fishing Seasons Represented[a]	Fuel Consumption[b] (litres/tonne)
Shrimp - NW Atlantic	Trawl	2,290/4,023	1 Freezer Trawler	~4,200	1993 to 1999 inclusive	850
Atlantic menhaden - US	Purse seine	540/1,800 to 750/2,000	13 Purse Seiners	~175,000	1998 & 1999	31.5
Gulf menhaden - US [c]	Purse seine	540/1,800 to 750/2,000	37 Purse Seiners	~400,000	1998 & 1999	39.2
Ground fish - NW Atlantic	Trawl	540/1,300 to 802/2,400	8 Trawlers	~10,000	1999	347
Cod - NW Atlantic	Norwegian Seine	545/1,250	2 Seiners	~1,000	1999	230
Scallops - Georges Bank	Dredge	309/765 to 330/990	5 Draggers	~5,500	1998 & 1999	350

Notes:

a. Data represents all fishing trips undertaken during the years indicated.

b. Calculated by dividing the total fuel consumed in litres for all vessels and seasons represented by the total resulting landings in round or live weight tonnes.

c. For the purposes of our project, the Gulf of Mexico is not considered part of the North Atlantic. However, data from the Gulf menhaden fishery may be used to help characterise purse seine fisheries generally.

estimates of the total fuel consumed by the basin-wide fishery will be relatively robust. In other instances, however, the data coverage may only amount to the equivalent of few percent of the total annual catch. Consequently, the resulting extrapolated estimates of total fuel consumed will be less accurate. As part of the final report, the uncertainties that result from extrapolating from sample sets of various sizes will be discussed.

Because acquiring fuel consumption data directly from fishers and fishing companies is a slow, labour intensive process and at best only a small fraction of all the fishing vessels active in the North Atlantic can be canvassed[9] I am

Example 1 - Fuel Inputs to All North Atlantic Atlantic Menhaden Fisheries in 1997

Omega Protein Limited, of Hammond, Louisiana, provided two years (1998 and 1999) of detailed catch and fuel consumption data from their fleet of 13 purse seiners based in Reedville, Virginia. From these data, representing a two year total catch of over 368,600 wet tonnes, I estimate that the Atlantic menhaden fishery consumed an average of 31.5 litres of diesel per tonne of fish landed (Table 1).

Multiplying this rate of fuel consumption by 322,239 tonnes, the total 1997 North Atlantic-wide Atlantic menhaden landings (as reported by the Food and Agriculture Organization (FAO) of the United Nations), I estimate that a total of 10 million litres of diesel were consumed in this fishery.

Example 2 - Fuel Inputs to All North Atlantic Scallop Fisheries in 1997

Anonymous sources provided a total of ten vessel-years of detailed catch and fuel consumption data representing five scallop draggers active in the North Atlantic during the late 1990's. From these data, representing a total catch of almost 11,000 live weight tonnes, I have estimated that the North Atlantic scallop fishery consumed an average of 350 litres of diesel per live weight tonne of scallops landed (Table 1).

Multiplying this rate of fuel consumption by 171,013 tonnes, the total 1997 North Atlantic-wide scallop catch (as reported by the FAO), I estimate that this fishery consumed a total of 60 million litres of diesel in 1997.

[9] While the data acquisition efforts to date have been reasonably successful, given the sensitivity that many individuals and companies can display regarding the release of information that could be perceived to be at variance with their interests, it is possible that

concentrating my efforts on fisheries and gear sectors that either: 1) account for relatively large proportions of the total North Atlantic catch of all species, or 2) use inherently energy intensive fishing gears. In this way, I hope to reduce the degree of error in the final estimate of the total energy inputs to North Atlantic fisheries. As a result, the large tonnage fisheries for small pelagic species such as herring, capelin and menhaden using purse seine, trawl, and seine gears are of particular interest as are trawl, and longline fisheries generally. However, where opportunities arise to acquire data representing smaller tonnage fisheries and less 'productive' gears, these will be pursued.

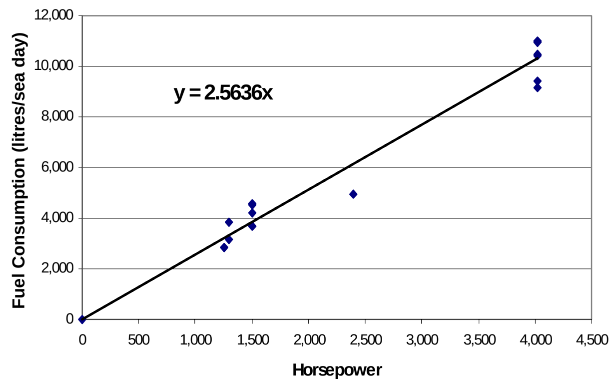


Figure 1. Daily Fuel Consumption Versus Propulsive Horsepower Relationship for Contemporary Trawlers and Seiners (17 vessel-seasons represented)

Inferring Fuel Consumption for Specific Gear Sectors Using Fleet Effort and Horsepower Characteristics

Based on a preliminary analysis of the fuel consumption and vessel data collected to date, it appears that for at least some gear sectors the *rate* of fuel consumption under normal operating conditions has relatively little to do with either the species being targeted or the resulting size of the catch. Instead, average fuel consumption rates seem to depend more on the size/power of the fishing vessels themselves together with the unique characteristics associated with deploying, fishing and retrieving the specific gear being used.

By way of example, a regression analysis was conducted of the relationship between average fuel consumption per day and main engine horsepower for the nine trawlers and two seiners (described in Table 1), representing a total of 17 vessel-seasons in which the number of sea days per season varied from 24 to 333 and averaged 178.

The results of the analysis, which was forced through the intercept based on the assumption that no fuel will be burned by a vessel without an engine, indicate that 2.56 litres of diesel are consumed per horsepower•sea-day (s.e. 0.054, r^2 0.965) (Figure 1).

From this and other gear-specific relationships that I have yet to quantify but am confident will emerge, either derived from fishing effort and main engine horsepower data alone or a combination of vessel characteristics[10], estimates will be made of total fuel consumption and resulting energy intensity for some fisheries for which I have been unable to directly acquire fuel consumption data. Specifically, this will be possible in situations in which catch data can be related to vessel characteristics and total days at sea for a given gear sector. For example, by multiplying the product of average fleet horsepower and total days at sea by the gear-specific fuel consumption rate derived above (2.56 litres per horsepower•sea-day) an estimate of total fuel input to a trawl fishery can be made[11].

For many fisheries, days at sea data together with the physical characteristic of the vessels engaged in a fishery - this typically includes vessel tonnage and/or horsepower - are available from national fisheries management agencies. For example, the *Sea Around Us Project* team has already acquired Canadian and foreign vessel catch, gross tonnage, horsepower and temporal fishing effort data for all fisheries that occur within Canada's Atlantic Exclusive Economic Zone from Fisheries and Oceans Canada (see Watson et al. 2000). At the time of writing, this data set was being re-formatted into a form that could be easily used by all team members. We are also in the process of ordering a comparable data set for all U.S.

---

additional direct fuel consumption data may be difficult to acquire.

[10] In the preliminary example given, the regression analysis was conducted using only fuel consumption per sea day and main propulsive horsepower. However, as more data becomes available representing a wider range of vessels and gear types, I will also perform multiple regression analyses in which average fuel consumption per sea day will be regressed against gross vessel tonnage, vessel length, the presence or absence of auxiliary engines in addition to main propulsive horsepower.

[11] This is the technique that Brown and Lugo (1981) employed to estimate the energy inputs to all U.S. fisheries over the period from 1967 to 1975. In doing so they derived generic fuel consumption rates for three gear sectors - trawlers, purse seiners and all other gears combined.

Atlantic fisheries from the National Marine Fisheries Service and are in the process of tracking down similar data from other North Atlantic fishing countries.

In situations in which fisheries agencies only report temporal fleet effort in terms of fishing days and not total days at sea, a fishery-appropriate correction factor will have to be applied. This is because not only do fishing vessels often burn fuel at higher rates when steaming than they do when actively fishing but the transit time to and from fishing grounds can account for a substantial portion of a vessel's total operating time and hence its fuel consumption. For example, amongst the groundfish trawlers and seiners for which I have acquired data, fully 23% of their time at sea during the 1999 fishing season was spent in transit to and from fishing grounds. As a result, for any comparable groundfish trawl fisheries I would apply a 1.3 times correction factor[12] to any temporal effort data that are reported only in terms of fishing days[13].

Where only the gross tonnage of vessels engaged in a fishery is available in conjunction with catch and temporal effort data, estimates of total fleet horsepower will be made using published fishing vessel descriptions and databases. For example, *Fishing Vessels of Britain and Ireland 2000*[14] will be used to construct gear-specific horsepower profiles for British and/or Irish fleets as needed while fleet characteristic data files are being solicited from both Fisheries and Oceans Canada and the National Marine Fisheries Service in the U.S. Finally, the project may acquire a database from Lloyd's Maritime Information Service in London that provides the physical characteristics, including engine power, of most of the world's fishing vessels over 100 GT.

As a simple illustration of the techniques described above, I have made a preliminary estimate of the fuel consumed by foreign vessels fishing for turbot (*Reinhardtius hippoglossoides*) in the Canadian Atlantic EEZ during 1996 (next page).

[12] The 1.3 correction factor was determined by taking the inverse of 0.77, the proportion of the total days at sea that eight trawlers and two seiners spent actively fishing for groundfish in 1999 in the NW Atlantic.

[13] However, in other fisheries, particularly those conducted in nearshore waters, no correction factor would be applied because transit times are negligible.

[14] *Fishing Vessels of Britain and Ireland 2000,* published by Fishing News, London, provides descriptions of all British and Irish fishing vessels over 12m in length including gross and net tonnage, main engine power, and type of fishing gear deployed.

Inferring Fuel Consumption When Only Catch Data are Available

For those fisheries in which only catch data are available and I have been unable to directly acquire fuel consumption data from one or more vessels active in the fishery, I will assign an energy intensity value based on the analyses of similar fisheries from other parts of the North Atlantic[15]. Factors to be considered when identifying a comparable fishery include: species caught, fishing gear used (if known), proximity to the fishing grounds, whether the fisheries are conducted on the same or similar populations of fish, and the known or probable purpose to which the catch is put.

To illustrate how this last factor may be useful in helping to constrain the energy intensity of a given fishery, consider the example of a fishery in which the catch is used entirely for reduction to fishmeal and oil. In this situation, the catch has a very low unit economic value and hence the costs of conducting the fishery must also be relatively low. As a result, the energy intensity of that fishery would likely be quite low since in most contemporary fisheries fuel costs represent a fairly large portion of total operating costs.

Finally, in rare instances, I may have to apply published energy intensity values from the same or a comparable fishery from the 1970s. Fortunately, by applying energy intensity values from 25 year ago to comparable contemporary fisheries should result in relatively conservative estimates of contemporary energy use given the changes in both stock, vessel and fleet sizes that have been likely to occur in the interim.

Using Total Fuel Consumption by a Given Region or Nation's Fisheries to Constrain The Results

I hope to identify data from which an estimate of the total annual fuel inputs to all fisheries within a given geographic region or political jurisdiction bordering the North Atlantic can be made. For example, for some countries it may be possible to quantify total fuel consumption by all commercial fisheries using fuel tax rebate data. While such an approach will not provide gear- or fishery-specific energy intensity estimates, it will help to confirm/constrain the estimates of the energy inputs to those fisheries.

[15] This is essentially the process that Hammer (1991) used to estimate the total fuel energy inputs associated with all domestic Swedish fisheries and the other fisheries whose products are traded by Sweden.

Example 3- Fuel Inputs to Foreign Vessels Fishing for Turbot in Canada's EEZ in 1996

In this example, fishing effort data for four size classes of trawlers, as recorded by NAFO (see Watson et al 2000), was used to estimate the total horsepower•sea-days required to land a given catch of turbot in 1996 (Table 2).

Table 2. Turbot Catch, Effort and Average Horsepower of Foreign Vessels Fishing in Canadian Waters in 1996

Trawler Size Class (GT)	Catch (tonnes)[a]	Fishing Days[a]	Sea Days[b]	Average Horsepower[c]	HP-Days
150 to 499	2,917	77	100.1	1,000	100,100
500 to 999	6,694	546	709.8	2,000	1,419,600
1000 to 1999	4,175	39	50.7	3,000	152,100
>2000	1,327	20	26.0	4,500	117,000
TOTALS:	15,113				1,788,800

Notes: a. Catch and corresponding fishing effort data for four classes of trawlers provided by Fisheries and Oceans Canada.
b. Sea-days calculated by multiplying fishing days by 1.3 based on the fishing-days to sea-days relationship that we have established for Canadian trawlers (see text above).
c. Preliminary estimates of the average horsepower of trawlers in these four size classes based on gross tonnage and horsepower data for 180 British fishing vessels (all gear types) as reported in the latest edition of *Olsen's Fisherman's Nautical Almanack* (Simpson, 2000).

Multiplying the total horsepower•days of effort (Table 2) by the generic trawler fuel consumption rate of 2.56 litre/horsepower•sea-day (see text above), I estimate that this fishery consumed approximately 4,580,000 litres of diesel in the process of catching 15,113 tonnes of turbot. This indicates an average fuel consumption rate of about 300 litres/tonne for this fishery.

Constructing Energy Consumption Time-Series

Where data are available, I will also construct fisheries specific fuel consumption time-series. The most appropriate method of doing this will be to use time series temporal fishing effort and fleet characteristic data together with the fishing gear specific fuel consumption rates that will be generated for late 1990s fisheries. Using this approach, the results will better reflect changes in total fleet effort and stock abundance over time. In addition, for those North Atlantic fisheries covered by energy analyses conducted during the 1970s, I will be able to evaluate energy intensity changes that have occurred over the last 25 to 30 years.

EXPRESSING THE RESULTS

Once fuel input estimates are generated for specific fisheries and for the entire North Atlantic, it will be possible to re-express the results in a variety of forms that either:

- facilitate comparisons both amongst fisheries and between fisheries and other food producing activities; or
- provide additional insights into the potential impacts of contemporary fisheries.

Specifically, energy intensities (MJ/tonne landed), and edible protein EROI ratios will be calculated for individual fisheries and as a weighted average of all North Atlantic fisheries. Employing both of these measures is useful because they provide different perspectives on the efficiency of fisheries (and food production activities generally), reflecting the often dramatic differences in the edible yield and protein contents of fish and shellfish. In addition, edible protein EROI ratios, or its inverse the energy cost of providing edible protein, have been shown to be particularly useful for analysing changes in fisheries over time (Brown and Lugo, 1981, Mitchell and Cleveland, 1993) and as a basis for comparing diverse food producing systems or technologies (Folke and Kautsky, 1991, Larsson et al, 1994, Pimentel, et al, 1996, Pimentel, 1997). Table 3 presents preliminary estimates of the energy intensity and edible protein EROI ratio for the three Examples provided above.

Table 3. Re-Expressing the Results of the Three Examples Provided

	Total 1997 Atlantic Menhaden Landings	Total 1997 Scallop Landings	1996 Foreign Fleet Turbot Landings in Canada's EEZ
Landings (round tonnes)[a]	322,239	171,013	15,113
Diesel consumed (litres)[a]	10,163,400	59,854,550	4,580,000
Rate of fuel consumption (l/t)[b]	31.5	350	300
Energy intensity (MJ/t)[c]	1,135	12,600	10,800
Edible protein EROI[d]	n/a	2.5%	16%
CO_2 emission intensity (kg/t)[e]	84	932	806
Total CO_2 emissions (tonnes)[e]	27,000	159,400	12,200
Ecological footprint (hectares of CO_2 assimilation forest)[f]	4,100	24,200	1,850

Notes:
a. Landings and total fuel consumption from Examples above.
b. Average fuel consumption rates from Table 1.
c. Calculated by multiplying the fuel consumption rate (litres/tonne of shell/fish harvested) by 36.036 MJ/litre, diesel fuel's net energy release upon combustion (calculated from data in Rose and Cooper, 1977, Tables 5.24 and 5.25).
c. Edible protein EROI was not calculated for menhaden as the bulk are not used for direct human consumption. For other species, protein EROI was calculated by dividing the edible protein energy content of a tonne of seafood (in MJ) by the fossil fuel energy consumed to harvest a round tonne. Protein energy content determined by multiplying 1000kg by the maximum edible meat yield rate (12.5% for scallops (Dominion Bureau of Statistics, 1931) and 56% for turbot (Bykov, 1983)), by the average protein energy content of the meat (scallops assumed to be the same as oysters at 10.6% (Pimentel et al, 1996) and 13% for turbot (Bykov, 1983)), by the nutritional energy of protein: 23.6 MJ/kg.
d. Calculated by multiplying the diesel fuel consumed (in MJ) by 73.9 gm CO_2/MJ, the average rate of CO_2 emissions from a variety of vessels under normal operating conditions (calculated from Lloyd's Register Engineering Services, 1995, Table 5, p. 17).
e. Calculated by dividing tonnes of carbon emitted (CO_2 emissions divided by 3.66) by 1.8 t C/ha•yr, a conservative estimate of the global average rate of carbon assimilation by the world's forests (Wackernagel and Rees, 1996).

To illustrate the potential contribution that North Atlantic fisheries make to global climate change, energy use related CO_2 emission intensities will be calculated for individual fisheries (e.g. tonnes CO_2 /tonne landed) along with the total CO_2 emissions for all fisheries (Table 3). Finally, to help place the scale of these emissions in context, the notional fossil fuel consumption-related ecological footprint will be calculated for specific fisheries (e.g. ha of CO_2 assimilation forest required/tonne landed) and all North Atlantic fisheries combined (Table 3). While this latter measure is a relatively new pedagogical tool (see Rees and Wackernagel, Wackernagel and Rees, 1996,) and the methods used to ‘footprint’ energy use are still undergoing refinement[16], it has been applied in the analyses of other fish producing systems (Larsson et al, 1994, Tyedmers, forthcoming dissertation) in addition to a range of other activities (Wackernagel and Rees, 1996, Folke et al, 1997). Finally, the results can then be combined with estimates of the marine primary productivity directly appropriated by the fisheries themselves (see Christensen and Walters, 2000) to produce a more complete picture of the ecological footprint of contemporary North Atlantic commercial fisheries.

ACKNOWLEDGEMENTS

This research, along with the rest of the *Sea Around Us Project*, is supported by the Environment Program of the Pew Charitable Trusts.

I would like to thank Mr. Mike Wilson of Omega Protein Ltd. along with the many others within the commercial fishing industry who wish to remain anonymous, for their time and effort providing me with the data used in this paper. Finally, I would also like to thank my colleagues Reg Watson, Lore Ruttan and Sylvie Guénette, and last but not least Brenda Tyedmers for their assistance in the preparation of aspects of this document.

[16] Most ecological footprint analyses have employed one of two methods when estimating the ecosystem support area associated with fossil fuel use. The first is to calculate the area of ecosystem required to produce a contemporary biologically sourced liquid fossil fuel substitute such as ethanol, methanol, soydiesel, or fish oil. The second, and the one that will used in this analysis, is to estimate the area of forest ecosystem required to sequester the CO_2 produced through the combustion of fossil fuels (see Wackernagel and Rees, 1996 for a review of these approaches).

REFERENCES

Brown, S. and Lugo, A.E. (1981) *Management and Status of U.S. Commercial Marine Fisheries,* Council on Environmental Quality. Washington. D.C. 45 pp.

Bykov, V.P. (1983) *Marine Fishes: chemical composition and processing properties*, Amerind Publishing Co. Pvt. Ltd., New Delhi. 333 pp.

Christensen, V. and Walters, C. (2000) Ecopath with Ecosim: Methods, Capabilities and Limitations. In: Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. Pauly, D. and Pitcher, T.J. (eds). *Fisheries Centre Research Reports* 8(2), 79-105.

Dominion Bureau of Statistics (1931) *Fisheries Statistics of Canada 1930*. F.A. Acland

Edwardson, W. (1976) The Energy Cost of Fishing. *Fishing News International* 15(2).

Folke C. and Kautsky, N. (1991) Ecological Economic Principles for Aquaculture Development. In: *Nutritional Strategies and Aquaculture Wastes* (Proceedings of the First International Symposium on Nutritional Strategies in Management of Aquaculture Waste). C.C. Cowey, and C.Y. Cho (eds) College of Biological Sciences, University of Guelph, Guelph, Canada, 207-222.

Folke, C., Jansson, A., Larsson, J. and Costanza, R. (1997) Ecosystem Appropriation by Cities. *Ambio* 26(3), 167-172.

Hammer, M. (1991) Marine Ecosystem Support to Fisheries and Fish Trade. In: Linking the Natural Environment and the Economy: *Essays from the Eco-Eco Group* (eds. C. Folke and T. Kaberger) Kluwer Academic Publishers, Dordrecht, 189-209.

Hooper, A.G. (1982) Energy Efficiency in Fishing Vessels. In: Fishing Industry Energy Conservation Conference (Proceedings of the Fishing Industry Energy Conservation Conference, Seattle, Washington. 26 October - 27 October, 1981). The Society of Naval Architects and Marine Engineers, New York, 55-81.

Leach, G. (1976) *Energy and Food Production.* IPC Science and Technology Press Ltd. Guildford. 137 pp.

Lloyd's Register Engineering Services (1995) *Marine Exhaust Emissions Research Programme.* Lloyd's Register of Shipping, London. 63 pp.

Larsson, J., Folke, C. and Kautsky (1994) Ecological Limitations and Appropriation of Ecosystem Support by Shrimp Farming in Colombia. *Environmental Management* 18(5), 663-676.

Lorentzen, G. (1978) Energibalanse i den norske fiskerinæring. *Meldingen SSF M 2*, 5-9. [In Norwegian]

Mitchell, C. and Cleveland, C.J. (1993) Resource Scarcity, Energy Use and Environmental Impact: A Case study of the New Bedford, Massachusetts, USA, Fisheries. *Environmental Management* 17(3), 305-317.

Nomura, M. (1980) Influence of Fish Behavior on Use and Design of Setnets. In: Fish Behavior and Its Use in the Capture and Culture of Fishes. (ICLARM Conference Proceedings 5). J.E. Bardach, J.J. Magnuson, R.C. May and J.M. Reinhart, eds. International Center for Living Aquatic Resources Management, Manila, 446-472.

Pimentel, D., Shanks, R.E. and Rylander, J.C. (1996) Bioethics of Fish Production: Energy and the Environment. *Journal of Agricultural and Environmental Ethics* 9(2), 144-164.

Pimentel, D. (1997) Livestock Production: Energy Inputs and the Environment. In: *Proceedings of the Canadian Society of Animal Science 47th Annual Meeting* (July 24-26, 1997, Montréal, Québec). S.L. Scott, and X. Zhao (eds), 16-26.

Rawitscher, M.A. (1978) *Energy Cost of Nutrients in the American Diet.* Ph.D. Thesis, The University of Connecticut. 141 pp.

Rees, W.E and Wackernagel, M. (1994) Ecological Footprints and Appropriated Carrying Capacity: Measuring the Natural Capital Requirements of the Human Economy. In: *Investing in Natural Capital: The Ecological Economics Approach to Sustainability.* A. Jansson, M. Hammer, C. Folke, and R. Costanza (eds), Island Press, Washington, DC, 362-390.

Rochereau, S.P. (1976) *Energy Analysis and Coastal Shelf Resource Management: Nuclear Power Generation vs. Sea-Food Protein Production in the Northeast Region of the U.S.* Ph.D. Thesis, Cornell University.

Rose, J.W. and Cooper J. R. (1977) *Technical Data on Fuel.* The British National Committee World Energy Conference, London.

Ruttan, L., Gayanilo, F.C. Jr, Sumaila, U.R. and Pauly, D. (2000) Small Versus Large-Scale Fisheries: A mult-species, multi-fleet model for Evaluating their interactions and potential benefits. In: Methods for evaluating the impacts on fisheries on North Atlantic ecosystems. Pauly, D. and Pitcher, T.J. (eds). *Fisheries Centre Research Reports* 8(2), 64-78.

Simpson, P. (2000) *Olsen's Fisherman's Nautical Almanack.* E.T.W. Dennis & Sons. Scarborough. 688 pp.

Tyedmers, P.H. (forthcoming Ph.D. dissertation) *Salmon and Sustainability: The Biophysical Costs of the Commercial Salmon Fishery and Intensive Salmon Culture in British Columbia, Canada*, The University of British Columbia.

Veal, D., Rawson M.V. and Hosking W. (1982) Structure, Strategy and Fuel Consumption in the Gulf Shrimp Fleet. In: *Fishing Industry Energy Conservation Conference* (Proceedings of the Fishing Industry Energy Conservation Conference, Seattle, Washington. 26 October - 27 October, 1981). The Society of Naval Architects and Marine Engineers, New York, 43-54.

Watson, R., Guénette, S., Fanning, P. and Pitcher, T. (2000) The Basis for Change I: Reconstructing fisheries catch and effort data. In: Methods for evaluating the impacts of fisheries on North Atlantic ecosystems. Pauly, D. and Pitcher, T. J. (eds). *Fisheries Centre Research Reports* 8(2), 23-39.

Wackernagel, M. and Rees, W.E. (1996) *Our Ecological Footprint: Reducing Human Impact on the Earth.* New Society Publishers, Gabriola Island, British Columbia. 160 pp.

Wiviott D.J. and Mathews, S.B. (1975) Energy Efficiency Comparison between the Washington and Japanese Otter Trawl Fisheries of the Northeast Pacific. *Marine Fisheries Review* 37(4) 21-24.

APPENDIX

Summary of Published Fisheries Energy Analysis

Reference	Data From	Fishery Characteristics: Based	Fishing Ground	Gear Used	Species Targeted	Annual Landings per vessel (t)	Vessel Characteristics: Length (m)	Tonnage	Main HP	Total Energy Inputs (GJ/t)	(GJ/year)	Includes*
Leach, 1976	1969	England	Various	Various	Various					34.6		unknown
Leach, 1976	?	Peru	coastal?	n/a	Anchoveta					0.5		F
Leach, 1976	1972	U.S.	Gulf of Mexico	various	Shrimp					358.0		F
Leach, 1976	1974	Australia	n/a	trawlers	Shrimp		>17			38.1		F,V
Leach, 1976	1972	Malta	n/a	Various	Various					40.3		F
Wiviott and Mathews, 1975	1971-72	Washington State	NE Pacific	Bottom Trawl	Various	449	24.4	86	300	9.6	4,306	F,O,V
Wiviott and Mathews, 1975	1971-72	Japan	NE Pacific	Bottom Trawl	Various	2,440	n/a	1,947	2,648	51.5	125,699	F,O,V
Rochereau, 1976	1972	NE, USA		Various	Various			5-9			237	F,G,O,V
Rochereau, 1976	1972	NE, USA		Various	Various			10-19			336	F,G,O,V
Rochereau, 1976	1972	NE, USA		Various	Various			20-29			484	F,G,O,V
Rochereau, 1976	1972	NE, USA		Various	Various			30-39			631	F,G,O,V
Rochereau, 1976	1972	NE, USA		Various	Various			40-59			777	F,G,O,V
Rochereau, 1976	1972	NE, USA		Various	Various			60-79			1,167	F,G,O,V
Rochereau, 1976	1972	NE, USA		Various	Various			80-99			1,556	F,G,O,V
Rochereau, 1976	1972	NE, USA		Various	Various			100-119			1,973	F,G,O,V
Rochereau, 1976	1972	NE, USA		Various	Various			120-139			2,390	F,G,O,V
Rochereau, 1976	1972	NE, USA		Various	Various			140-159			2,956	F,G,O,V
Rochereau, 1976	1972	NE, USA		Various	Various			160-169			3,522	F,G,O,V
Rochereau, 1976	1972	NE, USA		Various	Various			170-179			4,255	F,G,O,V
Rochereau, 1976	1972	NE, USA		Various	Various			180-199			4,988	F,G,O,V
Rochereau, 1976	1972	NE, USA		Various	Various			220-239			6,512	F,G,O,V
Rochereau, 1976	1972	NE, USA		Various	Various			250-299			7,832	F,G,O,V
Rochereau, 1976	1972	NE, USA		Various	Various			300-319			9,274	F,G,O,V
Rochereau, 1976	1972	NE, USA		Various	Various			320-339			10,592	F,G,O,V
Edwardson, 1976	1973	Scotland	Unknown	Pair trawl	Pelagic species	357	16.8			10.8	3,854	F,G,O,V
Edwardson, 1976	1973	Scotland	Unknown	Purse seine	Pelagic species	3,976	24.4			2.7	10,611	F,G,O,V
Edwardson, 1976	1973	Scotland	Unknown	Seine	Demersal species	294	19.8			15.3	4,512	F,G,O,V
Edwardson, 1976	1973	Scotland	Unknown	Trawl	Demersal species	628	24.4			19.7	12,376	F,G,O,V
Edwardson, 1976	1973	Scotland	Unknown	Trawl	Demersal species	697	36.6			35.8	24,923	F,G,O,V
Edwardson, 1976	1973	Unknown	Unknown	Trawl	Demersal species	1,869	64			56.3	105,225	F,G,O,V

Note: *: F = fuel, G = gear and/or bait, O = other operating inputs (including labour), V = Vessel building and maintenance

Reference	Data From	Based	Fishing Ground	Gear Used	Species Targeted	Annual Landings per vessel (t)	Length (m)	Tonnage	Main HP	Total Energy Inputs (GJ/t)	Total Energy Inputs (GJ/year)	Includes*
		----------------Fishery Characteristics--------------					--------Vessel Characteristics--------					
Rawitscher, 1978	1973	California	Central Pacific	Purse Seine	Tuna					31.6		F,G,O,V
Rawitscher, 1978	1974	California	Central Pacific	Purse Seine	Tuna					31.0		F,G,O,V
Rawitscher, 1978	1975	California	Central Pacific	Purse Seine	Tuna	1,570				62.3	97,692	F,O,V
Rawitscher, 1978	1973	Maine	Coastal Maine	Purse Seine	Herring					2.3		F,G,O,V
Rawitscher, 1978	1974	Maine	Coastal Maine	Purse Seine	Herring					2.4		F,G,O,V
Rawitscher, 1978	1974	Maine	Coastal Maine	Purse Seine	Herring	18.3				2.2	40	F,G,V
Rawitscher, 1978	1973	Washington	North Pacific	Troll	Chinook Salmon					87.3		F,O,V
Rawitscher, 1978	1974	Washington	North Pacific	Troll	Chinook Salmon					82.5		F,O,V
Rawitscher, 1978	1973	Washington	Coastal Washingon	Gillnet	Pink Salmon					13.4		F,O,V
Rawitscher, 1978	1974	Washington	Coastal Washingon	Gillnet	Pink Salmon					19.0		F,O,V
Rawitscher, 1978	1973	Maine		Trawl	Perch					7.9		F,O,V
Rawitscher, 1978	1974	Maine		Trawl	Perch					5.5		F,O,V
Rawitscher, 1978	1973	Pacific		Longline	Halibut					50.9		F,G,O,V
Rawitscher, 1978	1974	Pacific		Longline	Halibut					48.1		F,G,O,V
Rawitscher, 1978	1973	Rhode Is.	Offshore New England	Trawl	Flounder					22.1		F,O,V
Rawitscher, 1978	1974	Rhode Is.	Offshore New England	Trawl	Flounder					21.8		F,O,V
Rawitscher, 1978	1974	Rhode Is.	Offshore New England	Trawl	Flounder	62				20.2	1,248	F,O,V
Rawitscher, 1978	1973	Mass.		Trawl	Cod					19.7		F,O,V
Rawitscher, 1978	1974	Mass.		Trawl	Cod					17.9		F,O,V
Rawitscher, 1978	1973	Mass.		Trawl	Haddock					41.6		F,O,V
Rawitscher, 1978	1974	Mass.		Trawl	Haddock					33.7		F,O,V
Rawitscher, 1978	1973	Maine		Traps	Lobster					145.1		F,G,V
Rawitscher, 1978	1974	Maine		Traps	Lobster					141.0		F,G,V
Rawitscher, 1978	1973	Texas		Trawl	Shrimp					269.4		F,G,O,V
Rawitscher, 1978	1974	Texas		Trawl	Shrimp					311.8		F,G,O,V
Rawitscher, 1978	1973	Maryland		Traps	Crab					7.9		F,G,V
Rawitscher, 1978	1974	Maryland		Traps	Crab					9.5		F,G,V

Note: *: F = fuel, G = gear and/or bait, O = other operating inputs (including labour), V = Vessel building and maintenance

Reference	Data From	Based	Fishing Ground	Gear Used	Species Targeted	Annual Landings per vessel (t)	Length (m)	Tonnage	Main HP	Total Energy Inputs (GJ/t)	Total Energy Inputs (GJ/year)	Includes*
		----------------Fishery Characteristics-----------------					--------Vessel Characteristics--------					
Nomura, 1980	1975	Japan	distant water/high seas?	Longline	Tuna			192			21,622	F
Nomura, 1980	1975	Japan	distant water/high seas?	Longline	Tuna	259		229		133.5	34,595	F
Nomura, 1980	1975	Japan	distant water/high seas?	Longline	Tuna			344			37,838	F
Nomura, 1980	1975	Japan	offshore	Longline	Tuna	168		69		83.8	14,054	F
Nomura, 1980	1975	Japan	distant water/high seas?	Pole and line	Skipjack	1,290		284		41.9	54,054	F
Nomura, 1980	1975	Japan	distant water/high seas?	Pole and line	Skipjack			374			43,243	F
Nomura, 1980	1975	Japan	offshore	Pole and line	Skipjack			59			14,414	F
Nomura, 1980	1975	Japan	high seas?	Drift net - 'Mother boat'	Salmon	117		96		68	7,928	F
Nomura, 1980	1975	Japan	offshore?	Drift net - 'Catcher boat'	Salmon	214		65		43.8	9,369	F
Nomura, 1980	1975	Japan	high seas?	Angling/ jigging	Squid	542		300		43.9	23,784	F
Nomura, 1980	1975	Japan	offshore?	Angling/ jigging	Squid	580		99		20	11,604	F
Nomura, 1980	1975	Japan	East China Sea	Bottom trawl?	Various demersal	1,153		114		37.5	43,243	F
Nomura, 1980	1975	Japan	North Pacific	Trawl	Pollack	13,453		349		7.5	100,900	F
Nomura, 1980	1975	Japan	unspecified	Purse Seine	Various pelagic	11,171		111		10	111,712	F
Nomura, 1980	1975	Japan	coastal	Set net-(large)	Various	250		n/a		2.9	721	F

Note: *: F = fuel, G = gear and/or bait, O = other operating inputs (including labour), V = Vessel building and maintenance

		----------------Fishery Characteristics-----------------				Annual	--------Vessel Characteristics--------					
	Data		Fishing	Gear	Species	Landings	Length			Total Energy Inputs		
Reference	From	Based	Ground	Used	Targeted	per vessel (t)	(m)	Tonnage	Main HP	(GJ/t)	(GJ/year)	Includes*
Hopper, 1981	1970's	Unknown	North Sea	Trawl (beam)	Flat fish		20			51.5		F
Hopper, 1981	1970's	UK	Unknown	Trawl	Unknown		<24			34.3		F
Hopper, 1981	1970's	Norway	Unknown	Trawl (stern)	Unknown			>200		25.8		F
Hopper, 1981	1970's	Norway	South Norway	Longline	Unknown		>18			12.9		F
Hopper, 1981	1970's	Scotland	West Scottish coast	Gillnet	Unknown		20			8.6		F
Hopper, 1981	1970's	Norway	North Norway continental shelf	Longline	Unknown		>21			6.9		F
Hopper, 1981	1970's	Norway	Coastal Troms and Finnmark	Longline	Unknown		12.2			5.6		F
Hopper, 1981	1970's	Norway	Coastal Troms and Finnmark	Gillnet & Seine	Unknown					4.3		F
Hopper, 1981	1970's	Norway	Unknown	Purse Seine	Unknown					2.6		F
											(GJ/day)	
Veal et al, 1981	1980	US Gulf coast	Gulf of Mexico	Trawl	Shrimp		19.2		275		19.6	F
Veal et al, 1981	1980	US Gulf coast	Gulf of Mexico	Trawl	Shrimp		25.9		520		53.1	F
Veal et al, 1981	1980	US Gulf coast	Gulf of Mexico	Trawl	Shrimp		22.9		365		36	F

Note: *: F = fuel, G = gear and/or bait, O = other operating inputs (including labour), V = Vessel building and maintenance

How Good is Good?: A Rapid Appraisal Technique for Evaluation of the Sustainability Status of Fisheries of the North Atlantic

Jacqueline Alder, Tony J. Pitcher, David Preikshot, Kristin Kaschner and Bridget Ferriss

Fisheries Centre, University of British Columbia, Vancouver, Canada

Abstract

Sustainability is a key policy requirement for fisheries throughout the world. Until recently it was difficult to assess fisheries sustainability, especially when it required the integration of information on the ecology, as well social and economic aspects. RAPFISH is a new multi-disciplinary rapid appraisal technique for evaluating the comparative sustainability of fisheries based on a large number of easy-to-score attributes. Fisheries may be defined flexibly as entities with a broad scope, such as all the fisheries in a marine gulf, or with narrower scope, such as those in a single jurisdiction, target species, gear type or vessel. A set of fisheries may be compared, or the time trajectories of individual fisheries may be plotted. Evaluation attributes are chosen to reflect sustainability within each discipline and may be refined or substituted as improved information becomes available. Ordinations of sets of attributes are performed using multi-dimensional scaling (MDS), followed by scaling and rotation.

The choice of MDS as an ordination technique is justified. Ordinations are anchored by fixed reference points that simulate the best and worst possible fisheries using extremes of the attribute scores, while other anchors secure the ordination in a second axis normal to the first. Randomly scored reference points act as additional anchors. Monte Carlo simulations provide an indication of the variability of the analysis and therefore reflect how reliable an analysis may be. Sensitivity of each attribute on the final scores is estimated with a step-wise jack-knife procedure.

Separate RAPFISH ordinations are performed in ecological, economic, ethical, social and technological disciplines. Status results expressing sustainability in each of these fields are reported on a scale from zero to 100%. A further evaluation field, measuring compliance with the FAO Code of Conduct for Responsible Fisheries, is itself comprised of six sub-fields that articulate clauses in sections of the Code. Status scores from several fields are combined in kite diagrams to facilitate comparison of fisheries, or fisheries constructed to represent alternative policies. In this paper the method is applied to present day fisheries and some historical time series from the Gulf of Maine (39 fisheries) and the North Sea (77 fisheries). The results, which are compared with previous work from Newfoundland (19 fisheries), provide examples of the use of RAPFISH in a multidisciplinary evaluation of the sustainability component of the impacts of fisheries on marine systems, and in assessing compliance with the FAO Code of Conduct.

Introduction

This paper applies a recently developed rapid appraisal technique, RAPFISH, to evaluate sustainability of North Atlantic fisheries. The technique is scaleable and hierarchical and provides simple percentage scores, with their confidence limits, for fisheries entities that may be flexibly defined in space and time. In addition, RAPFISH may be used to score compliance with the FAO Code of Conduct for Responsible Fisheries.

RAPFISH relies upon ordination of scored attributes grouped in a number of evaluation fields. The fields cover ecological, economic, social, ethical and technological sustainability. We present a rationale for measuring sustainability in this way, and a full statistical justification of the numerical engine at the heart of the RAPFISH technique together with numerical methods of expressing the relative influence of attributes and uncertainty in the results.

In addition to a rigorous examination of the method, our aim in this paper is to compare RAPFISH analyses of selected fisheries from the North Atlantic (from the Gulf of Maine, North Sea, Canada) to show how interdisciplinary evaluations may be used to inform policy choices.

Rationale and Methods for Rapfish

The Concept of Sustainability

The 1990s saw a decade of change where fisheries management imperatives of maximising production and economic returns were replaced with managing for sustainability. This change was the product of a number of factors:

- Increasing environmental awareness amongst diverse stakeholders, reflected in events such as the Rio Earth Summit that highlighted the global need for improved management of natural resources including marine resources and instruments such as the Agreement for the Implementation of the Provisions of the United Nations Convention on the Law of the Sea of 10 December 1982 Relating to the Conservation and Management of Straddling Fish Stocks and Highly Migratory Fish Stocks;

- The collapse of a number of major fisheries, that highlighted not just the ecological but also the social and economic consequences of not managing for sustainability; and
- Empowerment of stakeholders, commercial and recreational fishers, as well as conservation groups who demanded a broader view of fisheries management

This change was reflected in political arenas where national and state fisheries legislation and policy was amended or rewritten to embrace the notion of sustainability (e.g. USA (Magnuson Act amendments), Canada (Oceans Act), European Union (Common Fisheries Policy proposed changes)). Some countries went even further to embrace the concept of ecosystem management (Australia). This worldwide change in the imperative has challenged conventional approaches to fisheries management. Until recently, prevailing approaches to assessing the sustainability of exploited marine species focused on determining the stock status of the target species relative to biological and in some cases ecological, reference points such as levels of fishing mortality, spawning biomass, or age structure (Smith 1993). Resource managers used these reference and target points, as indicators of the status of a resource and as early warning signs of exceeding target extraction levels.

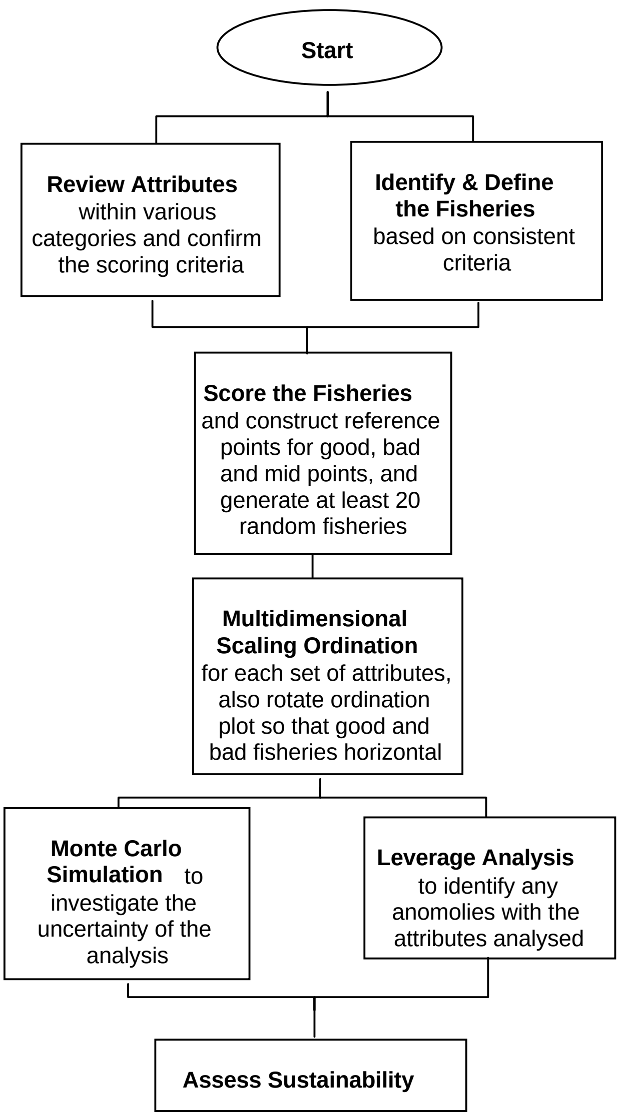


Figure 1 Elements of the process of applying RAPFISH to fisheries data.

These approaches, however, require substantial information, independent surveys and complex models to estimate past and present reference points representing management objectives for fisheries. The inherent uncertainty in fisheries research limits the ability of these complex models to estimate the sustainability indicators with a high degree of certainty (Walters 1998). The requirement for reliable data, complex models and widely-educated resources managers further limits less developed countries from assessing their fisheries with precision or accuracy. Conventional stock assessment approaches focus on the biological outcomes for single species and on the odd occasion ecological or economic issues. They do not, therefore, address adequately the question of sustainability.

The notion of sustainability is hotly debated amongst the community and there is no single agreed definition of what sustainability means (Buckingham-Hatfield and Evans 1996). There is some common ground in its meaning in that it is a multidisciplinary concept and therefore must include social and economic dimensions (Buckingham-Hatfield and Evans 1996). This debate is also found in fisheries management. Indeed, in assessing fisheries management regimes the ecological, social and economic consequences as well as technological and ethical outcomes need to be considered (McGoodwin 1990). The challenge for fisheries managers will be to assess the sustainability of fisheries using multidisciplinary approaches that integrate these diverse topics. RAPFISH is one such technique. It is a new multi-disciplinary rapid appraisal technique based on multivariate statistics that can be used to assess the sustainability of fisheries (Figure 1).

The Rapfish Technique

The RAPFISH technique uses simple, easily scored attributes from a range of disciplines to provide a rapid and cost effective appraisal of the sustainability of fisheries (Pitcher et al. 1998a) and compliance with the FAO Code of Conduct. The attributes are defined to reflect uncorrelated and discrete aspects of sustainability. The technique also provides managers with a considerable flexibility in defining fisheries, from a broadly defined geographically based fishery (e.g. North Sea Herring

Fishery 1990) to a fishery defined by its geographic range, target species, vessel type, and gear (e.g. Gulf of Maine Herring Sail Trawl Fishery 1890s). Fisheries should be defined at a scale such that impacts of changes in management or fishing practices can be identified. The inherent flexibility of the technique allows sets of fisheries or individual fisheries to be compared, or the trends of individual fisheries through time may be analysed (Pitcher 1999). Note that attributes should be fixed if cross-analysis comparisons are to be made.

The technique is based on a statistical ordination engine. The most appropriate method found to date is multidimensional scaling (MDS), a multivariate ordination method. MDS is used to construct a 'map' showing the relationships between a number of objects based on a table of distances between the objects (Manly 1994). The map can be in one or more dimensions, however, dimensions greater than three are difficult to visually represent and to interpret. In the case of RAPFISH, the individual fisheries are the objects, and their relative positions are based on the attribute scores from the various disciplines. A common set of attributes are scored for each fishery using a scale and for each attribute that is consistent among the different fisheries. Where a fishery is located on the map is only indicative of its relative sustainability compared to other fisheries analysed. Thus, there is a need that the attributes are representative of the objective of sustainability. In addition to the MDS algorithm that is used to place the fishery on the map, the attributes and the relevant scores are another key feature of RAPFISH and are discussed below.

Appendix 1 provides a full account of the choice of MDS as the statistical ordination engine for RAPFISH ordinations together with a discussion of methods of addressing uncertainty and the sensitivity of individual attributes.

Applying MDS in RAPFISH

The RAPFISH method used to assess the sustainability of a group of fisheries is outlined in Figures 1 and 2. The initial steps of defining the attributes and scores, identifying and scoring the fisheries are detailed in Pitcher (1999). These steps are summarised below.

Defining the Fisheries

In a RAPFISH analysis there is considerable flexibility in defining the fisheries. The definition can be based on a range of criteria including spatial, temporal, technological, anthropological and political measures. The choice of criteria applied to the fisheries do not affect the results, provided that roughly the same criteria are chosen for the fisheries to be compared and that each fishery is independent of the others. There is also scope within the RAPFISH technique for a hierarchical analysis since scores from groups of fisheries from a statistical region, ecosystem, gear type, or vessel type can be collapsed (Pitcher 1999).

Once the fisheries are scored, four 'reference fisheries' are constructed to act as anchor points for each evaluation field. One represents the ideal (or 'good') fishery, in which all attributes are scored in line with maximising sustainability characteristics

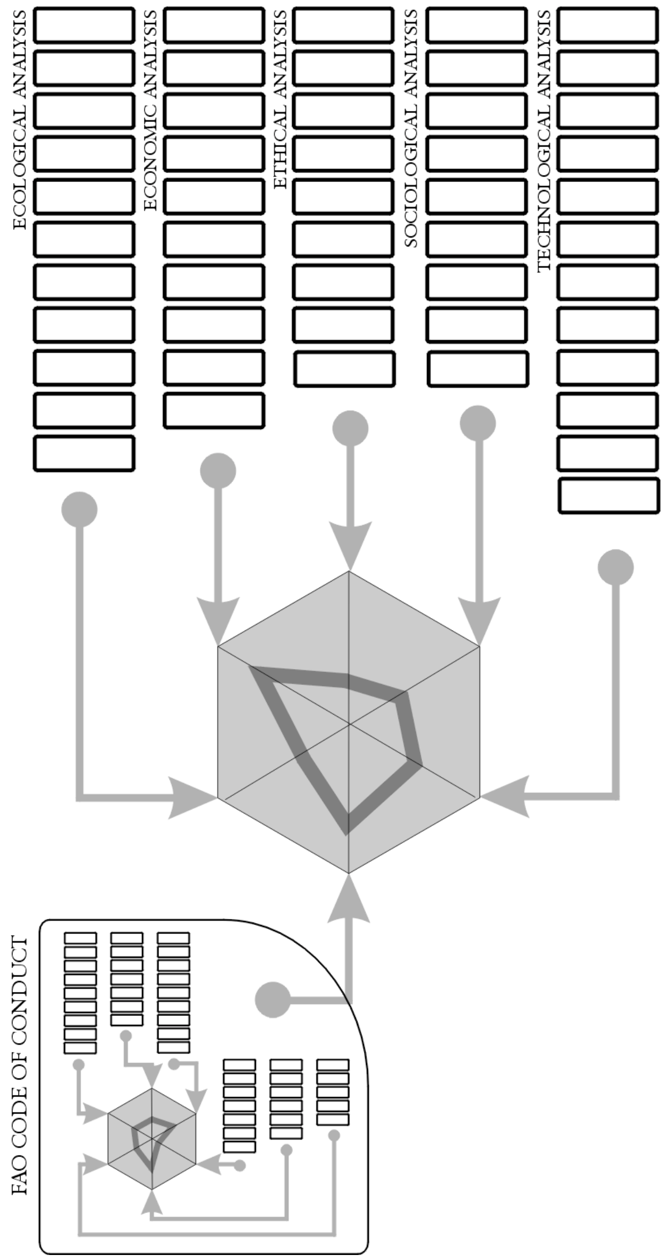


Figure 2. The RAPFISH procedure using a multidisciplinary kite to express sustainability. Boxes represent the attributes used to ordinate fisheries in each evaluation field. Kite apices represent a score between 0% = 'bad' (kite centre) and 100% = 'good' (the outer rim) from each field. Six evaluation fields are illustrated here, one of which, for the Code of Conduct, is comprised hierarchically of a five-field RAPFISH.

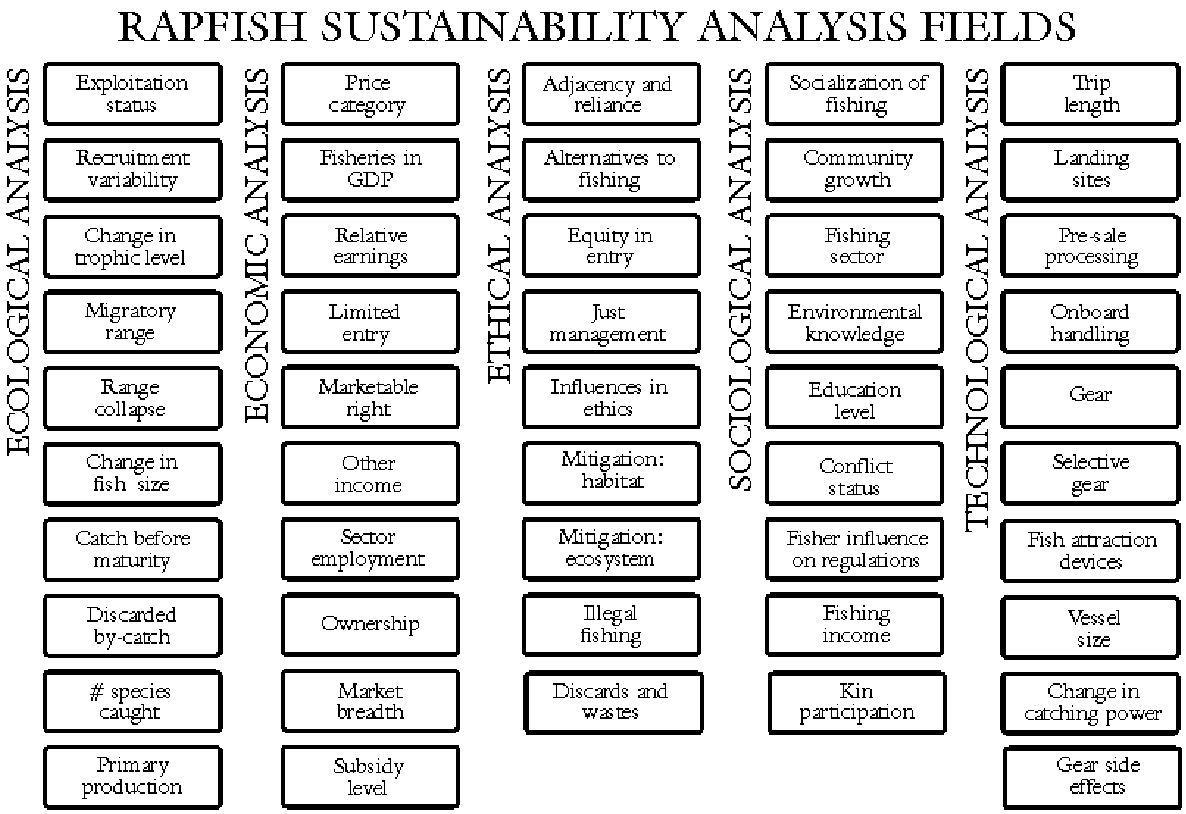


Figure 3. Showing the attributes within the five evaluation fields for assessing sustainability using the RAPFISH technique (attributes sets as of May 2000).

within each discipline. Another fishery representing a modelled 'bad' case is created in which all the attributes are scored as a worst-case scenario. These two extremes provide reference points with which to compare the sustainability scores of other fisheries. The other two anchor 'fisheries' are fixed reference points constructed from two halfway scores, these fisheries stabilising the vertical dimension of the plot against flipping (see Pitcher 1999 and Appendix 1). The addition of these points constrains the ordination so that points in the ordination can be compared to known good and bad points. Further research is under way to define a set of hypothesized points throughout the ordination so that comparisons can be made between analyses containing the same hypothesized points.

Random fisheries may also be added to increase the number of cases if the input data is small, i.e., if the number of fisheries is less than four times the number of attributes. These random fisheries help to avoid degenerate solutions. Many statistical packages can be used to generate fisheries scores (normalised) using a normal distribution with mean = 0 and standard deviation = 1. Constraints on the number of cases analysed in the MDS limit the total number of fisheries that can be analysed by SPSS to approximately 100; similar constraints apply to statistical packages. Further analysis, such as Monte Carlo simulations and sensitivity analysis, however, do not include these random fisheries. It is hoped eventually to replace these random fisheries with fixed anchor points to reduce flipping and improve the ability to overlay subsequent analyses. A regular grid of anchor points is envisaged (see Appendix 1).

Defining and Scoring Attributes

Work by Pitcher and Preikshot (2000) and others provides a well-developed set of sustainability attributes to assess a fishery with respect to its ecological, technological, social, economic and ethical characteristics (Figure 3). The individual attributes have been established through an iterative process with experts over the last three years. Attributes were also selected because they best measure and discriminate the objective of sustainability within an evaluation field. Pitcher (1999) using simulated fisheries and Preikshot (2000) using cluster analysis verified that the attributes used are reflecting the notion of sustainability. Sets of attributes have been added recently to encompass compliance with the FAO Code of Conduct (Pitcher 1999). These two sets of attributes are used in the *Sea Around Us Project* (SAUP) so that comparisons can be made between countries and fisheries. All attributes may not meet all situations outside of the SAUP resulting in the need for minor modifications and to ensure that attributes cover those aspects of the system that the stakeholders perceive to be important. Changes can be made without compromising the rigor of the technique, however, changes must be carefully

Table 1: An example of an input matrix of fisheries scores for ecological attributes. Labels in column 1 are codes used to refer to fisheries in one of the analyses.

	Ecological Attributes							
Fishery	Exploit	Recruit-ment	Catch	Trophic Change	Primary Product	By-Catch	Gear	Environ. Impacts
GoM_L89	0	2	5	3	2	4	3	0
GoM_S12	2	1	3	1	0	4	2	3
GoM_P56								

considered because experience has shown that defining attributes to reflect sustainability is not easy. If a number of changes are necessary it may be more appropriate to create a new evaluation field that suits users' specific needs.

If changes are necessary, they should be made in light of need to maintain the following attribute properties.

- Attributes within each set reflect the notion of sustainability, with sustainability meaning that the resource and its fishery can continue beyond the short term;
- The attributes are chosen for their ease of scoring and objectivity, including assigning 'good' and 'bad' to the extreme values in relation to sustainability for each attribute;
- The attributes are available to all fisheries through all time periods in the analysis (Pitcher 1999); and
- A number of attributes are used so that the discriminating power of the ordination method in MDS is maximised with three times as many fisheries as attributes used to ordinate them (Kruskal and Wish 1978; Stalans 1995).

The current list of attributes and their corresponding scores are shown in Appendix 2 (and an up-to-date list is maintained at www.fisheries.ubc.ca)

MDS Analysis – scoring attributes

The application of MDS in RAPFISH assists fisheries managers to evaluate the sustainability of fisheries past and present and to make comparisons between different types of fisheries as well as assessing national compliance with the FAO Code of Conduct. These differences or comparisons can be measured as distances between various fisheries in a multidimensional space that is defined by a range of attributes scored on an interval scale. As set out in Appendix 1, MDS is an appropriate multivariate method to evaluate these distances. Other multivariate methods, such as Cluster Analysis, Factor Analysis, Principle Components Analysis, Correspondence Analysis, are available but are not as appropriate as MDS in assessing fisheries sustainability, as discussed in Appendix 1.

In the RAPFISH analysis, each fishery is scored on several attributes, the scores generally range between 0 and 5. The result is a rectangular matrix with I rows representing fisheries and J columns representing the attribute scores (Table 1). The data within the matrix is interval since the extremes of the scoring scale represent good and bad. The scores vary with attributes having maximum values between 3 and 5. The scores need to be normalised to minimise the stress (Davison 1983) and to ensure that the assumption of monotonicity is not compromised (Pitcher and Preikshot 2000). This assumption was validated in RAPFISH by Pitcher (1999) when normalised scores are used with squared Euclidean distances in a metric MDS analysis.

One approach to standardising the scores is to convert them to normalised (Z) values:

$$Z = (x - \mu)/\sigma \qquad \text{...1)}$$

However, at this stage the normalised data do not express the distances between the fisheries. The squared Euclidean distances can be calculated using standard computer programs such as PROXIMITY in SPSS and DIST in SPLUS2000. The resulting distance matrix is used as input into the MDS analysis. Programs such as ALSCAL (Yound and Lewyckyj (1979) as cited in Manly 1994) and KYST (Kruskal, Young and Seery (1973) as cited in Carroll and Arabie (1998)) iteratively search for the best fit of the points in the specified dimension. These programs generally require additional parameters that specify whether the analysis is conditional or not, the scaling model (e.g. ASCAL an asymmetric Euclidean distance model), the data type, the number of iterations, the convergence and minimum stress levels, and minimum and maximum number of dimensions in the solution. The corresponding output can provide plots of the coordinates, matrix weights and coordinate weights, as well as a matrix of the coordinates of the points. The resulting dimensions are rotated and re-scaled for ease of interpretation.

Table 2. Example of a Monte Carlo data matrix.

			Dimension One			Dimension Two		
MC1 RUN	X1	X2	X3......	X_I	Y1	Y2	Y3......	Y_I
1								
2								
...								
50								

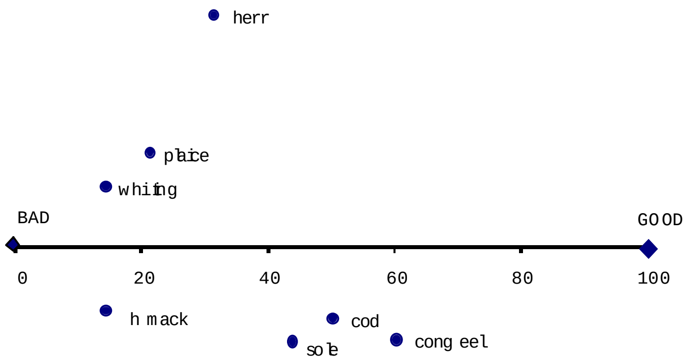


Figure 4. An example of the rotation to ensure the 'bad – good' axis is horizontal. Labels refer to fisheries in one of the analyses.

MDS Analysis – Rotation

The first dimension is rotated so that it is horizontal with the good (90°) and bad fisheries (270°) at either end of the axis. The first dimension can be re-scaled to percent with 0% as bad and 100% as good, so that the relative position of the fisheries can be assessed (Figure 4). Although the first axis is expressed as percent, calculating the % differences between points from two different ordination analysis is not valid because at this time the good and bad points are not necessarily the same points in the two analyses. The development of hypothesised reference points that would allow for comparing points from different ordination analysis (but with the same hypothesised points) to be compared is currently underway. established

MDS Analysis – computer package

RAPFISH currently uses the statistical package SPSS that contains the ALSCAL program. The SPPS package is used because:

- it handles a range of data types and MDS models including metric and nonmetric through the ALSCAL program;
- in two dimensions, ALSCAL is stable and meets most of the assumptions for MDS;
- widely available – most research institutions have access to the package;
- handles missing values;
- allows ties; and
- has a command language that can be used for complementary analysis.

MDS programs can be found in statistical packages such as SAS and SYSTAT, however, many are not as flexible in handling missing values or ties (Young and Hammer 1987).

The SPSS the programs PROXIMITY and ALSCAL are combined into one SPSS procedure (BatchRap) in the Rapfish analysis. The raw data are normalised prior to input into the PROXIMITY program. The PROXIMITY routine is only used to calculate the matrix of the Squared Euclidean distances as input into ALSCAL. The options selected in ALSCAL include using an Euclidean Model and setting the limits for convergences, stress and number of iterations, as well as specifying the number of dimension in the solution. The SPSS output provides the iteration history, the stress value, squared correlation coefficient, the coordinates

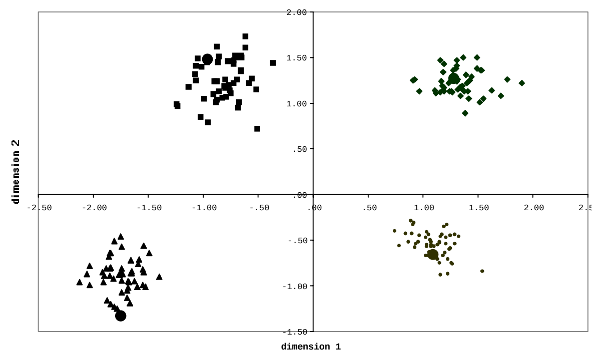


Figure 5. Distribution of points generated by Monte Carlo simulation for four different fisheries (the large dot represents the original point).

in two dimensions and plots of un-rotated and rotated fisheries (Appendix 3). The program also writes the information to files so that dimensions can be re-scaled (e.g. expressing the first dimension, sustainability, as a percent) or used for further analysis (e.g. Monte Carlo simulations or sensitivity analysis).

The ALSCAL program can not be used to derive parametric variables directly and therefore the confidence limits of the estimates can not be estimated. Monte Carlo simulations, however, are used in the SPSS analysis. The Monte Carlo procedure (MC1) perturbs each normalised fisheries score for each attribute by a random amount selected from a Gaussian distribution with mean

Table 3. Example of a sensitivity analysis data matrix.
.

				Dimension One			Dimension Two	
Attribute Removed	**X1**	**X2**	**X3 ....**	**X_I**	**Y1**	**Y2**	**Y3 ....**	**Y_I**
1								
2								
...								
J								

equal to zero and variance equal to 1. These perturbed values are used to estimate the distance matrix and undertake the MDS analysis. A minimum of 50 simulations are run producing a 50 x (2xI) (I = the number of fisheries) matrix of coordinates (Table 2). Examination of the points indicates that they are not normally distributed about the mean (Figure 5); consequently the median values are plotted with their 95% confidence intervals. The distribution of the MC points is influenced by the ALSCAL program that "scales the configuration so that the average coordinate is zero in all dimensions and the sum of the squared coordinates is equal to the number of objects multiplied by the number of dimensions" (Manly 1994, p. 175). The use of jackknife and bootstrap methods to estimate confidence intervals may be possible, but requires further investigation at this stage.

A jackknife procedure (MC3) can be used in RAPFISH to investigate the sensitivity of the attributes. The jackknife method is used to generate the MDS analysis using J-1 attributes for each analysis. The procedure generates a J x (2xI) matrix (Table 3) which is used to explore the sensitivity of the analysis to specific attributes. The standard error of the squared differences between the original points and the re-sampled points can be used to compare attributes (Pitcher 1999).

Presentation of RAPFISH results

In general we have five ways of presenting the results of RAPFISH ordinations (Pitcher 1999). First, two dimensional ordination plots (as in Figure 6 below) provide the most detailed information; other presentations lose the information about the vertical position in the ordination representing differences that are not related to sustainability (or compliance with the FAO Code of Conduct). Secondly, RAPFISH scores along the 'bad' to 'good' axis may be compared using bar charts. Bar charts swung vertically and drawn to the left and right of a vertical line enables comparison between two sets of fisheries.

Thirdly, time trajectories used to assess changes in sustainability graph RAPFISH scores against time. Fourthly, rank orders may replace actual RAPFISH scores, and attention may be drawn to fisheries falling into the upper and lower quartiles, so that rank orders in different RAPFISH evaluation fields may be compared.

Finally, a convenient way to represent scores on the different axes of sustainability is a polygonal kite diagram (e.g. Figure 8). Each axis represents one RAPFISH evaluation field. For each of the axes, a score of zero (0%) lies at the centre and a score of 100% lies on the rim of the polygon. For two- or three-way comparisons, the kite provides a simple visual representation, but more complex simultaneous comparisons produce muddled pictures. Figure 8 illustrates how scores from six fields go to make up the points of a kite. Comparison made with the kite may be of individual fisheries, or gear types, or large- and small-scale sectors, or fisheries for a certain species, or date. Kite diagrams can be used to present a hierarchy of RAPFISH analyses, as described later.

CASE STUDIES

Three case studies, the Gulf of Maine Fisheries, the German Fisheries and the United Kingdom Commercial fisheries were subjected to a multidisciplinary RAPFISH analysis. Only the present day fisheries were scored for compliance with the FAO Code of Conduct for Responsible Fishing, since most of the fisheries predate the introduction of the Code. These three case studies include historical fisheries as well as present fisheries to demonstrate the range of Rapfish applications. Details of the historical development of these fisheries are outlined in Appendices 4, 6, 8 and 10 and details the scores and the MDS results are listed in Appendices 7, 9 and 11. A fourth case study of the East Coast of Canada previously analysed by Melanie Power, Tony Pitcher Mary Gregory (*in prep*) was included in this report to provide for a comparison of present fisheries on both sides of the Atlantic.

(A) GULF OF MAINE FISHERIES

Fifteen defined fisheries from the Gulf of Maine (Appendix 4), including three with historical time series that span almost the entire period of European colonisation were analysed to investigate uncertainty and the sensitivity of attributes. Monte Carlo simulation examined the influence of uncertainty in attribute scoring, and jacknife resampling determined the relative influence of individual attributes.

Table 4. Stress and RSQ values for the Gulf of Maine RAPFISH ordinations, and Monte Carlo and sensitivity analysis over the five evaluation fields.

	Original Analysis		Monte Carlo		SensitivityAnalysis	
Evaluation Field	STRESS1	RSQ	STRESS1	RSQ	STRESS1	RSQ
Ecology	0.284	0.722	0.276 - 0.290	0.708 - 0.736	0.263 - 0.286	0.657 - 0.774
Economics	0.272	0.743	0.297 - 0.279	0.726 - 0.744	0.262 - 0.279	0.551 - 0.775
Social	0.281	0.671	0.273 - 0.293	0.640 - 0.686	0.265 - 0.294	0.647 - 0.706
Technological	0.277	0.656	0.272 - 0.285	0.633 - 0.670	0.269 - 0.294	0.624 - 0.686
Ethical	0.273	0.728	0.270 - 0.281	0.728 - 0.732	0.267 - 0.281	0.755 - 0.738

RAPFISH Analysis

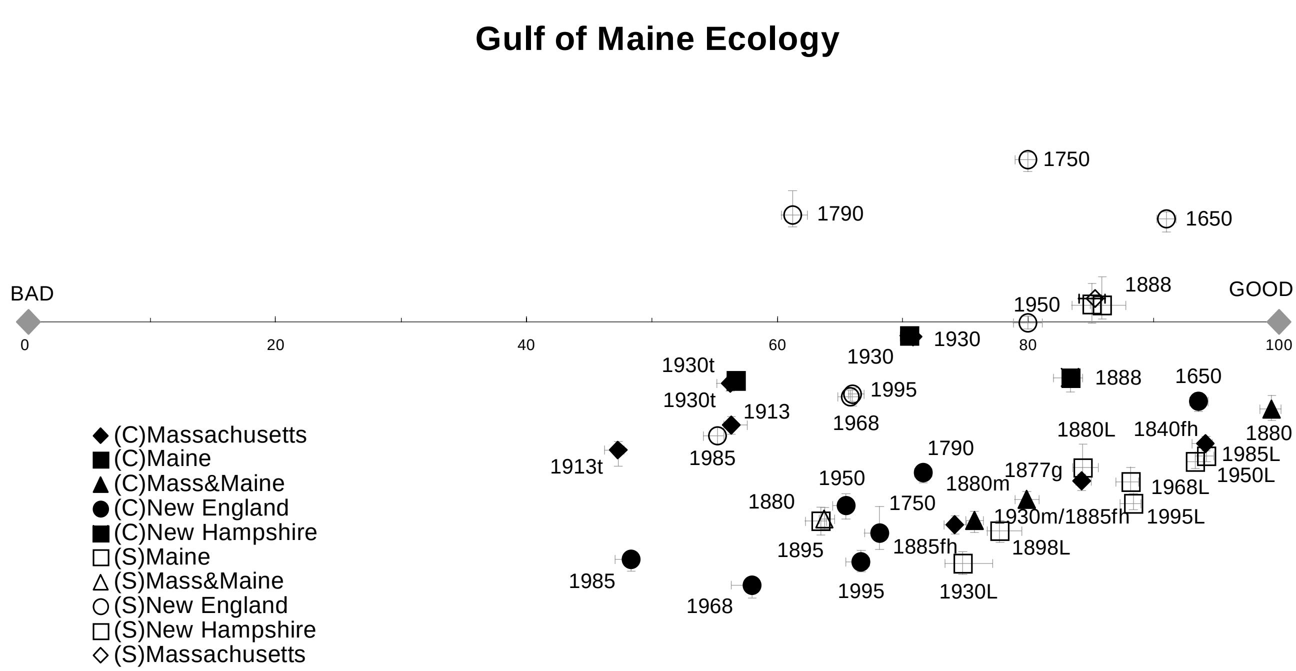
The Gulf of Maine fisheries were coded (Appendix 6) and analysed (Appendix 7) for the five evaluation fields of ecology, economics, social, technological and ethical as defined in Pitcher (1999). RAPFISH ordination with Monte Carlo error simulations and sensitivity analysis as described above were used to explore the uncertainty associated with the scores.

Two-D Rapfish ordination plots were constructed for all five evaluation fields. The scores in two dimensions were obtained within four iterations for all attribute sets. The initial STRESS1 values of all attribute sets ranged between 0.272 and 0.284 (Table 4). Although these stress values are high by statistical standards, they are considered acceptable given the high degree of measurement or sampling (scoring) error associated with the case study. Furthermore, the RSQ values that measure the proportion of variance of the disparities in the data that is accounted for by the corresponding distance, range between 0.734 and 0.656; these are considered acceptable values. In addition, Monte Carlo simulations (50 runs) and sensitivity analysis were also conducted to explore the uncertainty of this analysis.

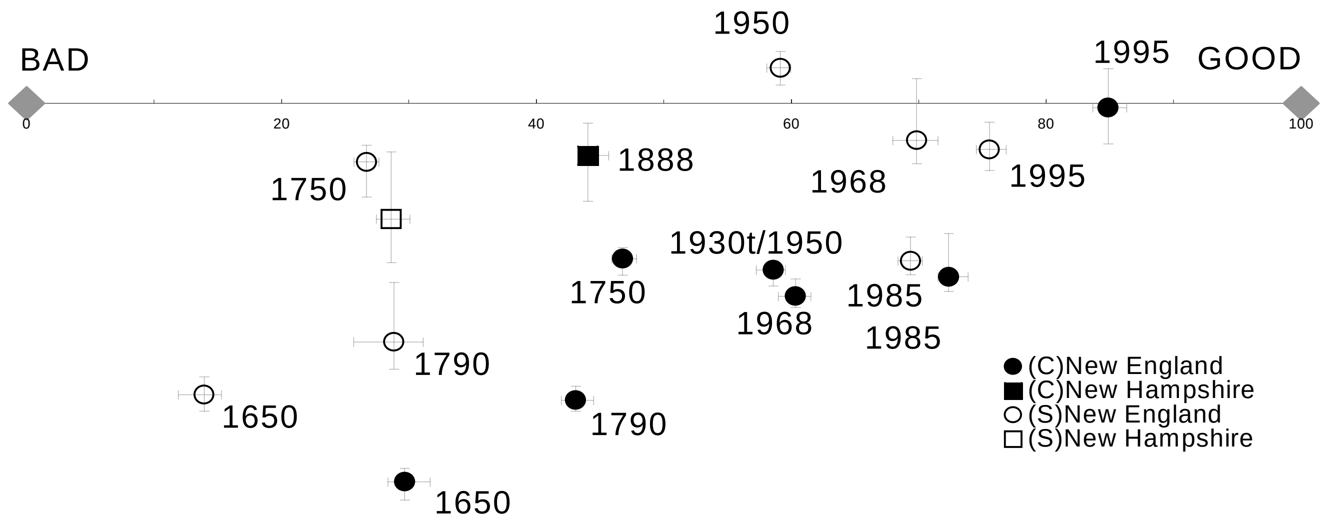
The median values of the scores generated in the Monte Carlo runs and their 95% confidence intervals were plotted to explore the uncertainty of the results (Figure 6). The 95% confidence intervals are small and generally within 5% of the median sustainability scores with the exception of economic attributes where the limit exceeded 5% for one fishery. The confidence intervals in the second dimension, however, are much higher as expected since the second dimension accounts for the non-sustainability information in fisheries. Because the variation is small in the sustainability scores the separation of some fisheries is clearly evident as is the clustering of similar scores.

The 2-D RAPFISH ordination plots (Figure 6) clearly indicate that sustainability scores of the fisheries vary significantly through time and between the fisheries for all evaluation fields (Figure 6). The analysis has shown that historical fisheries as far back as the 1650s were not necessarily sustainable and that some modern day fisheries are indeed more sustainable than in the past, while others have become less sustainable than their historical counterparts. A plot of the 95 confidence intervals of the median values for the 38 fisheries (Figure 6a) makes interpretation of the plot difficult and therefore in such cases subsets (Figures 6b to 6e) can be plotted making interpretation easier and analysis of trends easier.

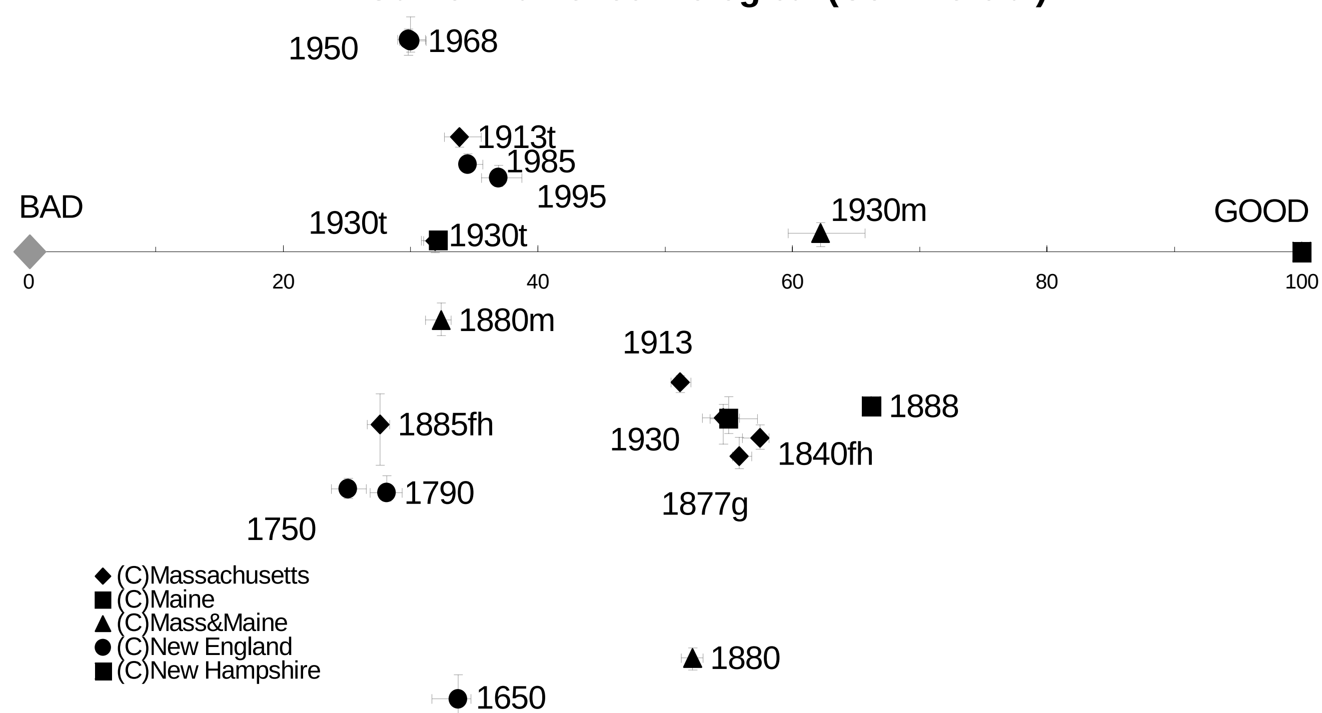
The standard error of the sustainability scores when a single attribute was omitted from the analysis was used to explore the importance of attributes to the analysis. As in other RAPFISH studies (Pitcher 1999; Pitcher and Preikshot 2000), the standard error was less that 14% for any single attribute (Figure 7). The highest standard error (14%) occurred when ownership within the fishery was omitted. This is because the attribute ownership separated out only four fisheries: the 1650 fisheries were foreign owned and two trawl fisheries were joint venture operations, while the remaining fisheries were owned locally. The standard error when the number of species caught was eliminated is also high (12%) due to the division of the fisheries into the small scale sector catching many species and the commercial sector catching only a few species. Similarly, 12% standard error when fishing income is dropped is due to a separation of income based on whether it is a small-scale inshore fishery or larger commercial offshore fishery.

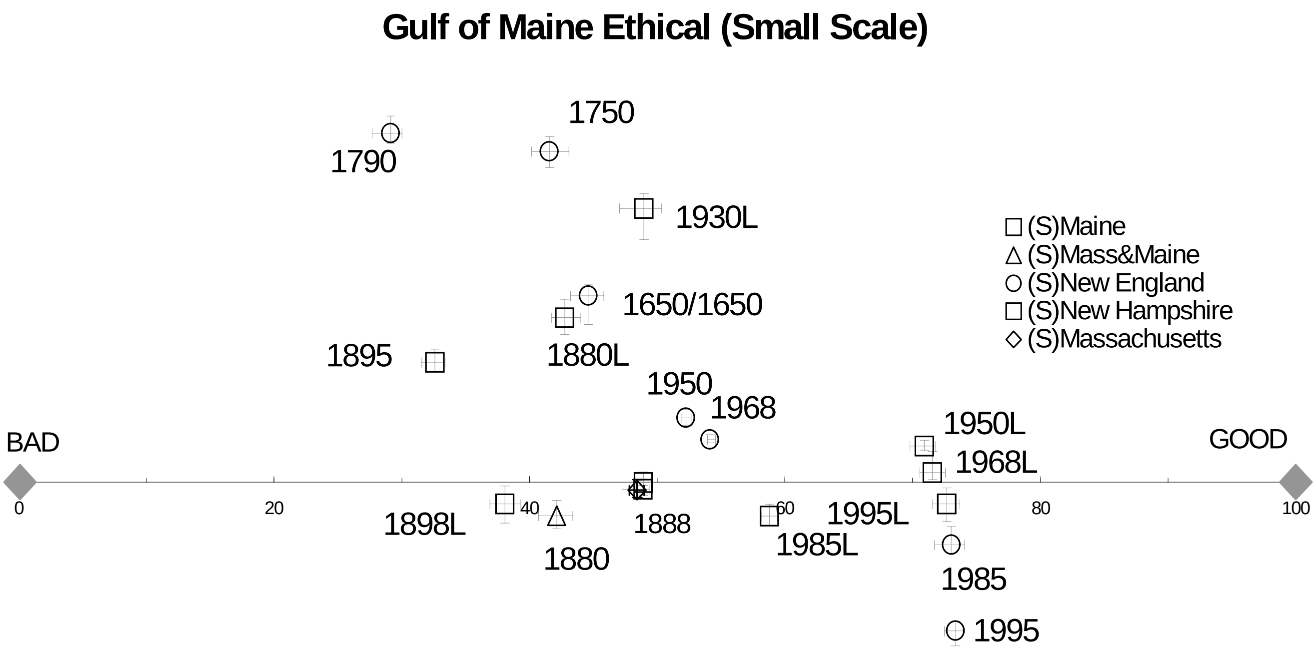


Gulf of Maine Social



Gulf of MaineTechnological (Commercial)





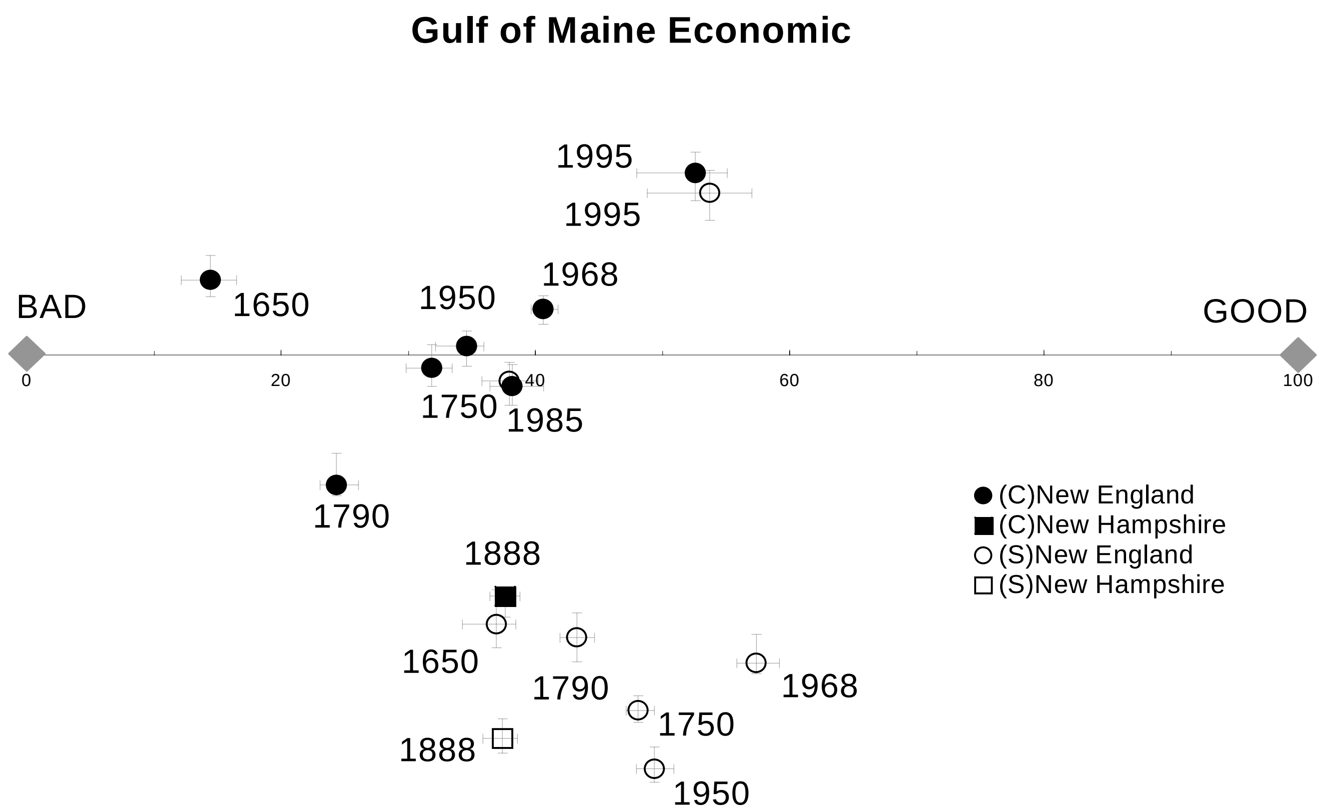


Figure 6. Two-dimensional RAPFISH ordinations, in five labelled evaluation fields, of the Gulf of Maine fisheries listed in Table 5. Symbols represent fisheries and numbers show time periods. Bars indicate upper and lower 95% confidence intervals for median values from 50 Monte Carlo simulations. Note that only the ecological plot contains the full 38 fisheries, the other four are subsets of the full fisheries

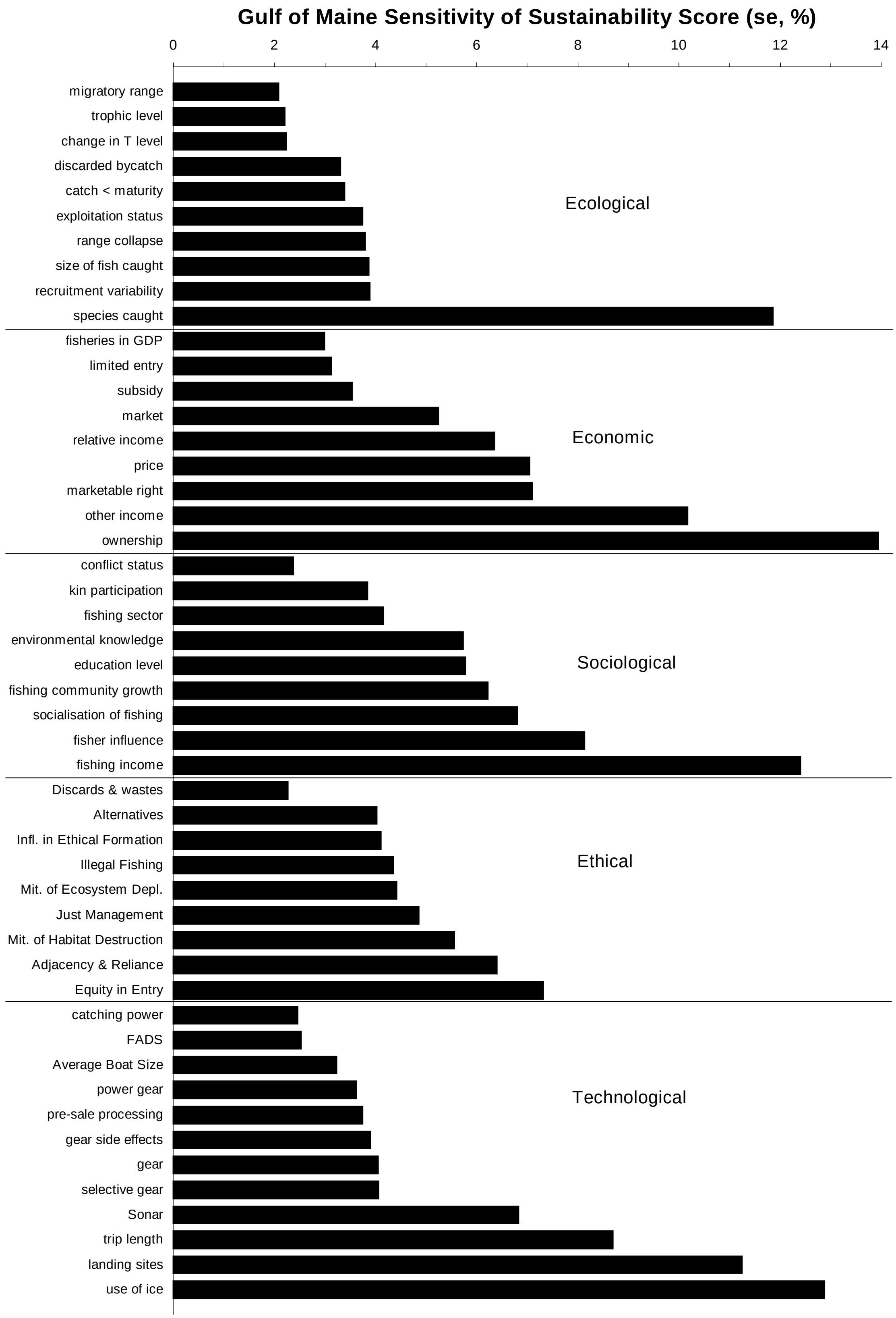


Figure 7. Gulf of Maine fisheries. Leverage (percentage influence on median) of individual attributes for five evaluation fields calculated from sensitivity analysis described in the text.

(B) East-West Analysis of North Atlantic Fisheries

The scores twelve previously defined fisheries from the present day from the Gulf of Maine, the United Kingdom and Germany and the scores of 19 fisheries from a Canadian study (Melanie Power, et al, *in prep.*) were combined to illustrate how Rapfish can be used to compare fisheries on both sides of the North Atlantic. These studies provided 22 fisheries on the west side and 9 fisheries on the east side of the Atlantic. This analysis is only for illustrative purposes since a) the above two cases studies and the Canadian East Coast study did not use identical attributes in all evaluation field and b) only four nations are included in the study and therefore the results are not necessarily representative of the entire area especially for the United States where only three fisheries from the Gulf of Maine are included in the analysis. Nevertheless there is sufficient information to illustrate the capabilities of Rapfish to undertake a more complete study of North Atlantic fisheries as part of the *Sea Around Us* Project.

This case study is divided into a general comparison of the evaluation fields for the four fisheries, followed by an analysis of compliance with the FAO Code of Conduct for Responsible Fisheries.

General Comparison

Scores in the five Rapfish evaluation fields (ecology, economic, social, technical and ethical) were combined with an overall Code of Conduct for Responsible Fisheries for the above case studies to examine differences in scores between fisheries on either side of the North Atlantic and differences between nations.

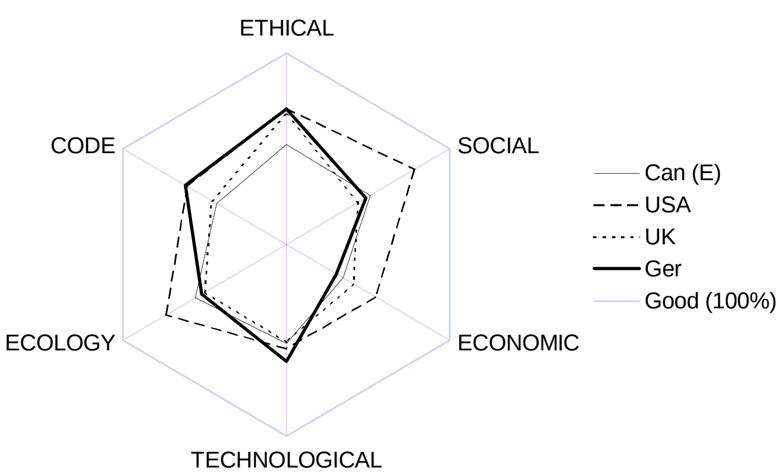


Figure 8. Multidisciplinary kite diagram expressing the North Atlantic Fisheries sustainability scores by country. The outer rim represents 100% = (good).

The kite (Figure 8) which expresses the average scores for the four nations studied shows the sustainability scores are highly variable between nations and evaluation fields. The USA (Gulf of Maine) had the highest average scores in 4 of the 6 evaluation fields. This result, however, represents only three fisheries from the Gulf of Maine. There were few differences between the average scores for technological sustainability that ranged between 50% and 60%. The average social sustainability score for the Gulf of Maine fisheries was substantially higher than for other nations. Average ethical scores were often the highest for all nations. The difference between the average ethical scores for German and United Kingdom fisheries was less than 2%, however, the difference between average scores for Canada and the Gulf of Maine was 19%. Economic sustainability scores were lower than for most other evaluation fields and the average scores was particularly low for German fisheries (30%) but the average Canadian score was only 3% higher. The highest average ecological sustainability score was for the Gulf of Maine (73%), overall scores were above 50% for the other three nations. Average scores for the combined Code of Conduct varied between the four nations with German and Gulf of Maine fisheries having higher scores than Canada and the United Kingdom.

Ethical

A ranking table (Table 5) used to explore ethical sustainability and shows clear differences between east and west fisheries in the North Atlantic. In the upper quartile 5 of the 8 fisheries were from the east while in the lower quartile all fisheries were from the west. Ethical scores from the east were all above the 50 percentile. Also in the upper quartile fisheries with high ethical scores were also scored high for code compliance. Similarly, fisheries that scored low for code compliance were often the in the lower quartile for ethical sustainability. This strong link between ethical sustainability scores and code compliance is also reflected in a table of correlation coefficients (Table 6).

Table 5. Rank orders(1 = highest rank) of the North Atlantic fisheries in the economic sustainability ordination, alongside rank orders from other evaluation fields. W = west and E = east side of the Atlantic.

	Evaluation Fields						
RANK ORDER	Econ.	Eth.	Ecol.	Soc.	Tech.	*Code*	Atlantic
Bay of Fundy (W)	1	3	14	6	6	2	W
German demersal	2	21	31	11	2	4	E
Gulf of Maine inshore	3	9	5	3	22	14	W
German herring	4	13	20	12	5	3	E
Gulf of Maine lobster	5	2	1	5	3	1	W
German shrimp	6	11	17	22	13	9	E
UK plaice	7	29	13	25	16	16	E
UK haddock	8	19	8	24	17	13	E
UK herring	9	12	16	23	14	15	E
Gulf of Maine trawl	10	8	4	1	26	10	W
German Cod	11	22	22	19	15	5	E
German mussel	12	25	15	21	25	11	E
Cod longline	13	24	26	16	18	22	W
Cod handline	14	23	27	14	1	26	W
UK cod	15	20	12	17	19	17	E
Snow Crab 19	16	16	2	2	12	6	W
Scallop	17	17	9	10	27	29	W
Cod trap	18	27	25	20	4	25	W
Cod inshore	19	26	29	28	10	27	W
Lobster (Ding)	20	18	7	18	8	7	W
Lobster	21	15	10	9	7	8	W
Shrimp (ES)	22	1	3	13	28	31	W
Snow Crab	23	10	11	7	9	20	W
Bay of Fundy (S)	24	4	23	26	21	12	W
Mackerel (Din)	25	5	24	4	24	19	W
Capelin	26	14	21	8	23	21	W
Mackerel (At)	27	7	30	15	20	18	W
Shrimp (N)	28	6	6	27	31	30	W
Cod gillnet	29	28	28	29	11	28	W
Cod trawl	30	30	18	30	30	23	W
Cod offshore	31	31	19	31	29	24	W

Table 6. Correlations among rank orders of 31 North Atlantic fisheries analysed by Rapfish in six fields. Shaded cells are non significant at the 5% level. (Spearman non-parametric correlations).

Economic	0.41				
Social	0.57	0.36			
Technological	-0.06	-0.23	0.21		
Ethical	0.15	0.18	0.36	0.46	
Code	0.27	0.21	0.40	0.46	0.65
	Ecology	Economic	Social	Technological	Ethical

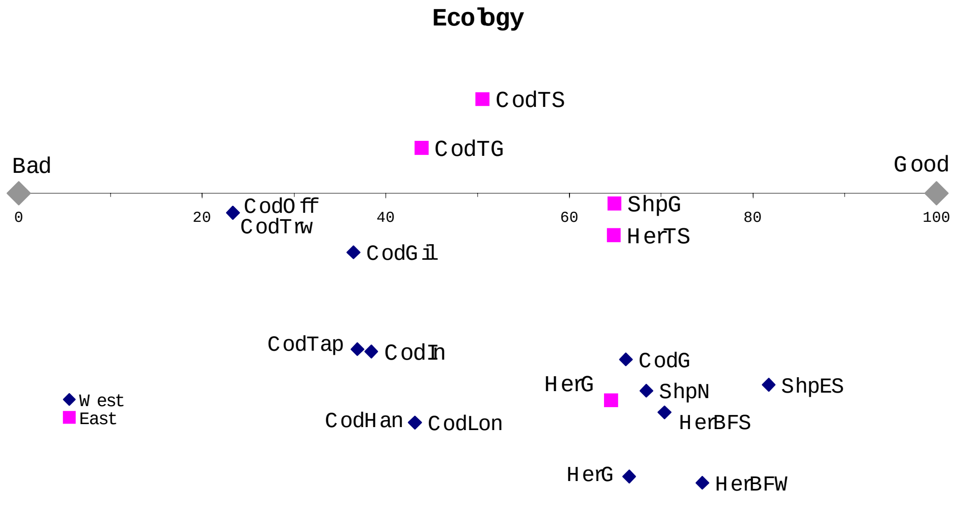


Figure 9: Two-dimensional ecological ordination of North Atlantic fisheries.

When the ranking table is resorted for economic sustainability west fisheries dominate both quartiles and most of the east fisheries are in the middle. The haddock fishery from the United Kingdom and the German demersal fisheries, however, scored in the upper and lower quartiles respectively. The fisheries in the upper quartile are also often small scale and inshore whereas in the lower quartile the fisheries are either cod or other offshore fisheries.

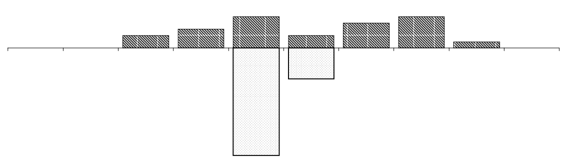


Figure 10. Frequency histogram of Rapfish social status scores for fisheries on the west (above the line) and east (below the line) side of the North Atlantic.

Ecology

A subset of east and west fisheries was plotted on a 2-d ordination plot (Figure 9) to compare similar species. In this plot, all cod fisheries except one scored low for ecological sustainability compared to herring and shrimp fisheries. There is no difference between east and west cod fisheries, however, there may be an east-west difference for the other fisheries where ecological sustainability scores are higher for the west than the east fisheries.

Social

A frequency histogram (Figure 10) shows a wide spread of social sustainability scores along the usual 0% to 100% status axis for west fisheries (20% to 90%) while in the east the range is much less (40% to 60%).

Technological

A one dimensional ordination (Figure 11) of technological sustainability scores also shows a wider spread of scores for west fisheries compared to the east fisheries. When particular species are compared the cod fisheries in the United Kingdom and Germany have similar scores, but they are lower than most cod fisheries from the east. The east herring fisheries generally scored higher than the west fisheries. The trend was also evident when shrimp fishery scores were compared.

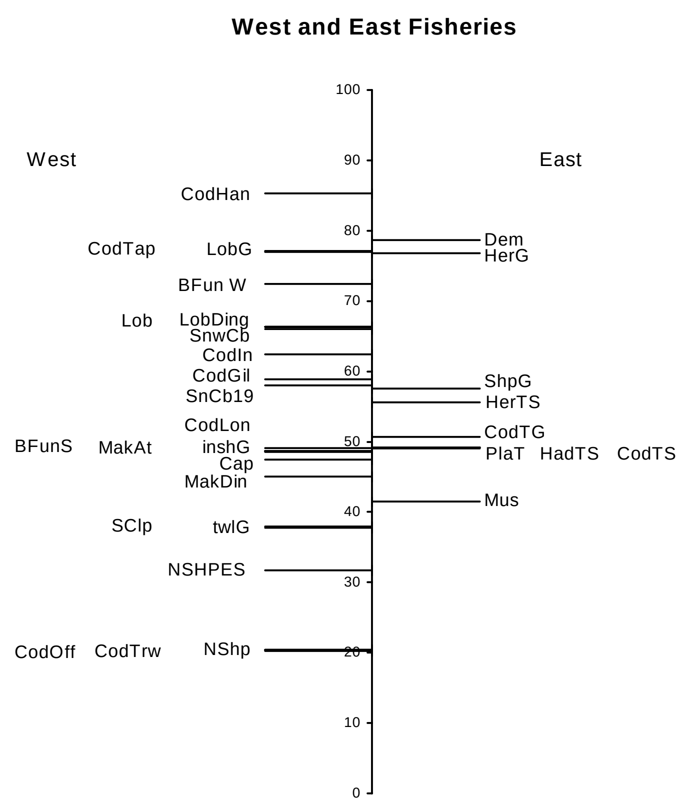


Figure 11. One dimensional ordination North Atlantic technological scores.

Compliance with the FAO Code of Conduct for Responsible Fisheries

RAPFISH Analysis

Twelve fisheries were coded (Appendix 9) and analysed (Appendix 10) for the six code of conduct evaluation fields (management objectives framework, precaution, monitoring-control-surveillance (MCS), social and economic, and stocks, fleets and gear) as defined by Pitcher (1999). Indigenous attributes were omitted from the analysis because they did not apply. The sample size of 12 is small and therefore the results presented here serve only to illustrate how the RAPFISH analysis can be used to evaluate compliance. A full analysis of North Atlantic Fisheries for compliance with the Code is proposed as part of the Sea Around Us Project.

The same RAPFISH method was used to undertake the MDS as in the previous case studies. The STRESS1 values (Table 7) were higher than in previous analysis and were anticipated because of the small number of fisheries used and the high degree of error in some of the scores.

The RSQ values, as expected, were lower than in the previous case studies. These results can be used to explore code compliance at the national and fishery level. When they are considered alongside results from a parallel analysis of Canadian East Coast Fisheries (Pitcher 1999) they provide an indication of their relative status.

Code of Conduct Evaluation Fields

General Comparison

The Canadian and German average scores are lower than the other nations especially for management objectives and framework (Figure 12). German and UK average compliance scores are higher than the others for stocks whereas in the Gulf of Maine average precaution scores are substantially higher. The average scores for MCS had the smallest range of only 10%.

Precaution

Table 8 lists the five highest and lowest ranking fisheries for compliance with the precautionary principle in the Code of Conduct.

Table 7. The STRESS1 and RSQ values for the combined Gulf of Maine, German and United Kingdom RAPFISH ordination

Attributes	STRESS1	RSQ
Management Objectives	0.273	0.669
Framework	0.286	0.714
Precaution	0.295	0.660
MCS	0.297	0.695
Social & Economic	0.308	0.666
Stocks, Fleets & Gear	0.289	0.666

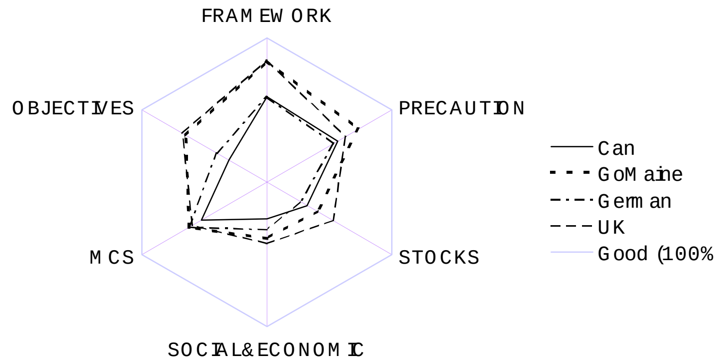


Figure 12. Kite diagram expressing the North Atlantic fisheries compliance scores by nation.

Stocks, Fleets and Gear

The scores for the stock, fleets and gear compliance for fisheries on either side of the North Atlantic are found along the entire length of the one-dimensional ordination axis (Figure 13). However, most of the fisheries are clustered in the lower half of the axis. As in previous analysis, the Canadian cod fisheries scores are low and for this compliance field the United Kingdom cod fishery compliance score falls with the range of Canadian scores. However, the score for the German cod fishery is well above the Canadian scores.

Framework

The top five fisheries for compliance with the framework code of conduct were mixed between the east and west side of the Atlantic, but German fisheries were the only European fisheries to rank in the top five. The bottom five fisheries were Canadian cod irrespective of the gear type (Table 9).

Management Objectives

When the scores for compliance with management objectives are ranked (Table 10) all five German fisheries are in the top quartile. The Gulf of Maine lobster fishery was the top ranked fishery and the Bay of Fundy weir fishery was the only Canadian fishery with a score in the top quartile. West fisheries, both inshore and offshore, dominated the lower quartile irrespective of the species. As expected, fisheries that ranked high for management objectives often scored high for other compliance evaluation fields.

Table 8 Rank orders of the top and bottom five North Atlantic fisheries in the precautionary compliance ordination, alongside rank orders from other evaluation fields. W = west, E = east side of the Atlantic, C = Canada, E= United Kingdom, G=Germany and U = United States (Gulf of Maine).
.

	Compliance Evaluation Fields						
Rank Order	**Prec.**	**Fwork.**	**Obj.**	**Stocks**	**Soc&Econ.**	**MCS**	**Area/Nation**
Gulf of Maine trawl	1	10	6	12	20	2	W/U
Gulf of Maine lobster	2	3	1	3	5	1	W/U
German herring	3	2	2	2	13	6	E/G
German cod	4	4	4	7	17	7	E/G
Lobster (Ding)	5	7	17	5	6	9	W/C
UK haddock	27	16	9	20	10	12	E/E
UK cod	28	19	12	26	12	8	E/E
Scallops	29	21	31	29	23	28	W/C
Shrimp(N)	30	24	30	30	30	29	W/C
Shrimp(ES)	31	25	29	31	31	30	W/C

Table 9. Rank orders of the top and bottom five North Atlantic fisheries in the framework compliance ordination, alongside rank orders from other evaluation fields. W = west, E = east side of the Atlantic, C = Canada, E= United Kingdom, G=Germany and U = United States (Gulf of Maine).

	Compliance Evaluation Fields						
Rank Order	**Fwork.**	**Obj.**	**Prec.**	**Stocks**	**Soc&Econ.**	**MCS**	**Area/Nation**
German demersal	1	3	12	4	14	5	E/G
German herring	2	2	3	2	13	6	E/G
Gulf of Maine lobster	3	1	2	3	5	1	W/U
German cod	4	4	4	7	17	7	E/G
Snow Crab19	5	21	7	10	1	4	W/C
Cod gillnet	27	23	20	28	26	25	W/C
Cod longline	28	15	21	16	22	21	W/C
Cod inshore	29	20	18	24	28	24	W/C
Cod trap	30	14	22	25	29	27	W/C
Cod handline	31	16	23	27	27	26	W/C

Table 10. Rank orders of the North Atlantic fisheries in the management objective compliance ordination, alongside rank orders from other evaluation fields. W = west, E = east side of the Atlantic, C = Canada, E= United Kingdom, G=Germany and U = United States (Gulf of Maine).

	Compliance Evaluation Fields						
Rank Order	**Obj.**	**Fwork.**	**Prec.**	**Stocks**	**Soc/Ec**	**MCS**	**Area/ Nation**
Gulf of Maine lobster	1	3	2	3	5	1	W/U
German herring	2	2	3	2	13	6	E/G
German demersal	3	1	12	4	14	5	E/G
German cod	4	4	4	7	17	7	E/G
German shrimp	5	11v	24	8	4	19	E/G
Gulf of Maine trawl	6	10	1	12	20	2	W/U
German mussel	7	12	13	9	3	20	E/G
Bay of Fundy (W)	8	9	8	1	2	3	W/C
UK haddock	9	16	27	20	10	12	E/E
Gulf of Maine inshore	10	6	10	23	8	31	W/U
Bay of Fundy (S)	11	13	9	11	9	14	W/C
UK cod	12	19	28	26	12	8	E/E
UK herring	13	18	25	21	16	13	E/E
Cod trap	14	30	22	25	29	27	W/C
Cod longline	15	28	21	16	22	21	W/C
Cod handline	16	31	23	27	27	26	W/C
Lobster (Ding)	17	7	5	5	6	9	W/C
Lobster	18	8	6	6	7	10	W/C
UK plaice	19	20	16	19	11	11	E/E
Cod inshore	20	29	18	24	28	24	W/C
Snow Crab19	21	5	7	10	1	4	W/C
Capelin	22	26	26	15	15	17	W/C
Cod gillnet	23	27	20	28	26	25	W/C
Mackerel (At)	24	14	14	13	18	22	W/C
Mackerel (Din)	25	15	15	14	19	23	W/C
Snow Crab	26	17	11	17	21	18	W/C
Cod trawl	27	22	19	18	25	15	W/C
Cod offshore	28	23	17	22	24	16	W/C
Shrimp(ES)	29	25	31	31	31	30	W/C
Shrimp(N)	30	24	30	30	30	29	W/C
Scallops	31	21	29	29	23	28	W/C

Social and Economic

A one dimensional ordination plot (Figure 14) indicates that overall east fisheries, especially small scale, scored higher for social and economic com-pliance than west fisheries. The lobster fisheries on the west side are the only fisheries that had comparable scores to the small scale coastal fisheries on the east side. Low cod fishery scores for social and economic compliance do-minate the lower end of the plot. The offshore fisheries of the east side, including cod, also scored higher than the Canadian cod fisheries.

Monitoring, Control & Surveillance (MCS

The top four fisheries for complying with the Code of Conduct for implementing MCS programs are from the west side (Table 11). These four fisheries were scored much higher (77% to 74%) than the fifth fishery from Germany (64%). The bottom five fisheries in this analysis were from the west, the lowest ranking fishery was the Gulf of Maine inshore fisheries. As expected fisheries that scored high for MCS often scored high for compliance with other evaluation fields. However, there are exception such as the Gulf of Maine inshore fisheries (Table-11).

Table 11. Rank orders of the top and bottom five North Atlantic fisheries in the management objective compliance ordination, alongside rank orders from other evaluation fields. W = west, E = east side of the Atlantic, C = Canada, E= United Kingdom, G=Germany and U = United States (Gulf of Maine).

	Compliance Evaluation Fields						
Rank Order	**MCS**	**Fwork.**	**Prec.**	**Stocks**	**Soc & Econ**	**Obj.**	**Area/Nation**
Gulf of Maine lobster	1	3	2	3	5	1	W/U
Gulf of Maine trawl	2	10	1	12	20	6	W/U
Bay of Fundy (W)	3	9	8	1	2	8	W/C
Snow Crab19	4	5	7	10	1	21	W/C
German demersal	5	1	12	4	14	3	E/G
Cod trap	27	30	22	25	29	14	W/C
Scallops	28	21	29	29	23	31	W/C
Shrimp(N)	29	24	30	30	30	30	W/C
Shrimp(ES)	30	25	31	31	31	29	W/C
Gulf of Maine inshore	31	6	10	23	8	10	W/U

DISCUSSION

Alternatives to Rapfish

Other approaches to assessing the sustainability of fisheries have been developed or are under development. The most advanced approaches include the Marine Stewardship Council Certification Program, the Fisheries Assessment Framework (Australia) and FAO's Sustainable Development Reference System.

Marine Stewardship Council Certification (MSCC)

The Marine Stewardship Council has initiated a global accreditation scheme for the commercial fishing sector. The scheme is based on defining performance criteria and guideposts for a particular fishery and then scoring the fishery. The fisheries are scored by accredited certifiers to reduce uncertainty and increase the consistency in the decision making process (MSC 1998a). Currently each fishery has its own set of criteria and guideposts. The MSC anticipates a generic set of criteria and guideposts will be available by March 2000. The performance criteria are used in conjunction with the Analytical Hierarchy Process as defined by the MSC. The fisheries are evaluated against three principles, Principles 1 and 2 focus on the biological or ecological aspects of the fishery while Principle 3 in 10 elements covers a diversity of issues spanning social, cultural, technological, cultural and ethical issues (MSC 1998a). Limited attention is given to specific social, ethical, economic

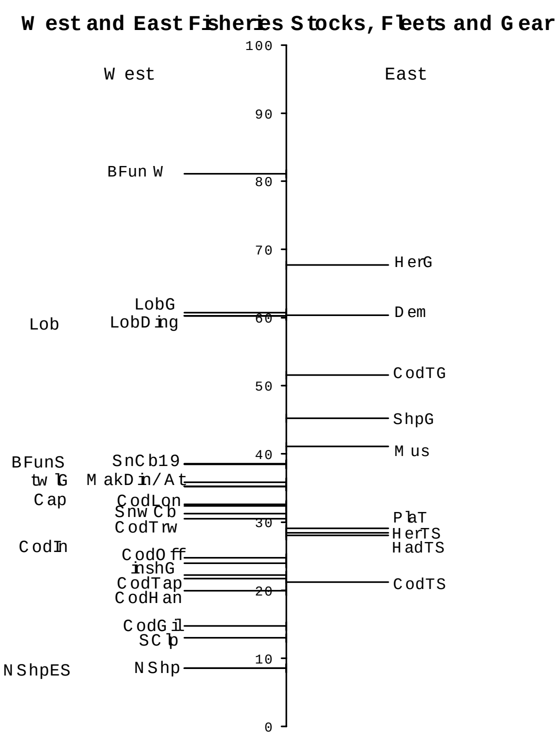


Figure 13. One dimensional ordination North Atlantic stock, fleets and gear compliance scores.

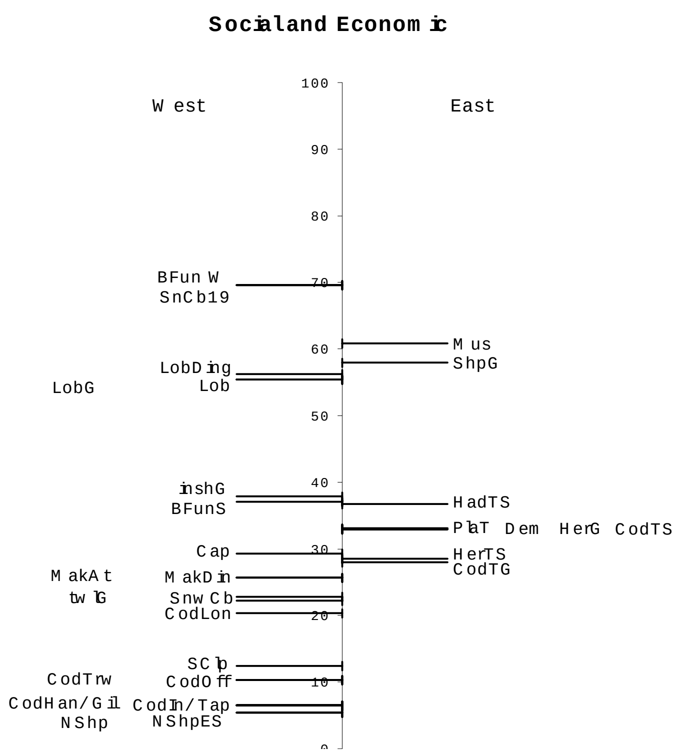


Figure 14. One dimensional ordination North Atlantic social and economic compliance scores.

and technological aspects of fisheries management and sustainability.

The performance criteria (indicators) and guideposts are based on these three principles, and usually more than one indicator is used to measure compliance with a principle. Within a single indicator the elements to be considered in an evaluation are also detailed so that the MSC Certifier can score a particular indicator. Each element is scored by consensus among the certifying team. Once the fishery is scored (as a percent) 'a normalised weight average score' for each principle and criteria category is calculated between 0 and 100% (MSC 1998b). This enables the team to assess how the fishery scores relative to the "full attainment (appropriate to the size and scale of the fishery) of a sustainable fishery (MSC 1998 p.17)".

The Western Australia Rock Lobster Fishery, which was the first fishery to be accredited, gives an indication of the scope and nature of the criteria and guideposts. In this fishery 5 indicators were defined and scored for Principle 1, 7 for Principle 2 and 5 for Principle 3 (MSC 1999). The element necessary for a 100% and an 80% score were provided for the certifiers.

The MSC Accreditation approach and the RAPFISH approach both score fisheries according to a well defined set of criteria. The scaling differs substantially as well as the number and scope of criteria with the RAPFISH approach using an interval scale for a number of criteria (nearly 90, see Appendix 2 for a list) and MCS using a percent scale for fewer criteria (17 for the WA Lobster Fishery (MSC 1999)). The RAPFISH approach enable managers to visually see how the fisheries has changed in terms of sustainability by the plots produced, and more importantly the kite diagrams are able to integrate the various aspects of sustainability into a single entity allowing managers to assess which areas of management need attention. The aggregating approach in MSC results in considerable loss of information while RAPFISH, which uses MDS, is able to retain much of the information as well as estimate the robustness of the assessment using Monte Carlo simulations and sensitivity analysis. The MSC approach requires accredited certifiers and requires a team of highly trained professionals which for the most part will be paid by the client who is seeking accreditation. The MSC guidelines indicate that a typical fishery requires 40 person days for establishing the accreditation and 12 person days each year to maintain the accreditation (MSC 1998). This requirement may limit its use in developing countries or for fisheries where profit margins are not exceedingly high. There are no such requirements with RAPFISH which can be undertaken by a range of stakeholders after minimal training.

Fisheries Assessment (FA) Framework (Australia)

The Bureau of Rural Science in Australia has developed a 'framework to assess fisheries with respect to ecologically sustainable development'. The framework focuses on providing a structure and process that can be used to meet the unique needs of each fisheries. The framework is based on the effects of fishing that are examined in terms of impact on ecological processes and the total quality of life. The direct effects of fishing on human society and the effects of fishing on the environment are included. The effects on human society and the environment are further subdivided in a hierarchical fashion. The structure can be adapted to the specific circumstances of any fishery through further subdivision to whatever level is desirable (Chesson & Clayton 1997). Criteria are developed within the framework and once developed a multicriteria analysis through time is used to assess the sustainability of the fisheries studied. This approach does not necessarily provide common measures or indicators for sustainability and is quite different to RAPFISH. However, once measures or indicators are developed within the framework RAPFISH can be used to assist in assessing the fishery.

Sustainable Development Reference Systems (SDRF)

The system is based on the FAO Code of Conduct and recognises the need to take a multi-disciplinary approach to assess sustainable development. SDRS sets objectives, related indicators and their respective reference points (FAO 1999) and includes economic, social, ecological and governance aspects as well as the FAO Code of Conduct. The indicators can be defined for the scale of the fisheries so that it can be used in a range of situations. The system also considers the aggregation and presentation of the information obtained. The system, however, is developed on a fishery by fishery basis.

Again the approach is similar to RAPFISH since indicators are scored according to a set criteria. However, the FAO guidelines also set reference points to indicate how the fishery is preforming for the specified criteria. The approach can also use kite diagrams to illustrate how well a particular fishery is preforming for a subset of indicators, however, there is no facility to integrate or include other fisheries (FAO 1999). The approach focuses on a fishery by fishery analysis and therefore no measures of error are possible and comparisons are limited between fisheries if separate criteria are developed for each fishery.

Use of Rapfish

Rapfish is a new and non-traditional approach to fisheries management. Consequently in some fisheries sectors scepticism has been expressed especially on the lack of uncertainty estimates and the dominance of attributes. This study of a total of 117 fisheries spanning several centuries in some fisheries, a range of gear and species, covering both sides of the North Atlantic has demonstrated that reasonable uncertainty estimates can be derived in both dimensions. Monte Carlo has been used in the past in MDS to investigate the uncertainty associated with STRESS estimates (Spence and Young 1978). The STRESS values obtained in this study were within a narrow range (0.02). The Monte Carlo simulations for all five-evaluation fields had similar results in not just STRESS values, but also in the confidence intervals as shown in Figure 6. In all evaluation fields the 95% confidence intervals were wider in the second dimension compared to the first. This is a consequence of the second dimension accounting for the non-sustainability component of the attributes.

Bootstrapping was not used in this analysis to investigate uncertainty with the scores because the resampling strategy allows for duplications that many statistical programs do not allow. In addition, analysis with duplicate (tied) data will result in higher STRESS values and therefore potentially give misleading results.

The sensitivity analysis results are quite comparable to previous sensitivity analysis (Pitcher 1999) where the standard error of the sustainability dimension (as a % score) was generally less than 14% for any attribute. There is no single attribute that dominates the Rapfish ordinations. As in the Monte Carlo study, the uncertainty of attributes in the second dimension was higher, but not substantially. This study and previous studies (Pitcher 1999) have shown that the attributes listed are effective in defining sustainability. The sensitivity analysis is also a useful tool when new attributes are proposed since their influence can be assessed.

The analysis of 117 fisheries has highlighted the need to determine a set of reference points so that fisheries from different MDS analysis can be compared, especially when the number of fisheries exceeds 100 (the limit for many statistical programs). In this analysis the three fisheries could not be combined since the total number of fisheries exceeded 100. Confirmatory analysis, however, may provide the basis for allowing for comparisons (Young and Arabie 1998). This aspect needs further investigation.

This study has clearly shown that Rapfish is based on proven multidimensional scaling theory and approaches within the broad subject of multivariate analysis. This basis combined with previous analysis and the analysis of the 117 fisheries in this study confirms Rapfish as an appropriate method to assess the sustainability of fisheries and compliance with the FAO Code of Conduct for Responsible Fisheries. The ability of Rapfish to express the uncertainty associated with the analysis further strengthens its appropriateness when compared to current approaches and traditional fisheries management approaches.

This study has also demonstrated how Rapfish can be used to undertake hierarchical analysis of fisheries as demonstrated in the east-west comparison analysis of North Atlantic fisheries. Rapfish enables managers to make comparisons between gear sector, time, fishing scale and geographic location at a very broad level or in considerable detail. There are also a number of options to illustrate these comparisons from 2-dimensional ordination plots to kite diagrams depending on the required analysis.

The strength of Rapfish lies in its ability to integrate a range of attributes into a single analysis and the visualisation of these results. All stakeholders easily understand the graphs and plots used to express the two-dimensional Rapfish analysis. Rapfish therefore provides a refreshing change for stakeholders to objectively come to grips with the issue of fisheries sustainability.

Other possible points for further discussion - Hierarchical use, scalability, enanbling comparisons by gear sector, by year and size scale /country. comparisons with compliance with international treaties.

ACKNOWLEDGEMENTS

The *Sea Around Us Project* is supported by the Environment Program of the Pew Charitable Trust. We thank Dr Norm Hall and Dr Leif Nøttestad for reviewing and enhancing this report.

REFERENCES

Arabie, P. (1973) Concerning Monte Carlo evaluations of nonmetric multidimensional scaling algorithms. *Psychometrika,* 38(4).

Buckingham-Hatfield, S. and Evans, B. (1996) Achieving sustainability through environmental planning. Environmental Planning and Sustainability. In: S. Buckingham-Hatfield and B. Evans, Wiley, Chichester, UK. 1-18.

Carroll, J. D. and Arabie, P. (1998) Multidimensional Scaling. Measurement, Judgement, and Decision Making. (ed. M. Birnbaum), San Diego, Academic Press.

Chesson, J. and Clayton, H. (1997) A framework for assessing fisheries with respect to ecologically sustainable development. Bureau of Resource Sciences Fisheries Resources Branch, Canberra, Australia.

Clark, K. R. and Warwick, R. M. (1997) Change in Marine Communities: An Approach to Statistical Analysis and Interpretation. Plymouth: Plymouth Marine Laboratory, 144 pp.

Davison, M. L. (1983) Multidimensional Scaling. USA, John Wiley & Sons, Inc.

FAO (1999) The development and use of indicators for sustainable development of marine capture fisheries.

FAO Technical Guidelines for Responsible Fisheries No. 8, Rome.

Green, P. E., Carmone, F. J. Jr., and Smith, S. M. (1989) Multidimensional Scaling: Concepts and Applications. Massachusetts, Allyn and Bacon.

Jacoby, W. G. (1991) Data Theory and Dimensional Analysis. Newbury Park, Sage Publications, Inc.

Kruskal, J. B. and Wish, M. (1978) Multidimensional Scaling. Beverly Hills, Sage Publications.

Leeuw, J. de. and Meulman, J. (1986) A special jacknife for multidimensional scaling. *Journal of Classification,* 3(1), 97-112.

Manly, B. F. J. (1994) Multivariate Statistical Methods: A Primer. London, International Thomas Publishing.

Marine Stewardship Counsel (1998a) Principles and Criteria for Sustainable Fishing.

Marine Stewardship Council (1998b) MSC Certification Methodology.

Marine Stewardship Council (1999) Performance Criteria and Scoring Guideposts Against the Marine Stewardship Council Principles and Criteria: Western Australian Rock Lobster Fishery Issue.

McGoodwin, J. (1990) Crisis in the World Fisheries: people problems and policies. Stanford Univ. Press. Stanford.

Palmer, M., (2000) Ordination Methods for Ecologists. http://www.olcstate.edu/artsci/botany/ordinate. [accessed 03/03/00]

Pitcher, T.J. (1999) RAPFISH, A Rapid Appraisal Technique For Fisheries, And Its Application To The Code Of Conduct For Responsible Fisheries. FAO Fisheries Circular No. FIRM/C: No. 947. 47 pp.

Pitcher, T.J. and Power, M.P. (2000) Fish Figures: Quantifying the Ethical Status of Canadian Fisheries, East and West. In: Coward, H., Ommer, R. and Pitcher, T.J. (eds). Just Fish: the ethics of Canadian fisheries. Institute of Social and Economic Research Press, St John's, Newfoundland, 225-253.

Pitcher T. J. and Preikshot, D. B. (2000) Rapfish: A Rapid Appraisal Technique to Evaluate the Sustainability Status of Fisheries. Fisheries Research (in press).

Pitcher, T.J., Bundy, A., Preikshot, D., Hutton, T. and Pauly, D. (1998) Measuring the unmeasurable: a multivariate interdisciplinary method for rapid appraisal of health of fisheries. In: Pitcher, T.J. Hart, P.J.B. and Pauly, D. (eds) Reinventing Fisheries Management Chapman and Hall, London, 31-54.

Pitcher, T.J., Mackinson, S., Vasconcellos, V., Nøttestad, L. and Preikshot, D. (1999) Rapid appraisal of the status of fisheries for small pelagics using multivariate, multidisciplinary ordination. In: Quinn, T.J. II, Funk, F., Heifetz, J., Ianelli, J.N., Powers, J.E., Schweigert, J.F., Sullivan, P.J., Zhang, C.I. (eds), Fishery Stock Assessment Models. Alaska Sea Grant, Fairbanks, 759-782.

Preikshot, D.B. (2000) An Interdisciplinary Assessment of Troical Small Scale Fisheries Using Multivariate Statitstics. Masters Thesis, Department of Zoology, Fisheries Centre. University of British Columbia. Vancouver, Canada.

Preikshot, D.B., Nsiku. E., Pitcher, T.J. and Pauly, D. (1998) An interdisciplinary evaluation of the status and health of African lake fisheries using a rapid appraisal technique. J. Fish Biol. 53 (Suppl A), 382-393.

Schiffman, S., Reynolds, M. L. and Young, F. W. (1981) Introduction to multidimensional scaling: theory, methods, and applications. Academic Press, London.

Smith, S. J. (1993) Risk Evaluation and biological reference points for fisheries management: A review. In: Kruse, G. Eggers, D.M., Marasco, R.J., Pautzke, C. Quinn, T.J., (eds) Management Strategies for Exploited Fish Populations. Alaska Sea Grant, Anchorage. 339-353.

Spence, I. and Young, F. W. (1978) Monte Carlo studies in nonmetric scaling. *Physicometrika*, 43(1).

Stalans, L.J. (1995) Multidimensional Scaling. In: Grimm, L.G. and Yarnold, P.R. (eds) Reading and Understanding Multivariate Statistics. American Psychological Association, Washington.

Statistica (2000) Multidimensional Scaling. http://www.statsoft.com/textbook/stmulsca.html. [accessed 11/03/00]

Walters, C.J. (1998) Designing fisheries management systems that do not depend on accurate stock assessment. In: Pitcher, T.J. Hart, P.J.B. and Pauly, D. (eds) Reinventing Fisheries Management Chapman & Hall, London, 279-288.

Weinberg, S. L., Carroll, J. D. and Cohen, H. S. (1984) Confidence regions for indscal using the jackknife and bootstrap techniques. *Psychometrika*, 49(4), 475-491.

Young, F. W. (1987) Multidimensional Scaling: History, Theory, and Applications. Hillsdale, Lawrence Erlbaum Associates.

Young, F. W. and Hamer, R. M. (1987) Multidimensional Scaling: History, Theory, and Applications. London, Lawrence Erlbaum Associates, Publishers.

APPENDIX 1
MULTIDIMENSIONAL SCALING AS THE ORDINATION ENGINE IN THE RAPFISH TECHNIQUE

MULTIVARIATE ALTERNATIVES TO MDS

Cluster Analysis

Cluster Analysis and MDS have some similarities – both can analyse proximity data and use distance models, and in either approach, the solution can be represented as coordinates in k-dimensions (Davison 1983). There are fundamental differences: cluster analysis can not express the relationship between the distance data as a linear or monotone function, distances in hierarchical cluster analysis are not spatial distances as in MDS, and the coordinate dimensions in MDS are continuous whereas in cluster analysis they are discrete (Davison 1983). Cluster analysis can complement MDS by identifying groups of similar points (fisheries) which may provide further analysis with respect to sustainability

Factor Analysis (FA)

FA is similar to MDS since it also measures the proximity of pairs of points expressed as angles between vectors (Davison 1983) and both use a Euclidean space (Schiffman et al. 1981). FA attempts to account for the variation in a number of original variables using a smaller number of vectors, and FA can be used to explore the relationships between different variables. The number of vectors used (10 or more) and expressing the distances as angles makes interpretation of the data difficult. MDS on the other hand explores the distances between the points and their relative positions in a few dimensions. FA is not appropriate where samples are small and cannot be replicated (Manly 1994), or where relationships are not linear.

Principal Component Analysis (PCA)

PCA allows researchers to investigate combinations of variables that reduce the number of variables so that data is simplified. PCA can not use distance or similarity matrices and more importantly the data should be approximately normally distributed. The normality requirement severely limits the use of Principal Component Analysis since the scored fisheries data are often not normally distributed. When data are normally distributed the output from a PCA are indices that are uncorrelated and represent different dimensions. The indices (principle components) are ordered such that the first accounts for the largest amount of variation (Manley 1994). The principal components, however, do not necessarily reflect sustainability as in RAPFISH.

Correspondence Analysis

Correspondence analysis examines the abundance of data, often in the form of frequency or contingency tables (Green et al. 1989). It can be used to spatially represent the frequency data, and therefore differs from MDS where distances or dissimilarities involving sustainability are spatially represented.

MULTIDIMENSIONAL SCALING

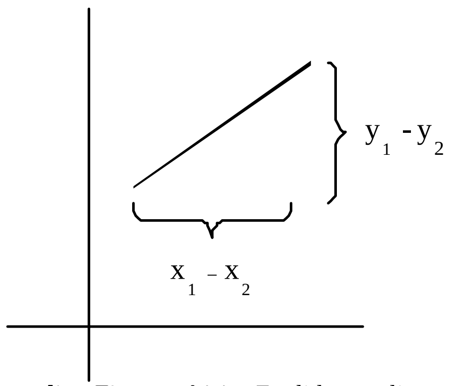
The Monte Carlo simulations indicated that scores are not necessarily normally distributed when a two dimensional analysis is undertaken and confidence intervals around the median values provide a robust indicator of the uncertainty. Monte Carlo runs to date indicate limited variation in the estimate of the median values in the first and second dimension. The sensitivity analysis (SA), based on jackknifing (without replacement) was used to investigate the influence of a specified attribute. Results in this study indicate that RAPFISH estimates are stable and not dominated by one particular attribute. Preliminary results comparing the distances between the original points and the SA points also found no significant differences.

Multi-attribute Utility Theory

Multi-attribute utility Theory (MAUT) has been suggested as an alternative to MDS within Rapfish. MAUT is often used to assist managers and stakeholder in deciding on the most appropriate option amongst several options. If fisheries are considered as options and the objectives was to decide which fishery is the most sustainable based on a defined set of criteria, then MAUT can be used. Applying this technique would require not only the table of scores against the various criteria but also weighting of the attributes as well as forming utility functions for each attribute by the stakeholders. In practice this would not be an easy task due to the diversity of stakeholder interests. Reviewers of Rapfish have questioned whether the methodology is already too complex for most stakeholders, defining the weightings and the utility functions add a layer of complexity to the current methodology. MAUT would limit the ability of managers to compare results with other fisheries since each weighting and utility function would be unique to the fishery or group of fisheries.

The Ordination Method

MDS is a distance based ordination method which seeks to map distances between 'objects' or points in a two or three dimension space as close as possible to the distances between the original (input) points in a multi-dimensional space. The ordination



Appendix Figure A1.1. Euclidean distance in 2 dimensional space.

technique approximates a configuration (ordination) of points in a t-dimensional space (usually 2 or 3 dimension) by using the Euclidean distances (Figure A1.1) between points in an initial configuration in the t-dimensional space and the distances between the original (input) points.

The Euclidean distance between points is calculated using Pythagorean Formula; in 2 dimensional space it is

$$d = \sqrt{|x_1 - x_2|^2 + |y_1 - y_2|^2}$$

In a n-dimensional space the Euclidean distance is

$$d = \sqrt{|x_1 - x_2|^2 + |y_1 - y_2|^2 + |z_1 - z_2|^2 + ...)}$$

A configuration is approximated by regressing the Euclidean distances in the initial or estimated configuration (d_{ij}) on the Euclidean distances between the original (input) points (δ_{ij}); this equation is then used to estimate the distances (disparities) of the original distances (δ_{ij}) scaled as close as possible to match the Euclidean distance (d_{ij}) in the configuration. The form of the regression equation will vary depending on whether it is linear, polynomial or monotonic (Manly 1994). An example of the regression and disparities are given:

$d_{ij} = a + b\delta_{ij} + e$ (an example of a linear equation)

$\hat{d} = a + b\delta_{ij}$ (disparity)

where d_{ij} is the Euclidean distance, a is the intercept, b is the slope, e is the error term and $\hat{d}$ is the disparity .

When a linear or polynomial regression is used the MDS is metric, and when a simple monotonic regression is used the MDS is nonmetric (Kruskal and Wish 1978). In a nonmetric MDS the ordering of the data distances is therefore important (Manly 1994) and consequently maximises the rank-order correlation between distance measures and distance in the ordination [configuration] space (Palmer 2000).

Generally one of three methods is used to estimate the regression: least squares based on distances (KRYST), alternating least squares based on squared distances (ALSCAL) and maximum-likelihood-based procedure (MULTISCALE) (Schiffman et al. 1981; Carroll and Arabie 1998). Each method makes assumptions about the data to be analysed and generates different output. The KYST and ALSCAL can not estimate confidence levels directly, while the MULTISCALE approach can be used to estimate confidence regions. However, MULTISCALE has several strong assumptions about the distribution of the data and can only be used in metric MDS. Estimating confidence intervals is discussed below.

How well the configuration (ordination) of points reflects the original data is termed 'goodness of fit' or 'stress'. The stress is measured between the Euclidean distances and the disparities, if the stress is not reduced significantly, the coordinates of the points in the configuration are moved to reduce the stress. New Euclidean distances are calculated along with a new regression, disparity and stress when the points are moved. Most algorithms therefore evaluate different configurations iteratively with the goal of maximising the goodness of fit or minimising the stress Figure A1.2.

δ_{ij} calculated → configure → d_{ij} → $\hat{d} = a + b\delta_{ij}$ → STRESS calculated

points adjusted

Kruskal's 'STRESS Formula 1' or STRESS1 is often used to measure stress. The general formula is (Manly 1994):

$$\text{STRESS1} = \{\Sigma(d_{ij} - (\hat{d}_{ij})^2/\Sigma \hat{d}_{ij}\}^{1/2}$$

In nonmetric MDS the configurations are evaluated

Appendix Figure A1.2: A generalised flow chart for computing MDS.

iteratively to minimise the stress which can be expressed as:

$$\text{STRESS} = \Sigma[d_{ij} - f(\delta_{ij})]^2$$

Where $f(\delta_{ij})$ is a nonmetric monotone function of the original distances (Statistica 1995).

A low stress value generally indicates a good fit when the analysis is in two or more dimensions. The actual value of the stress coefficient depends on the formula used, for Kruskal's STRESS 1 coefficient values greater than 0.10 in 2 dimension indicated a poor fit. If the data has high levels of measurement or sampling error then the stress value may exceed .10 (Kruskal and Wish 1978). However, for the same degree of fit, STRESS2 values are usually twice as much as for STRESS1 (Kruskal and Wish 1978).

Stress values close to zero, however, should be treated with caution since they may indicate a degenerate solution as discussed below. Similarly, large stress values may indicated that convergence to a solution was not reached.

There are a number of factors that affect stress, the most likely include:

- Introducing a distance matrix where the symmetrical points are significantly different from each other;
- Introducing replicates that are significantly different from each other; and
- The number of ties results in only a few distinct values (Kruskal and Wish 1978).

The ordination usually provides a set of points (coordinates) configured for either two or three dimensional space. Because the configuration represents the relative position of the points, rotating or reflecting the configuration and altering the scales will not change the relative positions of the points (Manly 1994). The final orientation of the axes and their corresponding scales are therefore subjective and generally based on which orientation is easily explained.

Appendix Table A1.1: Programs for computing MDS (based on Manly, 1994).

Algorithm (Metric - NonMetric)	Source	Authors
ALSCAL (both)	SPSS SAS	Young and Lewyckyj (1979) as cited in Manly 1994.
KYST (both)	Separate program available from Bell Laboratories.	Kruskal, Young and Seery (1973) as cited in Carroll and Arabie (1998).
MULTISCALE (metric)	Separate program available from International Education Services.	Ramsay (1989) as cited in Carroll and Arabie (1998).
MDSCAL (non-metric)	Primer.	Kruskal as cited in Clark and Warwick 1994.
NMDS (non-metric)	Written as a ‘C’ module which can be called by such programs as SPLUS.	Ludwig and Reynolds 1988 as cited in Manly 1994.

Clearly MDS is a computationally intensive method since calculating the Euclidean and regressed distance involves several calculations as well as the calculations required to derive the stress parameter. Current computer technology, however, makes this task much easier and faster so that several fisheries with a number of attributes can be analysed. There are a number of algorithms available to conduct a MDS analysis. Different programs use different algorithms (Table A1.1) and therefore do not give the same result (Manly 1994), however, the relative positions of the fisheries on the maps should be similar.

MDS Methodology Issues

Degenerate Solutions

Degenerating solutions are characterised by Stress values close to 0. They occur in MDS analysis when there is a relatively small amount of empirical data used to estimate a relatively large amount of information (Jacoby 1991) or there is clustering of the distances. Degenerate solutions can be avoided by increasing the amount of input data (Jacoby 1991) or by seeking a solution in a higher dimension (Davison 1983).

Sample sizes

The question of how many attributes to use and how many points (fisheries in the case of RAPFISH) to sample in a MDS analysis is raised irrespective of the nature of the project. Since the number of dimensions (K) that can be explored increases with the number of attributes (J), as many attributes as possible should be used (Schiffman et al 1981). When the analysis is focused in two dimensions the recommended number of attributes ranges between 9 (Kruskal and Wish 1978) and 12 (Schiffman et al. 1981). The general recommendation is:

$J - 1 >= 4xK ; K <= 3$

to achieve statitistical stability (Kruskal and Wish 1978).

The number of points (N) that should be sampled is also a function of the number of attributes. The more points sampled the better the fit of the data. Stalans (1995) recommends that three times as many points are sampled than the number of attributes:

Minimum (N) = 3*J.

Researchers are also interested in the impact of the attributes on the analysis, that is, do one or two attributes dominate the ordination? This dominance can be explored using multiple regression analysis (regression coefficients and canonical correlation) and sensitivity analysis (see below).

Significant regression coefficients in a multiple regression of the first and second dimensons indicate that one of the attributes may be dominating the analysis. However, the regression applies only to individual ordinations and does not necessarily identify which attribute. Canonical correlation coefficients can infer dominant attributes since high positive correlations indicate that a particular attribute score is likely to score high on an ordination axis. Negative correlations also imply that low attribute scores were associated with high values on an ordination axis (Pitcher and Preikshot in press). These correlations only infer or imply, because the MDS dimensions are jointly determined

Flipping
In an MDS analysis, points can "flip" from one iteration to another. This is due to the fact that the configurations always have a fair degree of randomness associated with them and therefore fitting the best solution may involve moving (or "flipping") nearby points. This phenomenon is likely related to the interdependence of nearby points, however, little information is available. Flipping in an analysis can be investigated by using sensitivity analysis (Kruskal and Wish 1978). Pitcher (1999) and Kavanagh (pers. comm.) found that 'flipping' could be minimised by the use of fixed anchor points – an extension of the anchor points system is currently under investigation.

Confidence Limits
MDS, like all multivariate methods, estimates the coordinates that are used to place the relative points on a map. Indeed, MDS is often described as a "arranging objects in a space with a particular number of dimensions" (Statsoft 2000 – URL). Because these coordinates are estimated, reported details of the error or degree of uncertainty associated with each coordinate would enhance the robustness of the MDS analysis. As discussed above most of the algorithms used to estimate the coordinates can not estimate uncertainty or confidence limits directly. The MULTISCALE approach can be used to estimate the standard error of the log likelihood estimate and the standard error of the coordinate estimates, however, it is a metric MDS and requires a large sample sizes and normally distributed data (Schiffman et al. 1981). This limits its use in RAPFISH since the data is often not normally distributed.

Confidence limits or levels of uncertainty can be estimated indirectly by using resampling methods such as Monte Carlo, Bootstrap or Jackknife (Weinberg et al. 1984). Monte Carol resampling maintains the attribute values for each fisheries, but randomly permutes them to estimate the test statistic or parameter. This permutation/parameter estimation process is repeated several times (minimum of 100 times) to obtain the distribution of the parameter. Confidence intervals can then be estimated based on the distribution of the data generated in the Monte Carlo simulations.

The bootstrap approach maintains not only the attribute values for each fishery, but does not permute the values. The resampled fisheries are generated by sampling randomly, with replacement, to obtain a data set the same size as the original set. MDS analysis is conducted on this new data set. This resampling/parameter estimation sequence is replicated many times and once completed the replicates are used to estimate the bias, mean and standard error for the parameter. It should be noted that in some bootstrap programs the resulting data set may have replicates (ties) and therefore the original data set should be large and preferably with few, if any ties.

The jackknife approach also maintains the attribute values for each fishery, but it leaves out one fishery and recalculates the parameters based on a sample size of n-1. Bias, mean and standard error are estimated but calculated differently to those in the bootstrap method. For analysis with a large number of points the perturbation will be very small (deLeeuw and Meulman 1986).

Which method is most appropriate to investigate uncertainty depends on which parameter is sought, the nature of the data and the program used (Table A1.2). Programs that prohibit ties can not be used for some bootstrap methods since some sample with replacement and therefore if the same point is sampled twice, the distances will be zero and the MDS program will fail. Because bootstrap and jackknife methods resample the input data directly, large sample sizes should be used with these methods.

Appendix Table A1.2: Summary of re-sampling Methods.

Resampling Method	Sampling Strategy	Estimates
Monte Carlo	Permutations	parameters
Bootstrap	Sampling with replacement	Bias, mean, standard error, percentile, sensitivity
Jackknife	Sample (n-1)	Bias, mean, standard error, percentile (calculations different to BS), sensitivity

Sensitivity Analysis
Sensitivity analysis enables researchers to examine the impact of an attribute or the appropriateness of the number of dimensions.

The jackknife and bootstrapping methods mentioned above are used to investigate the sensitivity of both points in the ordination, and the attributes used in a MDS (deLeeuw and Meulman 1986; Spence and Young 1978; Arabie 1973). If an analysis is stable then small perturbations in the data produce small changes in the solution (deLeeuw and Meulman 1986). In a jackknife or bootstrap analysis the difference between the sums of squares of full set of attributes and the generated set of attributes (one attribute missing in the case of a jackknife) can be compared. The difference can be expressed as a standard error for each attribute (Pitcher 1999). A sensitivity analysis enables researchers to investigate the appropriateness of the dimension specified by examining changes in the stress values in different dimensions, the stability of stress values and coordinates.

Hypothesis Testing/Confirmatory MDS
The strength of MDS lies in its ability to map points, however, MDS can also be used to evaluate a hypothesis about an ordination. Confirmatory MDS evaluates how well an ordination can be reproduced by an ordination that is hypothesised or where the ordination has been specified a priori. The prior points can be fixed to a previous value, proportional to another value or completely unconstrained (Young and Hammer 1987; Carroll and Arabie 1998). MDS approaches are also used to analyse the combined data set. Some or all of the coordinates are specified (hypothesised) and the remaining coordinates (unhypothesised) are estimated in some forms of confirmatory MDS approaches (Davison 1983). Confirmatory MDS allows researchers to constrain the ordination so that points in the ordination can be compared to known (hypothesised) points and in some cases allow for comparisons to be made between analysis where the same hypothesised points are used.

Ties/Clustering
When points in the map are at the same position, no information is available about the relationship between the two points since their distances are equal to zero. If there are numerous ties then the stress value can be high since the distances will be either clustered or take on only a few distinct values (Davison 1983; Schiffman et al. 1981). In MDS there are two approaches to handling ties. The primary approach breaks up the rank order of the ties so that the stress is minimised and the secondary approach replaces the ranks by their average (Schiffman et al. 1981). In analysis where there are a number of ties or "no differences" the results should be interpreted cautiously (Schiffman et al. 1981).

Missing data
The impact of missing values in MDS depends on how they are handled by the computer program. Incorporating missing values has the advantage of increasing the number of attributes and samples, and therefore allowing analysis in higher dimensions (Schiffman et al. 1981). Some programs omit those cases where the data is missing in one or more of the attributes, others estimate the missing value and substitute it into the analysis. Young and Hammer (1987) suggest that when data is discrete the optimal scaling process assigns a single number and when the data is continuous it assigns a continuum of numbers.

APPENDIX 2: ATTRIBUTES CURRENTLY USED IN RAPFISH ANALYSES FOR ECOLOGICAL, TECHNOLOGICAL, ECONOMIC, SOCIAL AND ETHICAL EVALUATION FIELDS. (Revised March 2000 by RAPFISH Group).

	Scoring	Good	Bad	Notes
Ecological analysis				
Exploitation status	0; 1; 2; 3	0	3	FAO-like scale: under- (0); fully- (1); heavily- (2); or over-exploited (3) [can consult FAO website for status]
Recruitment variability	0; 1; 2	0	2	COV: low <40% (0); medium 40-100% (1); or high >100% (2)
Change in trophic level	0; 1; 2	0	2	Is trophic level of fisheries sector decreasing: no (0), somewhat, slowly (1); rapidly (2).
Migratory range	0; 1; 2	0	2	# of jurisdictions encountered during migration (includes international waters): 1-2 (0); 3-4 (1); >4 (2)
Range collapse	0; 1; 2	0	2	Is there evidence of geographic range reduction: no (0); a little (1); a lot, rapid (2).
Size of fish caught	0; 1; 2	0	2	Has average fish size landed changed in past 5 years; no (0); yes, a gradual change (1); yes, a rapid large change (2).
Catch before maturity	0; 1; 2	0	2	percentage caught before maturity: none (0); some (>30%) (1); lots (>60%) (2)
Discarded by-catch	0; 1; 2	0	2	percentage of target catch: low 0-10% (0); medium 10-40% (1); high >40% (2)
Species caught	0; 1; 2	0	2	includes species caught as by-catch: low 1-10 (0); medium 10-100 (1); high >100 (2)
Primary production	0; 1; 2; 3	3	0	g C/m^2/year: low 0-50 (0); medium 50-90 (1); high 90-160 (2); very high >160 (3)
Economic analysis				
Profitability	0; 1; 2; 3; 4	4	0	Highly Profitable (0); marginally profitable (1); break even (2); losing money (3); big losses (4)
Fisheries in GDP	0; 1; 2	2	0	Importance of fisheries sector in national economy: low(0); medium (1); high(2)
Average wage	0; 1; 2; 3; 4	4	0	Do fishers make more or less than the average person? Much less (0); less (1); the same (2); more (3); much more (4)
Limited entry	0; 1; 2; 3; 4	4	0	includes informal limitations: Open Access (0); Almost none (1); very little (2); some (3); lots (4)
Marketable right	0; 1; 2	2	0	marketable right/quota/share? (0); some (1); mix (2); full ITQ, CTQ or other property rights (2)
Other income	0; 1; 2; 3	0	3	in this fishery, fishing is mainly: casual (0), part-time (1); seasonal (2); full-time (3)
Sector employment	0; 1; 2	0	2	employment in formal sector of this fishery: <10% (0); 10-20% (1); >20% (2)
Ownership/Transfer	0; 1; 2	0	2	profit from fishery mainly to: locals (0); mixed (1); foreigners (2)
Market	0; 1; 2	0	2	market is principally: local/national (0); national/regional (1); international (2)
Subsidy	0; 1; 2	0	2	Are subsidies (including hidden) provided to support the fishery?: no (o); somewhat (1); large subsidies (2).
Sociological analysis				
Socialization of fishing	0; 1; 2	2	0	fishers work as: individuals (0); families (1); community groups (2)
Fishing community growth	0; 1; 2	0	2	Growth over past ten years: <10% (0); 10-20% (1); >20% (2).
Fishing sector	0; 1; 2	0	2	households in fishing in the community: <1/3 (0); 1/3-2/3 (1); >2/3 (2)
Environmental knowledge	0; 1; 2	2	0	Level of knowledge about environmental issues and the fishery: none (0); some (1) ; lots (2)
Education level	0; 1; 2	2	0	education level compared to population average: below (0); at (1); above (2)
Conflict status	0; 1; 2	0	2	level of conflict with other sectors: none (0); some (1); lots (2)
Fisher influence	0; 1; 2	2	0	strength of direct fisher influence on actual fishery regulations: almost none (0); some (1); lots (2)
Fishing income	0; 1; 2	2	0	fishing income as % of total family income: <50%; 50-80%; >80%
Kin participation	0; 1; 2; 3; 4	4	0	do kin sell and/or process fish? None (0); very few relatives (1-2 people) (1); a few relatives (2); some relatives (3); many kin (4)
Technological analysis				
Trip length	days	Low	High	average days at sea per fishing trip
Landing sites	0; 1; 2; 3	0	3	are landing sites: dispersed (0); somewhat centralised (1); heavily centralised (2); distant (3)
Pre-sale processing	0; 1; 2	2	0	processing before sale, ex. gutting, filleting: none (0); some (1); lots (2)
Onboard handling	0; 1; 2; 3	3	0	none (0); some (ex. salting and boiling) (1); sophisticated (ex. flash freezing, champagne ice) (2); live tanks (3)

Gear	0; 1	0	1	gear is: passive (0); active (1)
Selective gear	0; 1; 2	2	0	device(s) in gear to increase selectivity? few (0); some (1); lots (2)
FADS	0; 0.5; 1	0	1	are FADS: not used (0); bait is used (0.5); used (1)
Vessel size	0; 1; 2; 3; 4	0	4	Average length of vessels: <5 m (0); 5-10 m (1); 10-15 (2); 15-20 (3); >20 (4)
Catching power	0; 1; 2: 3: 4	0		Have fishers altered gear and vessel to increase catching power over past 5 years?: No (0); very little (1); little (2); somewhat (3); a lot, rapid increase (4)
Gear side effects	0; 1; 2	0	2	Does gear have undesirable side effects (e.g. cyanide, dynamite, trawl); no (0); some (1); a lot (2).
Ethical analysis				
Adjacency and reliance	0; 1; 2; 3	3	0	geographical proximity & historical connection: not adjacent/no reliance (0); not adjacent/some reliance (1); adjacent/some reliance (2); adjacent/strong reliance (3)
Alternatives	0; 1; 2	2	0	alternatives to the fishery within community: none (0); some (1); lots (2)
Equity in entry to fishery	0; 1; 2	2	0	is entry based on traditional/historical access/harvests? not considered (0); considered (1); traditional indigenous fishery (2)
Just management	0; 1; 2; 3; 4	4	0	inclusion of fishers in management: none (0); consultations (1); co-mgmt/gov't leading (2); co-mgmt/comm. leading (3); genuine co-mgmt with all parties equal (4)
Influences – ethical formation	0; 1; 2; 3; 4	4	0	structures which could influence values: strong negative (0); some negative (1); neutral (2); some positive (3); strong positive (4)
Mitigation – habitat destruction	0; 1; 2; 3; 4	4	0	Attempts to mitigate damage to fish habitat: much damage (0); some damage (1); no ongoing damage or mitigation (2); some mitigation (3); much mitigation (4)
Mitigation – ecosystem depletion	0; 1; 2; 3; 4	4	0	Attempts to mitigate fisheries-induced ecosystem change: much damage (0); some damage (1); no damage or mitigation (2); some mitigation (3); much mitigation (4)
Illegal fishing	0; 1; 2	0	2	illegal catching/poaching/transshipments: none (0); some (1); lots (2)
Discards & wastes	0; 1; 2	0	2	discard and waste of fish: none (0); some (1); lots (2)

APPENDIX 3: SAMPLE SPSS OUTPUT FOR RAPFISH ORDINATION

Proximities

Case Processing Summary[a]

Cases					
Valid		Missing		Total	
N	Percent	N	Percent	N	Percent
99	100.0%	0	.0%	99	100.0%

a. Squared Euclidean Distance used

Alscal

ALSCAL is writing OUTFILE results to file:
C:\Rapfish\temp\alscout

Iteration history for the 2 dimensional solution (in squared distances)

Young's S-stress formula 1 is used.

Iteration	S-stress	Improvement
1	.43874	
2	.35340	.08534
3	.34529	.00812
4	.34481	.00048

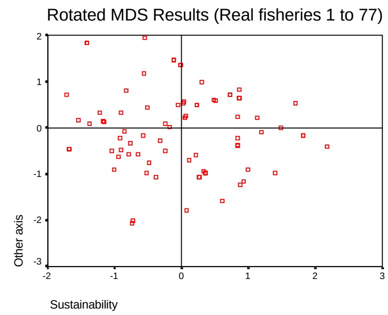
Iterations stopped because
S-stress improvement is less than .001000

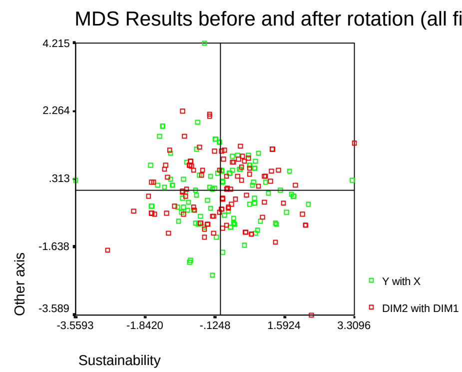
Stress and squared correlation (RSQ) in distances

RSQ values are the proportion of variance of the scaled data (disparities) in the partition (row, matrix, or entire data) which is accounted for by their corresponding distances. Stress values are Kruskal's stress formula 1.

For matrix
Stress = .24497 RSQ = .74925

Configuration derived in 2 dimensions





Stimulus Coordinates

Dimension

Stimulus Number	Stimulus Name	1	2
1	VAR1	.7328	.4510
2	VAR2	-.9132	-.0464
3	VAR3	-.8484	-.1851
4	VAR4	1.1107	.4060
5	VAR5	-1.3288	-.6745
6	VAR6	-1.3430	.7122
7	VAR7	-2.1192	-.6137
8	VAR8	.2634	.0300
9	VAR9	1.0947	.4070
10	VAR10	.4392	.4027
11	VAR11	-.4590	.4404
12	VAR12	-.4227	.5774
13	VAR13	1.2377	.2693
14	VAR14	-.9040	-.7056
15	VAR15	-1.2747	-1.2420
16	VAR16	-.6371	-.5813
17	VAR17	-1.2541	1.1322
18	VAR18	.3200	.7986
19	VAR19	1.8551	.1502
20	VAR20	.0283	1.1151
21	VAR21	.5341	.6478
22	VAR22	-.1327	1.1255
23	VAR23	.0901	.8986
24	VAR24	.9535	.1179
25	VAR25	-.0115	.5624
26	VAR26	-.1852	-.7545
27	VAR27	.7216	.6218
28	VAR28	-.8689	1.5581
29	VAR29	.5617	.9803
30	VAR30	1.4437	.5591
31	VAR31	-.7128	.6923
32	VAR32	.1722	.0643
33	VAR33	-.7559	.7078
34	VAR34	-.7559	.7078
35	VAR35	.1566	.3967
36	VAR36	.2745	-.4177
37	VAR37	1.3747	-1.4877
38	VAR38	-1.1104	-.4554
....			
61	VAR61	1.2852	1.1762
62	VAR62	-.4799	-.9509
63	VAR63	-.4799	-.9509
64	VAR64	.6338	-1.2093
65	VAR65	.6338	-1.2093
66	VAR66	.7680	-1.2567
67	VAR67	.7680	-1.2567
68	VAR68	2.0983	-1.0014
69	VAR69	2.0983	-1.0014
70	VAR70	-.7168	.8288
71	VAR71	-.7168	.8288
72	VAR72	.1691	-1.0162
73	VAR73	-.1570	-.7535
74	VAR74	-.3199	-.9705

Appendix 4a: Historical Development and Description of the Gulf of Maine Fisheries

A general pattern in the division of catch in the Gulf of Maine emerged early. Offshore, commercial, full time sectors targeting groundfish developed first. This fishery used hooks and lines to catch such species as cod, haddock, halibut, and flatfish. Inshore, a small-scale, part time fishery emerged in which farmers and coastal townsfolk took advantage of seasonally abundant fish and invertebrates to supplement both diet and income (Table A4.1) . Through the 1900s the character of the offshore fishery changed as hook and line gave way to trawls. In the inshore fisheries also fewer part time fishers were involved so that full time fishers dominated both sectors. These trends can be seen for both Maine and Massachusetts, although the local regions differ in mix of species that were landed (Figures A4.1 and A4.2). Such differences are a function of population and biogeography. For example, the existence of large population centres with appropriate port and market infrastructure such as Boston, Gloucester, and Portland, determined where the offshore fisheries would be based. The presence of huge lobster populations in the near shore areas of Maine, and far away from large commercial centres, however, has seen this industry develop through small scale operators in almost every coastal town of that state.

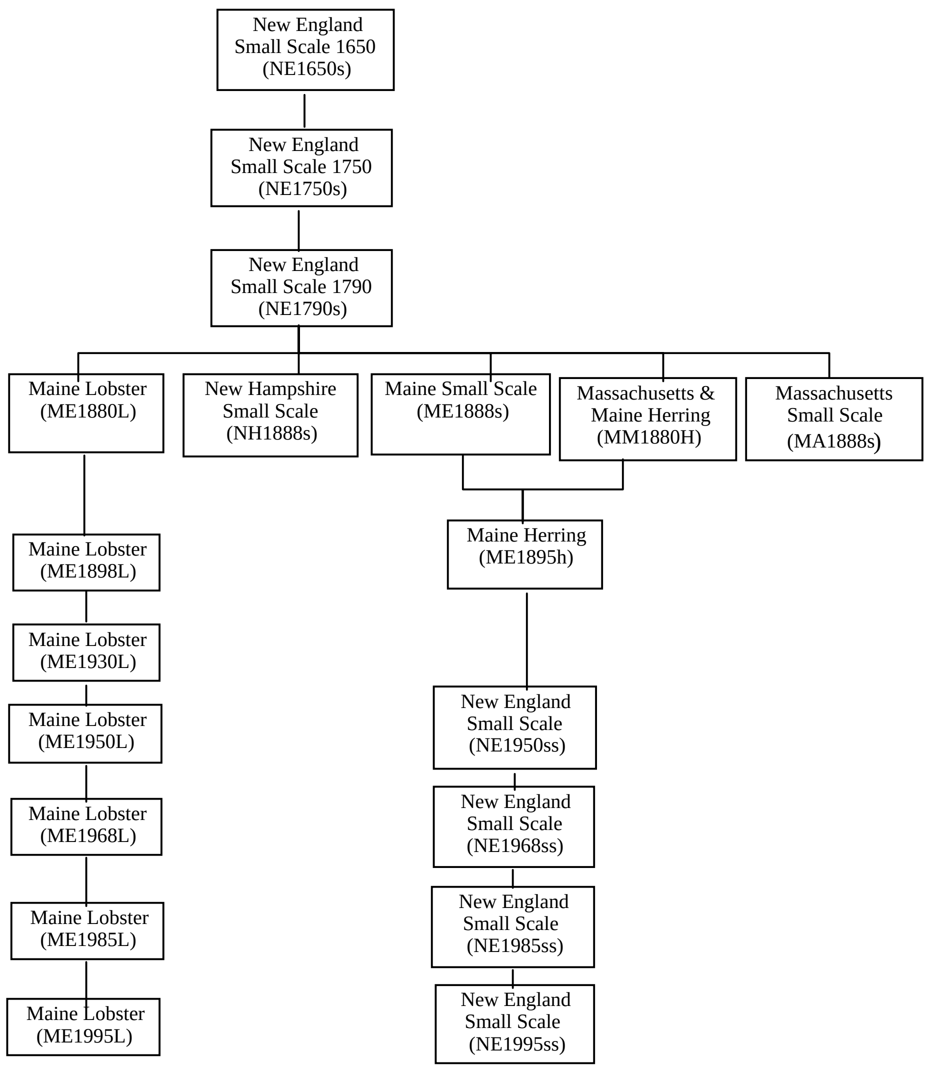


Figure A4.1. Historical development of the major small-scale Gulf of Maine fisheries used in the RAPFISH ordination.

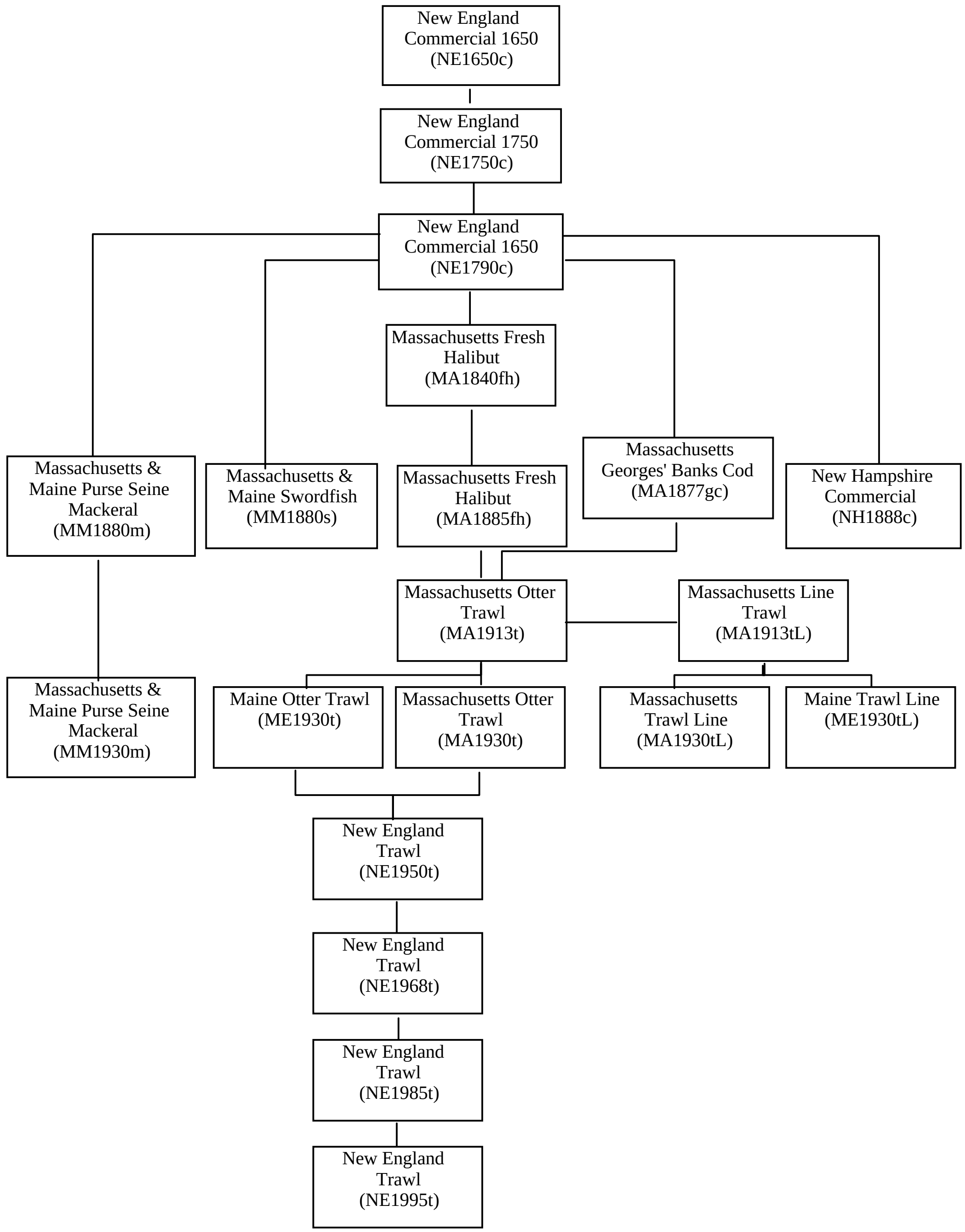


Figure A4.2: Historical development of the major large-scale commercial Gulf of Maine fisheries used in the RAPFISH ordination.

Table A4.1. Gulf of Maine fisheries in the RAPFISH ordination with their corresponding codes (as plotted on the figures) and brief notes on some of their features. Full details of the history and sources for these fisheries are provided in Appendix 4a and 5 respectively.

Definition	**Description**	**Code**
Small Scale & Inshore		
New England 1650 1750 1790	Gleaning, net and line fishery on seasonally available stocks such as herring, salmon, clams and lobsters	 NE1650s NE1750s NE1790s
New Hampshire 1888		NH1888s
Maine 1888		ME1888s
Massachusetts 1888		MA1888s
Main & Massachusetts Herring 1880 1895	 Seasonal weir fishery on spawning adults Weir fishery on small fish for 'anchovy' canneries	 MM1880h ME1895h
New England 1950 1968 1985 1995	Dredging, net, and line fisheries for seasonally available stocks such as groundfish, herring and clams	 NE1950ss NE1968ss NE1985ss NE1995ss
Maine Lobster 1880 1898 1930 1950 1968 1985 1995	Pot trap fishery	 ME1880L ME1898L ME1930L ME1950L ME1968L ME1985L ME1995L
Commercial & Offshore		
New England 1650 1750 1790	Longline ground fishery off large rigged vessels targeting groundfish especially cod and haddock	 NE1650c NE1750c NE1790c
New Hampshire 1880	Longline ground fishery from dories carried on large rigged vessels	NH1888c
Main & Massachusetts Swordfish 1880	Pelagic harpoon fishery from large rigged vessels, ceased to be important when the stocks were locally depleted. Fishing is now in international waters	MM1880s
Massachusetts Fresh Halibut 1840 1885	Longline ground fishery from dories carried on large rigged vessels	 MA1840fh MA1885fh
Main & Massachusetts Purse Seine Mackerel 1880 1930	 Purse seine pelagic fishery from mid sized rigged vessels Purse seine pelagic fishery from mid sized motor vessels	 MM1880m MM1930m
Massachusetts Georges' Bank Cod 1877	Longline ground fishery from dories carried on large rigged vessels	MA1877gc
Massachusetts Line Trawl 1913 1930	 Longline ground fishery from dories carried on large rigged vessels targeting cod and haddock	 MA1913tl MA1930tl
Maine Line Trawl 1930	Longline ground fishery from dories carried on large motor vessels catching cod and haddock	ME1930tl
Massachusetts Otter Trawl 1913 1930	 Otter trawl pelagic fishery from mid sized motor vessels catching codd and haddock	 MA1913t MA1930t
Maine Otter Trawl 1930	Otter trawl ground fishery from mid sized motor vessels catching cod and haddock	ME1930t
New England Trawl 1950 1968 1985 1995	 Otter and sterntrawl ground fishery from mid sized motor vessels Otter and sterntrawl fishery from mid sized motor vessels Otter and sterntrawl ground fishery from mid sized motor vessels catching cod, haddock, flounder, skates, dogfish and redfish	 NE1950t NE1968t NE1985t NE1995t

Appendix 4b: Historical Development of the German and United Kingdom Fisheries

The German fisheries in this study (Table A4.2) developed from fisheries that were established before the fourteenth century. Evolving from traditional small-scale deep-sea cutter fisheries targeting the same species, two offshore, commercial, full-time sectors catching pelagic and demersal fish were established. These fisheries used bottom and later mid-water trawls as well as long-lines to catch such species as cod, haddock, herring, and flatfish. Parallel to these and in increasingly heavier competition with the herring trawlers, a traditional lugger driftnet fishery for herring (salted directly onboard) operated in the Southern North Sea for centuries.

As the deep-sea fishery developed and expanded further offshore by improving boat and gear technology the species composition in the demersal fishery altered from cod and haddock to saithe and to some extent whiting and redfishes. Since the 1920s serial depletion forced German fisheries to spread out further and further and by 1975 most of the deep sea fleet was operating in distant waters and only a small proportion of all catches still stems from the North Sea. The catch was by saithe at that time. However, due to overexploitation of this species, today, the catch of Germany's deep-sea demersal fishery is made up of very small proportions of a variety of species with saithe only contributing 30% to the overall catch. Similarly, the proportion of herring in the catch of the pelagic trawling and the lugger fleet has declined steadily since the 1950s, when it made up more than 90% of the total catch. The economically struggling lugger fishery shifted to saithe in the 1970s, but was discontinued shortly after. Today, less than a third of the catches of the remaining pelagic trawling fleet are herring. Inshore, however, artisanal fisheries for oysters, shrimp, flatfish and herring were well established early this century but have since either ceased, declined or shifted to aquaculture production.

The traditional cutter deep-sea fleets once fished exclusively for flatfish, herring and sprat in the North Sea. Competition from large-scale commercial fisheries continuously diminished in importance of this fleet during the past century. The pelagic segment of this fishery temporarily experienced a revival when there was a huge expansion in the industrial fishing sector. By 1950 large numbers of juvenile herring and sprat were caught solely for the purpose of being processed into fishmeal. Due to the collapse of the herring stocks caused by recruitment overfishing, by 1975 this industrial cutter fishery was targeting mainly sand-eel and sprat and was discontinued around 1980. Although the deep-sea cutter fishery for flatfish and some demersal species is still on going, there is very little offshore fishing in the North Sea today.

The artisanal fisheries continued to develop into the nineteenth century and started to target mussel and hydrozoids (for ornaments) in addition to the traditional coastal species at the turn of the century. They diminished, however, greatly in importance with the onset of the industrialization and some fisheries ceased to operate (oysters and hydrozoid) or merged with the cutter deep-sea fleet targeting the same species (herring, sprat (in estuaries) and flatfish) between 1925 and 1950. The shrimp and mussel fisheries, formerly rather insignificant fisheries, are all that remain of the major coastal fisheries. In the 1950s and 1960s the shrimp fishery was threatened by serious recruitment over-fishing with 90% of the catch as juveniles processed into fish meal. Since then the focus has shifted back to mature shrimps caught for human consumption and today this fishery has been restored to highest valued fishery in Germany, surpassing the entire deep-sea fishing sector in economic importance. The current mussel fisheries have essentially turned into an aquaculture, which, however, still relies on the remaining wild mussel banks to harvest seedlings to be raised to maturity in mussel farms.

The United Kingdom fisheries in this study are commercial offshore fisheries and they have followed a similar trend to the German Fisheries. The English fisheries changes occurred at the same time as the German fisheries as boat and gear developed. Similar changes in the Scottish fisheries were delayed since the fishery was less centralised and more family based than the other countries. Today cod and haddock comprise much of the UK fisheries, however, there is concern with the sustainability of these fisheries. A beam trawl is used in this fishery today, the trawl produces a large amount of discards as well as catching juvenile plaice.

Table A4.2. German and United Kingdom North Sea fisheries evaluated using RAPFISH, together with their corresponding codes and brief notes on some of their features. Full details of the sources for these fisheries are provided in Appendix 4b and 5 respectively.

Fishery	Description	Code
GERMAN FISHERIES		
Deep Sea Demersal Fishery	initially cod, haddock and flatfishes primarily, used beam trawls and long-lines but changed to otter trawls in 1900s. Haddock and cod declined and whiting and saithe caught with otter trawls and long-lines. By 1950 mainly saithe, some herring and very little mackerel and cod caught. By 1975 fishing in distant waters and the catch is almost exclusively saithe. By 1997 redfish is an increasing component of the catch along with saithe, minor quantities of cod and haddock are taken. Today, these fisheries have decreased dramatically in importance.	GHS1880D, GHS1900D, GHS25D, GHS50D, GHS75D, GHS97D
Deep Sea Pelagic fishery	Primarily herring was found offshore in the North Sea using steamers with mid-water trawls. By 1975 the fishery moved mostly into distant waters. A decline of North Sea herring stocks resulted in a decrease of herring in the catch and by 1997 only 30% of the catch was herring, with mackerel (5%), horse mackerel (5%). Today, these fisheries have decreased dramatically in importance.	GHS25H, GHS50H, GHS75H, GHS97H
Cutter Deep Sea fishery - industrial	herring (including juvenile) and some sprat, full-time small scale fishers between the mainland and off-shore islands in the southern North Sea, Wadden sea and river estuaries used sailing cutters with specialised bottom trawls. By 1950 catch was primarily juvenile herring and some sprat, by trawling. Industrial fishing of "oil" herring commenced. By 1975 the catch was mainly sandeel and sprat with some juvenile herring. By 1997 these industrial fisheries are no longer operating.	CDS1880I, CDS1900I, CDS25I, CDS50I, CDS75I
Cutter Deep Sea fishery - flatfishes	mainly flounder, plaice, sole, lemon sole, etc between the mainland and off-shore islands in the southern North Sea, used sailing cutters with beam trawls. In 1900s demersals such as cod and haddock also caught in long-lines, otter trawls were also used. By 1925 most of the catch was plaice (over exploitation) with haddock, lemon sole and cod making up less than 12% of the catch. Long-lining for tuna occured during 1920s and by 1950s plaice declined with cod, whiting and tuna making up the rest of the catch. By 1975 the catch composition changed again with cod (50%), saithe (30%), haddock (7%) plaice(6%).	CDS1880F, CDS1900F, CDS25F, CDS50F, CDS75S, CDS97S
Lugger Herring Fishery	seasonal driftnet fishery targeted exclusively herring in the southern North Sea area, used sailing luggers, catch processed into the salt-herring. Until 1900s fishing grounds slowly expanded further out into the North Sea. With the start of the herring trawler fishery around the 1925, this fishery declined despite attempts to modernize the fleet.	GH1880S, GH1900S, GH25S, GH50S
Lugger Herring Fishery	evolved from the traditional lugger fishery in the 1950s, targeted fresh herring using mainly driftnets on a seasonal basis in the off-shore areas especially in the southern area of the North Sea. By 1975 most of the catch was saithe, The fishery continued to decline in profitability, and indeed ceased to exist a few years later.	GH50F, GH75F
Coastal fishery shrimps	Originally catching shrimps and some flatfish in passive weirs, on a casual basis. In 1900s and 1925 some full time fishing on a seasonal basis, immature shrimps were caught and small beam trawls dominated. By 1950 the fishers were full-time during the season, but over 90% of the catch was immature shrimp, this period corresponded to expansion of the fishery for juvenile shrimp for fish meal. With the decreasing profitability of the industrial fishery by 1975, fishery shifted back to focus on shrimps caught for human consumption, becoming in turn the most profitable of all German fisheries by 1997.	CF1880S, CF1900S, CF25S, CF50S, CF75S, CF97S
Coastal fishery mussels	originally on a casual basis in coastal and island areas in the Wadden Sea used manual dredges or collected at low tide. By 1925 larger fishing vessels with dredges were used and by 1950 fishing wass primarily full-time. By 1975 the industry was aquaculture based, and this fishery now collects juvenile animals for grow-out on farms.	CF1900M, CF25M, CF50M, CF75M, CF97M

Coastal fishery oysters	oysters, artisanal fishery on a seasonal basis, mainly inshore or off the islands areas in the Wadden sea fished used quite large fishing vessels with dredges. The fishery peaked in 1870s but due to over-exploitation the fishery ceased to exist after 1925.	CF1880O, CF1900O, CF25O
Coastal fishery hydrozoids	an ornamental artisinal fishery for hydrozaons and hydrallmania located in coastal or island areas in the Wadden Sea, used rakes and some set nets on a casual basis around 1880. Due to high profitability of fishery, fishing power expanded rapidly with larger dredge trawlers used. By 1925 the fishery declined and essentially ceased to exist around 1940.	CF1880H, CF1900H, CF25H
Coastal fishery estuary	an artisanal partially full-time mixed fishery for herring, sprat, sturgeon, salmon and eel located in coastal or river areas relying mainly on set nets and some small scale trawling gear. By the 1900s sturgeon and salmon depleted. By 1925 small scale trawls dominated and the focus shifted almost exclusively to herring and sprat (some eels).	CF1880R, CF1900R, CF25R
Coastal fishery flatfish	mainly flounder, plaice, sole, lemon sole, etc., artisanal fishers in inshore coastal areas and off-shore islands in the Wadden Sea and river estuaries, used small vessels with set nets, long-lines and beam trawls. By 1925 the fishery decreased in importance and most fishing was casual. By 1950 the fishery was no longer operating except offshore.	CF1880F, CF1900F, CF25F
UNITED KINGDOM		
1900s		
Scottish Line Fishery	haddock, cod, and plaice were caught in a local, small-scale fishery with strong family involvement. Eventually the fishery was out-competed by trawlers. After 1910 line fisheries faded in importance in Scotland.	SC10CL SC10PL SC10HAL
Scottish Trawl Fishery	trawl fleet catching cod, haddock, and plaice, with some boats fishing north of the North Sea to find less diminished stocks.	SC10CT SC10PT SC10HAT
Herring Drift Net Fishery	primarily herring especially in Scotland	SC10HD EN10HD
English Trawl Fishery	Primarily cod, haddock and plaice were caught	EN10CT EN10HAT EN10PT
1950s		
Herring Drift Net Fishery	The herring fishery had lost its importance and profitability in both England and Scotland by the 1950s.	SC55HD EN55HD
Trawl and Seine Net Fishery	World War II allowed the haddock, plaice, and cod stocks a brief recovery. By 1955, however, the catches had decreased to pre-war or close to pre-war levels.	EN55HATS SC55HATS EN55CTS SC55CTS EN55PTS SC55PTS
1990s		
Herring Pelagic Trawl and Purse Seine	Herring catches increased after the fishery closure (ending in the early 1980's) but were decreasing again by 1990. The herring fishery was of a lower importance, in the 1990s, due to industrial herring fisheries conducted by other countries.	EN90HTS SC90HTS
Trawl and Seine Fisheries	The cod and haddock stocks were severely over fished by the early 1990's. Cod and haddock comprised a large part of the UK fisheries.	EN90HATS SC90HATS EN90CTS SC90CTS
Plaice Beam Trawl Fishery	relatively efficient in catching plaice and other species, but a very destructive with a high level of by-catch. The plaice stock was not as depleted as that of the cod and haddock, but juvenile plaice were being caught and discarded	EN90PBT SC90PBT

APPENDIX 5(PART 1): ORIGINAL DATA USED FOR THE GULF OF MAINE FISHERIES IN THE FIVE 'RAPFISH' FIELDS. SOURCES OF SCORES ARE DOCUMENTED INL APPENDIX 11 AND DISCUSSED IN APPENDIX 4.

Fishery	Abbreviation	SOCIAL	socialisation of fishing	fishing community growth	fishing sector	environmental knowledge	education level	conflict status	fisher influence	fishing income	kin participation	TECHNOLOGICAL	trip length	landing sites	pre-sale processing	use of ice	gear	selective gear	power gear	FADS	Sonar	Average Boat Size	catching power	gear side effects	ETHICAL	Adjacency & Reliance	Alternatives	Equity in Entry	Just Management	Infl. in Ethical Formation	Mit. of Habitat Destruction	Mit. of Ecosystem Depl.	Illegal Fishing	Discards & wastes
New England mid 1600s small scale	NE1650s		1.0	2.0	2.0	0.0	0.0	1.0	2.0	1.0	1.0		0.5	1.0	1.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	1.0	0.0		2.0	1.0	0.0	3.0	3.0	1.0	0.0	0.0	2.0
New England mid 1600s commercial	NE1650c		0.0	2.0	2.0	0.0	0.0	1.0	2.0	2.0	0.0		30.0	2.0	2.0	0.0	1.0	2.0	0.0	0.5	0.5	1.0	1.0	0.0		2.0	1.0	0.0	3.0	3.0	1.0	0.0	0.0	2.0
New England mid 1700s small scale	NE1750s		1.0	2.0	1.0	1.0	0.0	2.0	1.0	1.0	1.0		0.5	1.0	1.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	2.0	0.0		3.0	0.0	0.0	2.0	3.0	1.0	1.0	2.0	1.0
New England mid 1700s commercial	NE1750c		0.0	2.0	1.0	1.0	1.0	2.0	1.0	2.0	0.0		22.0	2.0	2.0	0.0	1.0	1.0	0.0	0.5	0.5	2.0	2.0	1.0		3.0	1.0	0.0	2.0	3.0	1.0	1.0	2.0	1.0
New England late 1700s small scale	NE1790s		1.0	2.0	1.0	1.0	0.0	2.0	2.0	1.0	1.0		0.5	1.0	2.0	0.0	0.0	1.0	0.0	1.0	0.0	0.0	1.0	1.0		3.0	0.0	0.0	3.0	1.0	0.0	0.0	2.0	1.0
New England late 1700s commercial	NE1790c		0.0	2.0	1.0	1.0	0.0	2.0	2.0	2.0	0.0		22.0	2.0	2.0	0.0	1.0	1.0	0.0	0.5	0.5	2.0	1.0	1.0		3.0	0.0	0.0	3.0	2.0	0.0	0.0	1.0	1.0
Maine 1888 small scale	Me1888s		1.0	2.0	0.0	1.0	0.0	0.0	0.0	2.0	1.0		0.5	1.0	1.0	1.0	0.0	1.0	0.0	0.0	0.5	1.0	2.0	1.0		3.0	2.0	0.0	1.0	2.0	1.0	2.0	0.0	1.0
Massachussets 1888 small scale	Ma1888s		1.0	2.0	0.0	1.0	0.0	0.0	0.0	2.0	1.0		0.5	1.0	1.0	1.0	0.0	1.0	0.0	0.0	0.5	1.0	2.0	1.0		3.0	2.0	0.0	1.0	2.0	1.0	2.0	0.0	1.0
New Hampshire 1888 small scale	NH1888s		1.0	2.0	0.0	1.0	0.0	0.0	0.0	2.0	1.0		0.5	1.0	1.0	1.0	0.0	2.0	0.0	0.0	0.5	1.0	2.0	1.0		3.0	2.0	0.0	1.0	2.0	1.0	2.0	0.0	1.0
New Hampshire 1888 commercial	NH1888c		0.0	2.0	0.0	1.0	1.0	0.0	0.0	2.0	0.0		5.0	1.0	0.0	1.0	0.0	0.0	0.0	0.5	0.5	2.0	1.0	1.0		3.0	1.0	0.0	1.0	2.0	0.0	1.0	0.0	1.0
Massachusetts fresh halibut 1840	MA1840fh		0.0	2.0	0.0	1.0	1.0	0.0	0.0	2.0	0.0		5.0	2.0	0.5	1.0	0.0	1.0	0.0	0.5	0.5	1.0	2.0	0.0		3.0	2.0	0.0	1.0	2.0	2.0	0.0	0.0	1.0
Massachusetts fresh halibut 1885	MA1885fh		0.0	2.0	0.0	2.0	2.0	0.0	0.0	2.0	0.0		28.0	2.0	1.5	2.0	0.0	1.0	0.0	0.5	0.5	2.0	2.0	0.0		3.0	2.0	0.0	1.0	0.0	2.0	0.0	0.0	1.0
Massachusetts George's Bank cod late 1870's	MA1877gc		0.0	2.0	0.0	1.0	0.0	0.0	0.0	2.0	0.0		7.0	2.0	1.0	1.0	0.0	1.0	0.0	0.5	0.5	1.0	2.0	0.0		3.0	1.0	0.0	1.0	2.0	2.0	1.0	0.0	1.0
Massachusstets + Maine purse seine mackerel 1880	MM1880m		0.0	2.0	0.0	1.0	1.0	2.0	1.0	2.0	0.0		1.0	2.0	1.0	1.0	1.0	0.0	0.0	1.0	0.5	2.0	2.0	1.0		3.0	1.0	0.0	2.0	1.0	0.0	0.0	0.0	2.0
Massachussets + Maine swordfish 1880	MM1880s		0.0	2.0	0.0	2.0	1.0	0.0	0.0	2.0	0.0		12.0	2.0	0.5	0.0	1.0	2.0	0.0	0.0	0.0	1.0	0.0	0.0		3.0	1.0	0.0	1.0	3.0	2.0	2.0	0.0	0.0
Massachussets + Maine herring 1880	MM1880h		2.0	2.0	0.0	1.0	0.0	1.0	0.0	2.0	0.0		0.5	0.0	0.5	0.0	0.0	1.0	0.0	1.0	0.0	1.0	0.0	1.0		3.0	1.0	1.0	1.0	1.0	1.0	0.0	0.0	0.0
Maine 1895 herring	ME1895h		1.0	2.0	2.0	1.0	0.0	2.0	0.0	2.0	0.0		0.5	0.0	0.5	0.0	0.0	1.0	0.0	1.0	0.0	0.0	2.0	1.0		3.0	0.0	1.0	1.0	1.0	0.0	0.0	0.0	1.0
Maine 1880 lobster	ME 1880l		1.0	2.0	2.0	1.0	1.0	0.0	0.0	0.0	0.0		0.5	0.0	0.0	0.0	0.0	2.0	0.0	0.5	0.0	0.0	1.0	0.0		3.0	1.0	0.0	1.0	3.0	0.0	1.0	1.0	0.0
Maine 1898 lobster	ME 1898l		1.0	2.0	2.0	1.0	1.0	0.0	0.0	2.0	0.0		0.5	0.0	0.0	0.0	0.0	2.0	0.0	0.5	0.0	0.0	1.0	0.0		3.0	1.0	0.0	1.0	1.0	0.0	2.0	1.0	0.0
Massachusstets otter trawl fisheries 1913	MA1913t		0.0	1.0	0.0	1.0	1.0	2.0	1.0	2.0	0.0		5.0	2.0	0.5	2.0	1.0	0.0	1.0	0.0	0.5	2.0	2.0	2.0		2.0	2.0	0.0	1.0	1.0	0.0	0.0	0.0	2.0
Massachusetts trawl line fishery 1913	MA1913tl		0.0	1.0	0.0	1.0	1.0	2.0	0.0	2.0	0.0		7.0	2.0	1.0	1.0	0.0	1.0	0.0	0.5	0.5	2.0	1.0	1.0		3.0	2.0	1.0	2.0	2.0	2.0	2.0	0.0	1.0
Massachusetts + Maine purse seine mackerel 1930	MM1930m		0.0	1.0	0.0	1.0	1.0	1.0	0.0	2.0	0.0		1.0	1.0	0.0	1.0	1.0	0.0	1.0	0.0	0.0	1.0	0.0	1.0		2.0	2.0	0.0	1.0	2.0	1.0	1.0	0.0	1.0
Maine lobster 1930	ME1930l		1.0	1.0	1.0	2.0	1.0	0.0	1.0	2.0	1.0		0.5	0.0	0.0	0.0	0.0	2.0	0.0	0.5	0.5	0.0	0.0	0.0		3.0	1.0	1.0	2.0	1.0	2.0	1.0	2.0	1.0
Maine trawl line 1930	ME1930tl		0.0	1.0	0.0	1.0	1.0	1.0	1.0	2.0	0.0		6.9	2.0	0.0	1.0	0.0	1.0	0.0	0.5	0.5	2.0	0.0	1.0		3.0	1.0	0.0	2.0	3.0	2.0	2.0	0.0	1.0
Maine otter trawl 1930	ME1930t		0.0	1.0	1.0	1.0	1.0	1.0	2.0	2.0	0.0		7.8	2.0	0.5	1.0	1.0	0.0	1.0	0.0	0.5	2.0	2.0	2.0		3.0	1.0	0.0	2.0	3.0	0.0	0.0	0.0	2.0
Massachusetts trawl line 1930	MA1930tl		0.0	1.0	0.0	1.0	1.0	1.0	1.0	2.0	0.0		6.9	2.0	0.0	1.0	0.0	1.0	0.0	0.5	0.5	2.0	0.0	1.0		3.0	2.0	0.0	2.0	3.0	2.0	2.0	0.0	1.0
Massachusetts otter trawl 1930	MA1930t		0.0	1.0	0.0	1.0	1.0	1.0	2.0	2.0	0.0		7.8	2.0	0.5	1.0	1.0	0.0	1.0	0.0	0.5	2.0	2.0	2.0		3.0	2.0	0.0	2.0	3.0	0.0	0.0	0.0	2.0
New England inshore 1950	NE1950ss		1.0	1.0	0.0	2.0	1.0	1.0	1.0	1.0	0.0		0.5	1.0	0.5	1.0	0.5	1.0	0.0	0.5	0.5	1.0	0.0	1.0		2.0	1.0	1.0	2.0	3.0	1.0	1.0	0.0	1.0
New England trawl 1950	NE1950t		0.0	1.0	0.0	1.0	1.0	1.0	2.0	2.0	0.0		8.0	2.0	0.5	2.0	1.0	0.0	1.0	0.0	2.0	2.0	1.0	2.0		2.0	2.0	0.0	2.0	1.0	0.0	0.0	0.0	2.0
Maine lobster 1950	ME1950l		1.0	1.0	1.0	2.0	1.0	0.0	1.0	2.0	0.0		0.5	0.0	0.0	1.0	0.0	2.0	1.0	0.5	1.0	1.0	1.0	0.0		3.0	2.0	1.0	2.0	4.0	2.0	3.0	0.0	0.0
New England inshore 1968	NE1968ss		1.0	1.0	0.0	2.0	1.0	2.0	2.0	1.0	0.0		0.5	1.0	0.5	1.0	0.5	1.0	0.0	0.5	0.5	1.0	0.0	1.0		2.0	1.0	1.0	2.0	3.0	1.0	2.0	0.0	1.0
New England Trawl 1968	NE1968t		0.0	1.0	0.0	1.0	1.0	2.0	2.0	2.0	0.0		8.0	2.0	0.5	2.0	1.0	0.0	1.0	0.0	2.0	2.0	1.0	2.0		1.0	2.0	0.0	3.0	0.0	0.0	0.0	0.0	2.0
Maine lobster 1968	ME1968l		1.0	1.0	1.0	2.0	1.0	1.0	2.0	2.0	0.0		0.5	0.0	0.0	1.0	0.0	2.0	1.0	0.5	1.0	1.0	0.0	0.0		2.0	2.0	1.0	3.0	4.0	2.0	3.0	0.0	0.0
New England inshore 1985	NE1985ss		0.0	1.0	0.0	2.0	1.0	2.0	2.0	2.0	0.0		0.5	1.0	1.0	1.0	0.5	2.0	1.0	0.5	1.0	1.0	2.0	2.0		1.0	2.0	1.0	3.0	4.0	2.0	3.0	0.0	0.0
New England Trawl 1985	NE1985t		0.0	1.0	0.0	1.0	2.0	2.0	2.0	2.0	0.0		8.0	2.0	1.0	2.0	1.0	0.0	1.0	0.0	1.0	1.0	2.0	2.0		1.0	2.0	1.0	3.0	3.0	0.0	3.0	0.0	1.0
Maine lobster 1985	ME1985l		0.0	1.0	1.0	2.0	1.0	0.0	2.0	2.0	0.0		0.5	0.0	0.0	1.0	0.0	2.0	1.0	0.5	1.0	1.0	1.0	0.0		2.0	1.0	1.0	2.0	3.0	2.0	2.0	0.0	0.0
New England inshore 1995	NE1995ss		0.0	0.0	0.0	2.0	1.0	2.0	1.0	2.0	0.0		0.5	2.0	1.0	1.0	0.5	2.0	1.0	0.5	1.0	1.0	1.0	2.0		1.0	2.0	1.0	2.0	4.0	2.0	4.0	0.0	0.0
New England Trawl 1995	NE1995t		0.0	0.0	0.0	2.0	2.0	2.0	1.0	2.0	0.0		8.0	2.0	1.0	2.0	1.0	0.0	1.0	0.0	1.0	1.0	1.0	2.0		1.0	2.0	1.0	2.0	3.0	3.0	4.0	0.0	1.0
Maine lobster 1995	ME1995l		0.0	0.0	1.0	2.0	1.0	0.0	2.0	2.0	0.0		0.5	0.0	0.0	1.0	0.0	2.0	1.0	0.5	1.0	1.0	0.0	0.0		2.0	1.0	1.0	4.0	3.0	3.0	2.0	0.0	0.0
Good	G		2.0	2.0	2.0	2.0	2.0	0.0	2.0	2.0	4.0		0.5	0.0	2.0	3.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		3.0	2.0	2.0	4.0	4.0	4.0	4.0	0.0	0.0
Bad	B		0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0		30.0	3.0	0.0	0.0	1.0	1.0	1.0	1.0	2.0	2.0	2.0	2.0		0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.0	2.0

APPENDIX 5 (PART 2): ORIGINAL DATA USED FOR THE GULF OF MAINE FISHERIES IN THE FIVE 'RAPFISH' FIELDS.

Fishery	Abbreviation	ECOLOGICAL	exploitation status	recruitment variability	change in T level	migratory range	range collapse	size of fish caught	catch < maturity	discarded bycatch	species caught	primary production	ECONOMIC	Price	fisheries in GDP	relative income	limited entry	marketable right	other income	ownership	market	subsidy
land mid 1600s small scale	NE1650s		0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	1.0	3.0		0.0	1.0	1.0	1.0	0.0	2.0	1.0	0.0	0.0
land mid 1600s commercial	NE1650c		0.0	2.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	3.0		1.0	1.0	1.0	1.0	0.0	3.0	2.0	2.0	0.0
land mid 1700s small scale	NE1750s		0.0	0.0	0.0	0.0	1.0	1.0	0.0	2.0	1.0	3.0		1.0	2.0	1.0	0.0	0.0	2.0	0.0	0.0	0.0
land mid 1700s commercial	NE1750c		3.0	2.0	0.0	0.0	1.0	1.0	1.0	2.0	0.0	3.0		2.0	2.0	1.0	1.0	0.0	3.0	0.0	2.0	2.0
land late 1700s small scale	NE1790s		1.0	0.0	2.0	0.0	1.0	1.0	2.0	1.0	1.0	3.0		1.0	2.0	1.0	0.0	0.0	2.0	0.0	1.0	0.0
land late 1700s commercial	NE1790c		2.0	2.0	1.0	0.0	1.0	1.0	1.0	1.0	0.0	3.0		1.0	2.0	1.0	0.0	0.0	3.0	0.0	2.0	2.0
88 small scale	Me1888s		1.0	0.0	1.0	1.0	2.0	0.0	0.0	0.0	1.0	3.0		1.0	2.0	0.0	0.0	0.0	3.0	0.0	0.0	0.0
ssets 1888 small scale	Ma1888s		1.0	0.0	1.0	1.0	2.0	0.0	0.0	0.0	1.0	3.0		1.0	2.0	0.0	0.0	0.0	3.0	0.0	0.0	0.0
pshire 1888 small scale	NH1888s		1.0	0.0	1.0	1.0	2.0	0.0	0.0	0.0	1.0	3.0		1.0	0.0	0.0	0.0	0.0	3.0	0.0	0.0	0.0
pshire 1888 commercial	NH1888c		1.0	2.0	0.0	1.0	0.0	0.0	0.0	1.0	1.0	3.0		2.0	0.0	1.0	0.0	0.0	3.0	0.0	1.0	0.0
setts fresh halibut 1840	MA1840fh		0.0	2.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	3.0		1.0	2.0	2.0	1.0	0.0	3.0	0.0	0.0	0.0
setts fresh halibut 1885	MA1885fh		2.0	2.0	0.0	1.0	2.0	1.0	0.0	1.0	0.0	3.0		3.0	2.0	2.0	2.0	0.0	3.0	0.0	1.0	0.0
setts George's Bank cod late 1870's	MA1877gc		2.0	2.0	0.0	1.0	0.0	0.0	0.0	1.0	0.0	3.0		1.0	2.0	1.0	0.0	0.0	3.0	0.0	1.0	0.0
sstets + Maine purse seine mackerel 1880	MM1880m		2.0	2.0	0.0	1.0	1.0	0.0	0.0	2.0	0.0	3.0		2.0	2.0	1.0	0.0	0.0	3.0	0.0	1.0	0.0
ssets + Maine swordfish 1880	MM1880s		0.0	1.0	0.0	2.0	0.0	0.0	0.0	0.0	0.0	3.0		2.0	2.0	1.0	0.0	0.0	3.0	0.0	0.0	0.0
ssets + Maine herring 1880	MM1880h		3.0	2.0	0.0	1.0	1.0	1.0	2.0	1.0	0.0	3.0		2.0	2.0	1.0	1.0	0.0	3.0	0.0	0.0	0.0
95 herring	ME1895h		3.0	2.0	1.0	1.0	1.0	1.0	2.0	1.0	0.0	3.0		0.0	2.0	1.0	2.0	1.0	2.0	0.0	1.0	0.0
80 lobster	ME 1880l		2.0	2.0	0.0	0.0	0.0	1.0	1.0	0.0	0.0	3.0		1.0	2.0	1.0	1.0	0.0	2.0	0.0	1.0	0.0
98 lobster	ME 1898l		3.0	2.0	0.0	0.0	1.0	1.0	1.0	0.0	0.0	3.0		4.0	2.0	1.0	1.0	0.0	2.0	0.0	1.0	0.0
sstets otter trawl fisheries 1913	MA1913t		3.0	1.0	2.0	1.0	2.0	1.0	2.0	2.0	1.0	3.0		1.0	1.0	2.0	1.0	0.0	3.0	0.0	1.0	0.0
setts trawl line fishery 1913	MA1913tl		3.0	1.0	1.0	1.0	2.0	1.0	2.0	1.0	1.0	3.0		1.0	1.0	2.0	1.0	0.0	3.0	0.0	1.0	0.0
setts + Maine purse seine mackerel 1930	MM1930m		3.0	2.0	0.0	1.0	1.0	0.0	1.0	1.0	0.0	3.0		1.0	0.0	2.0	1.0	0.0	3.0	0.0	1.0	0.0
ster 1930	ME1930l		2.0	2.0	0.0	0.0	1.0	2.0	1.0	0.0	0.0	3.0		5.0	2.0	0.0	1.0	1.0	2.0	0.0	0.0	0.0
wl line 1930	ME1930tl		2.0	1.0	0.0	1.0	0.0	1.0	1.0	1.0	1.0	3.0		1.0	1.0	2.0	2.0	0.0	3.0	0.0	1.0	0.0
er trawl 1930	ME1930t		3.0	1.0	1.0	1.0	1.0	1.0	2.0	2.0	1.0	3.0		1.0	1.0	2.0	1.0	0.0	3.0	0.0	1.0	0.0
setts trawl line 1930	MA1930tl		2.0	1.0	0.0	1.0	0.0	1.0	1.0	1.0	1.0	3.0		1.0	0.0	2.0	2.0	0.0	3.0	0.0	1.0	0.0
setts otter trawl 1930	MA1930t		3.0	1.0	1.0	1.0	1.0	1.0	2.0	2.0	1.0	3.0		1.0	0.0	2.0	1.0	0.0	3.0	0.0	1.0	0.0
land inshore 1950	NE1950ss		2.0	1.0	0.0	0.0	0.0	0.0	1.0	1.0	1.0	3.0		0.0	0.0	1.0	1.0	0.0	1.0	0.0	0.0	0.0
land trawl 1950	NE1950t		3.0	2.0	0.0	1.0	1.0	1.0	1.0	2.0	0.0	3.0		1.0	0.0	2.0	2.0	0.0	3.0	0.5	1.0	2.0
ster 1950	ME1950l		1.0	2.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.0		5.0	1.0	1.0	2.0	0.0	3.0	0.0	1.0	1.0
land inshore 1968	NE1968ss		3.0	1.0	1.0	0.0	1.0	1.0	1.0	1.0	1.0	3.0		2.0	0.0	0.0	1.0	1.0	1.0	0.0	0.0	1.0
land Trawl 1968	NE1968t		3.0	2.0	2.0	1.0	2.0	1.0	1.0	2.0	0.0	3.0		2.0	0.0	1.0	2.0	1.0	3.0	0.5	1.0	2.0
ster 1968	ME1968l		2.0	2.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	3.0		5.0	1.0	1.0	2.0	1.0	3.0	0.0	1.0	2.0
land inshore 1985	NE1985ss		3.0	1.0	1.0	1.0	2.0	2.0	1.0	1.0	1.0	3.0		3.0	0.0	1.0	2.0	0.0	3.0	0.0	1.0	2.0
land Trawl 1985	NE1985t		3.0	2.0	1.0	1.0	2.0	2.0	2.0	2.0	0.0	3.0		3.0	0.0	1.0	2.0	0.0	3.0	0.0	1.0	2.0
ster 1985	ME1985l		1.0	2.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.0		5.0	1.0	1.0	1.0	0.0	3.0	0.0	1.0	1.0
land inshore 1995	NE1995ss		3.0	1.0	1.0	1.0	0.0	1.0	1.0	1.0	1.0	3.0		3.0	0.0	0.0	2.0	2.0	3.0	0.0	1.0	2.0
land Trawl 1995	NE1995t		3.0	2.0	2.0	1.0	0.0	1.0	1.0	1.0	0.0	3.0		3.0	0.0	0.0	2.0	2.0	3.0	0.0	1.0	2.0
ster 1995	ME1995l		2.0	2.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	3.0		5.0	1.0	0.0	2.0	2.0	3.0	0.0	1.0	1.0
	G		0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.0		5.0	2.0	4.0	2.0	2.0	0.0	0.0	0.0	0.0
	B		3.0	2.0	2.0	2.0	2.0	2.0	2.0	2.0	2.0	0.0		0.0	0.0	0.0	0.0	0.0	3.0	2.0	2.0	2.0

Appendix 6: Results of the MDS Rapfish ordination on the five evaluation fields for the 38 Gulf of Maine fisheries. Values for sustainability axes only, as percentage of the best possible.

Fishery	ethical	social	economic	technological	ecological
New England mid 1600s small scale	44.8	15.5	36.7	69.9	91.3
New England mid 1600s commercial	44.8	31.2	14.8	33.1	93.3
New England mid 1700s small scale	40.9	27.1	47.6	69.7	80.2
New England mid 1700s commercial	46.2	47.4	32.7	26.1	67.8
New England late 1700s small scale	29.1	31.5	43.3	69.5	61.5
New England late 1700s commercial	37.8	44.7	25.2	28.3	71.4
Maine 1888 small scale	48.8	27.7	40.0	61.0	85.6
Massachussets 1888 small scale	48.8	27.7	40.0	61.0	85.6
New Hampshire 1888 small scale	48.8	27.7	37.3	64.7	85.6
New Hampshire 1888 commercial	41.6	43.9	37.5	58.1	82.1
Massachusetts fresh halibut 1840	44.6	43.9	48.4	57.0	93.8
Massachusetts fresh halibut 1885	35.5	68.2	49.7	26.5	73.6
Massachusetts George's Bank cod late 1870s	45.9	35.4	38.3	55.5	83.7
Massachusstets + Maine purse seine mackerel 1880	34.9	50.5	40.1	32.6	79.2
Massachussets + Maine swordfish 1880	54.1	57.1	45.0	52.6	98.8
Massachussets + Maine herring 1880	42.7	32.1	47.2	75.0	64.4
Maine 1895 herring	32.6	28.3	52.5	76.4	63.1
Maine 1880 lobster	43.1	21.2	45.3	77.8	83.9
Maine 1898 lobster	37.9	34.2	51.2	77.8	77.9
Massachusstets otter trawl fisheries 1913	31.9	56.8	42.4	35.7	48.1
Massachusetts trawl line fishery 1913	57.8	53.0	42.4	51.3	57.2
Massachusetts + Maine purse seine mackerel 1930	45.6	52.4	39.8	64.0	75.1
Maine lobster 1930	47.7	63.2	63.4	79.6	75.8
Maine trawl line 1930	53.5	54.7	43.6	54.5	70.2
Maine otter trawl 1930	41.4	55.1	42.4	32.9	56.7
Massachusetts trawl line 1930	56.5	54.7	40.0	54.5	70.3
Massachusetts otter trawl 1930	44.5	58.7	39.8	32.9	56.7
New England inshore 1950	52.2	59.2	48.9	59.1	79.3
New England trawl 1950	35.2	58.7	34.4	31.5	65.4
Maine lobster 1950	70.3	60.7	47.8	72.9	93.1
New England inshore 1968	53.9	69.5	57.3	58.9	66.4
New England Trawl 1968	25.7	61.3	40.2	31.6	57.9
Maine lobster 1968	71.4	64.0	51.8	77.2	87.5
New England inshore 1985	72.9	70.1	38.2	52.6	55.1
New England Trawl 1985	64.7	73.4	38.2	35.9	48.4
Maine lobster 1985	58.7	64.2	45.7	72.7	93.1
New England inshore 1995	73.3	75.8	52.0	48.5	66.2
New England Trawl 1995	68.6	83.8	52.0	37.9	66.5
Maine lobster 1995	71.9	75.3	59.4	77.1	87.5

APPENDIX 7 (PART 1): ORIGINAL DATA USED FOR THE GERMAN NORTH SEA FISHERIES IN THE FIVE 'RAPFISH' FIELDS. SOURCES OF SCORES ARE DOCUMENTED IN APPENDIX 5 AND DISCUSSED IN APPENDIX 4.

Fishery	Abbreviation	**Ecological**	Exploitation status	Recruitment variability	Change in trophic level	Migratory range	Range collapse	Size of fish caught	Catch before maturity	Discarded by-catch	Species caught	Primary production	**Economic**	Profitability	Fisheries in GDP	Average Income	Management Regime	Other income	Sector employment	Ownership	Market	Subsidy
1880 Deep Sea Demersal Fishery	GHS1880D		3.0	1.0	0.0	2.0	1.0	1.5	1.5	1.0	0.5	1.5		4.0	0.0	3.5	0.5	3.0	0.0	1.0	2.0	0.5
1880 Lugger Herring Fishery (salted)	GH1880S		2.0	2.0	0.0	2.0	0.0	0.0	0.5	0.5	0.0	1.5		1.8	0.1	1.0	0.5	2.0	1.0	1.0	1.0	1.5
1880 Cutter Deep Sea Fishery - industrial	CDS1880I		1.5	2.0	0.0	2.0	0.0	0.0	1.0	0.5	1.5	1.5		1.5	0.2	1.0	0.5	2.5	1.0	0.0	1.0	0.0
1880 Cutter Deep Sea Fishery - flatfish	CDS1880F		3.0	1.0	0.0	2.0	1.5	2.0	1.5	1.0	1.5	1.5		1.5	0.2	1.0	0.5	2.5	1.0	0.0	2.0	0.0
1880 Coastal Fishery - shrimps	CF1880S		0.5	2.0	0.0	0.8	0.5	0.0	0.5	0.0	0.5	1.5		2.0	0.0	0.5	0.5	0.5	0.5	0.0	0.0	0.0
1880 Coastal Fishery - oysters	CF1880O		3.0	2.0	0.0	0.5	2.0	0.0	0.0	0.0	0.0	1.5		0.0	0.1	3.0	2.0	3.0	0.5	0.5	2.0	0.0
1880 Coastal Fishery - hydrozoids	CF1800H		0.0	2.0	0.0	0.5	0.0	0.0	0.0	0.5	0.0	1.5		1.0	0.0	3.0	0.0	0.0	0.0	0.0	2.0	0.0
1880 Coastal Fishery - estuary	CF1880R		2.5	1.5	0.0	2.0	1.0	1.5	1.0	0.5	1.5	1.5		1.5	0.1	1.0	1.0	2.0	1.0	0.0	0.0	0.5
1880 Coastal Fishery -flatfish	CF1880F		3.0	1.0	0.0	2.0	1.5	2.0	1.5	1.0	1.5	1.5		1.0	0.1	1.0	3.0	1.5	1.0	0.0	0.0	0.5
1900 Deep Sea Demersal Fishery	GHS1900D		3.0	1.0	0.0	2.0	2.0	1.5	1.5	1.0	0.5	1.5		2.5	0.3	3.0	0.5	3.0	2.0	1.0	2.0	2.0
1900 Lugger Herring Fishery (salted)	GH1900S		2.5	2.0	0.0	2.0	1.5	1.0	0.5	0.5	0.0	1.5		2.0	0.2	1.0	0.5	2.0	1.0	1.0	1.0	1.5
1900 Cutter Deep Sea Fishery - industrial	CDS1900I		2.5	2.0	0.0	2.0	1.8	1.0	1.0	0.5	1.0	1.5		3.0	0.1	1.0	0.5	2.8	0.8	0.0	1.0	1.0
1900 Cutter Deep Sea Fishery - flatfish	CDSH1900F		3.0	1.0	0.0	2.0	1.5	2.0	2.0	1.0	1.0	1.5		3.0	0.1	1.0	0.5	2.8	0.8	0.0	2.0	1.0
1900 Coastal Fishery - shrimps	CF1900S		1.5	2.0	0.0	0.8	1.5	0.0	0.0	0.5	0.8	1.5		2.0	0.1	0.5	0.5	1.5	0.5	0.0	0.5	1.0
1900 Coastal Fishery - mussels	CF1900M		0.0	2.0	0.0	0.5	0.0	0.0	1.0	0.0	1.0	1.5		2.0	0.0	0.0	0.0	0.2	0.0	0.0	2.0	0.0
1900 Coastal Fishery - oysters	CF1900O		3.0	2.0	0.0	0.5	1.5	0.0	0.0	1.0	1.0	1.5		2.0	0.0	2.0	2.0	2.0	0.0	0.5	2.0	0.0
1900 Coastal Fishery - hydrozoids	CF1900H		2.0	2.0	0.0	0.5	2.0	2.0	0.0	1.0	0.0	1.5		1.5	0.0	2.0	0.0	2.0	0.1	0.0	2.0	0.0
1900 Coastal Fishery - estuary	CF1900R		2.5	2.0	0.0	2.0	2.0	1.8	1.0	0.5	1.0	1.5		3.0	0.1	0.5	1.0	2.3	0.8	0.0	0.0	0.5
1900 Coastal Fishery -flatfish	CF1900F		3.0	1.0	0.0	2.0	2.0	2.0	2.0	1.0	1.3	1.5		3.0	0.1	0.5	3.0	1.5	0.8	0.0	0.0	0.5
1925 Deep Sea Pelagic Fishery	GHS25H		2.5	2.0	0.5	2.0	2.0	2.0	1.0	0.0	0.0	1.5		1.5	0.5	2.5	0.5	3.0	2.0	1.0	1.0	2.0
1925 Deep Sea Demersal Fishery	GHS25D		3.0	1.0	0.5	2.0	1.5	1.5	1.5	1.5	1.0	1.5		3.0	0.4	2.5	1.5	3.0	2.0	1.0	1.0	2.0
1925 Lugger Herring Fishery (salted)	GH25S		2.5	2.0	0.5	2.0	1.5	1.5	0.5	0.5	0.0	1.5		3.0	0.1	1.5	0.5	2.0	1.0	1.0	1.0	1.3
1925 Cutter Deep Sea Fishery - industrial	CDS25I		2.5	2.0	0.5	2.0	1.8	2.0	1.5	0.5	0.5	1.5		2.0	0.0	1.5	0.5	3.0	0.5	0.5	1.0	1.0
1925 Cutter Deep Sea Fishery - flatfish	CDSH25F		3.0	1.0	0.5	2.0	1.5	2.0	2.0	1.0	1.0	1.5		1.8	0.0	1.5	1.5	3.0	0.5	0.5	0.5	1.0
1925 Coastal Fishery - shrimps	CF25S		2.0	2.0	0.5	0.8	1.5	1.0	1.0	1.0	1.0	1.5		3.0	0.1	1.0	0.5	2.0	1.0	0.0	0.5	1.0
1925 Coastal Fishery - mussels	CF25M		1.0	2.0	0.5	0.5	0.5	1.0	1.0	1.0	1.0	1.5		2.0	0.0	0.0	0.8	0.1	0.0	0.0	2.0	0.0
1925 Coastal Fishery - oysters	CF25O		3.0	2.0	0.5	0.5	2.0	2.0	1.0	1.0	1.0	1.5		4.0	0.0	0.0	2.0	0.5	0.0	0.0	2.0	0.0
1925 Coastal Fishery - hydrozoids	CF25H		3.0	2.0	0.5	0.5	2.0	2.0	0.0	1.0	0.0	1.5		3.0	0.0	0.0	1.5	3.0	0.1	0.0	2.0	0.0
1925 Coastal Fishery - estuary	CF25R		2.5	2.0	0.5	2.0	2.0	2.0	2.0	0.5	0.5	1.5		2.5	0.0	1.0	0.5	2.5	0.5	0.0	1.0	1.0
1925 Coastal Fishery -flatfish	CF25F		3.0	1.0	0.5	2.0	2.0	2.0	2.0	1.0	1.3	1.5		3.0	0.0	1.0	2.0	2.0	0.5	0.0	0.5	1.0
1950 Deep Sea Pelagic Fishery	GHS50H		2.8	2.0	0.5	2.0	2.0	1.5	1.0	0.3	0.0	1.5		1.5	0.5	2.0	1.0	3.0	2.0	1.3	1.0	2.0
1950 Deep Sea Demersal Fishery	GHS50D		2.8	1.0	0.5	2.0	1.3	1.5	1.0	2.0	1.0	1.5		3.5	0.0	2.0	2.0	3.0	2.0	1.3	1.0	2.0
1950 Lugger Herring Fishery (salted)	GH50S		2.8	2.0	0.5	2.0	1.5	1.0	0.5	0.5	0.0	1.5		3.0	0.1	1.5	1.0	2.0	0.5	1.0	1.0	0.8
1950 Lugger Herring Fishery (fresh)	GH50F		2.8	2.0	0.5	2.0	1.5	1.0	0.5	0.5	0.0	1.5		3.0	0.0	1.5	1.0	2.0	0.0	1.0	1.0	0.0
1950 Cutter Deep Sea Fishery - industrial	CDS50I		2.8	2.0	0.5	2.0	2.0	1.0	2.0	0.5	0.5	1.5		2.0	0.0	1.5	1.0	3.0	0.5	0.8	1.0	0.8
1950 Cutter Deep Sea Fishery - flatfish	CDS50F		2.8	1.0	0.5	2.0	1.8	1.5	1.5	2.0	1.5	1.5		3.0	0.0	1.0	2.0	3.0	0.5	0.6	1.0	0.8
1950 Coastal Fishery - shrimps	CF50S		2.5	2.0	0.5	0.8	0.5	2.0	2.0	1.5	1.0	1.5		1.5	0.1	1.0	1.5	2.5	1.5	0.6	1.0	1.0
1950 Coastal Fishery - mussels	CF50M		1.0	2.0	0.5	0.5	1.0	0.0	1.0	0.5	0.5	1.5		1.3	0.0	1.0	1.5	1.0	0.0	0.7	2.0	0.0
1975 Deep Sea Pelagic Fishery	GHS75H		3.0	2.0	0.5	2.0	2.0	1.0	2.0	0.3	0.5	2.0		4.0	0.0	2.0	3.0	3.0	2.0	1.0	1.3	2.0
1975 Deep Sea Demersal Fishery	GHS75D		3.0	1.0	0.5	2.0	1.5	1.0	0.5	0.5	0.0	2.0		2.5	0.0	2.0	2.5	3.0	2.0	1.0	1.5	2.0
1975 Lugger Herring Fishery (fresh)	GH75F		3.0	1.0	0.5	2.0	2.0	1.0	0.5	0.5	0.0	2.0		4.0	0.0	1.5	2.5	3.0	0.0	1.0	1.3	1.0
1975 Cutter Deep Sea Fishery - industrial	CDS75I		3.0	2.0	0.5	2.0	2.0	0.0	1.5	0.5	1.0	2.5		2.5	0.0	2.0	0.5	3.0	0.0	0.8	1.3	1.0
1975 Cutter Deep Sea Fishery - flatfish	CDS75F		3.0	1.5	0.5	2.0	2.0	1.0	1.0	2.0	1.5	2.5		2.5	0.0	1.5	2.5	3.0	0.0	0.8	1.5	1.0
1975 Coastal Fishery - shrimps	CF75S		2.0	2.0	0.5	0.8	1.5	1.0	2.0	2.0	1.0	2.5		1.5	0.0	2.0	1.5	2.5	2.0	0.8	1.8	1.0
1975 Coastal Fishery - mussels	CF75M		1.5	2.0	0.5	0.5	0.5	0.0	0.0	0.0	0.0	2.5		1.8	0.0	3.0	1.5	2.5	0.8	0.5	1.8	1.5
1997 Deep Sea Pelagic Fishery	GHS97H		2.5	2.0	0.0	2.0	1.0	0.0	1.0	0.3	0.5	2.5		1.5	0.0	2.0	2.5	3.0	2.0	1.0	1.3	1.5
1997 Deep Sea Demersal Fishery	GHS97D		3.0	1.0	0.0	2.0	2.0	1.0	0.6	0.6	0.5	2.5		4.0	0.0	2.0	2.5	3.0	1.5	1.0	1.5	1.5
1997 Cutter Deep Sea Fishery - flatfish	CDS97F		3.0	1.0	0.0	2.0	2.0	0.0	1.0	2.0	1.0	3.0		3.0	0.0	2.0	2.5	3.0	0.5	0.5	1.5	1.8
1997 Coastal Fishery - shrimps	CF97S		2.0	2.0	0.0	0.8	0.5	0.5	0.5	1.5	1.0	3.0		1.8	0.0	2.0	2.0	3.0	2.0	0.5	1.8	1.8
1997Coastal Fishery - mussels	CF97M		3.0	2.0	0.0	0.5	1.0	2.0	0.0	0.0	0.0	3.0		1.0	0.0	3.0	2.0	3.0	1.5	1.0	1.8	1.8
Good	G		0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.0		5.0	2.0	4.0	2.0	2.0	0.0	0.0	0.0	0.0
Bad	B		3.0	2.0	2.0	2.0	2.0	2.0	2.0	2.0	2.0	0.0		0.0	0.0	0.0	0.0	0.0	3.0	2.0	2.0	2.0

APPENDIX 7 (PART 2): ORIGINAL DATA USED FOR THE GERMAN NORTH SEA FISHERIES IN THE FIVE 'RAPFISH' FIELDS.

Fishery	Abbreviation	Social	Socialization of fishing	Fishing community growth	Fishing sector	Environmental knowledge	Education level	Conflict status	Fisher influence	Fishing income	Kin participation	Technological	Trip length	Landing sites	Pre-sale processing	Onboard Harvest Handling	Gear	Selective gear	FADS	Vessel size	Catching power	Gear side effects	Ethical	Adjacency and reliance	Alternatives	Equity in entry to Fishery	Just management	Influences – ethical formation	Mitigation – habitat destruction	Mitigation – ecosystem depletion	Illegal fishing	Discards & wastes
1880 Deep Sea Demersal Fishery	GHS1880D		0.5	0.0	0.0	0.3	0.0	1.5	1.0	2.0	0.5		12.0	2.0	1.0	2.0	1.0	0.0	0.3	4.0	0.0	1.5		1.0	1.3	0.0	2.0	3.0	0.5	1.5	0.0	1.0
1880 Lugger Herring Fishery (salted)	GH1880S		0.5	2.0	0.0	0.3	0.0	1.5	1.0	1.0	0.0		30.0	2.0	1.5	1.0	0.5	1.5	0.0	4.0	4.0	0.5		1.5	1.0	1.5	2.0	3.0	2.0	2.0	0.0	0.0
1880 Cutter Deep Sea Fishery - industrial	CDS1880I		1.0	2.0	2.0	0.3	0.0	1.5	1.0	2.0	2.0		13.0	2.5	1.0	1.5	1.0	0.0	0.0	3.0	3.0	0.5		3.0	0.0	1.5	1.8	3.0	1.8	2.0	0.0	1.0
1880 Cutter Deep Sea Fishery - flatfish	CDS1880F		1.0	2.0	2.0	0.3	0.0	1.5	1.0	2.0	2.0		13.0	2.5	0.0	3.0	1.0	0.0	0.0	3.0	3.0	0.8		3.0	0.0	1.5	1.8	3.0	1.0	1.5	0.0	1.0
1880 Coastal Fishery - shrimps	CF1880S		1.0	0.0	1.0	0.3	0.0	0.0	0.5	0.0	2.5		1.0	0.0	0.0	0.0	0.3	0.5	0.0	0.0	1.0	0.0		2.0	1.0	1.5	1.5	3.0	2.0	2.0	0.0	0.0
1880 Coastal Fishery - oysters	CF1880O		1.0	0.0	1.0	0.3	0.0	1.0	0.5	2.0	2.0		1.0	2.0	0.5	0.5	1.0	1.0	0.0	3.5	1.0	2.0		2.0	0.5	2.0	1.5	3.0	0.5	0.5	1.0	0.0
1880 Coastal Fishery - hydrozoids	CF1800H		1.0	0.0	1.0	0.3	0.0	0.0	0.5	0.0	0.5		1.0	2.0	0.5	0.0	0.0	2.0	0.0	0.0	0.0	0.0		0.0	1.0	1.5	1.5	3.0	2.0	2.0	0.0	0.0
1880 Coastal Fishery - estuary	CF1880R		1.0	0.0	1.5	0.3	0.0	0.5	0.5	2.0	2.0		1.0	0.5	0.0	3.0	0.5	0.5	0.0	0.0	0.0	0.5		3.0	0.0	1.5	1.5	3.0	2.0	2.0	0.0	0.0
1880 Coastal Fishery -flatfish	CF1880F		2.0	0.0	1.3	0.3	0.0	0.5	0.5	1.5	2.0		1.0	0.0	0.0	3.0	0.5	0.5	0.5	0.0	0.0	0.8		2.5	0.3	2.5	1.5	3.0	2.0	1.0	0.0	1.0
1900 Deep Sea Demersal Fishery	GHS1900D		0.0	2.0	0.0	0.3	1.0	1.5	1.0	2.0	0.5		12.0	2.5	1.0	2.0	1.0	0.0	0.0	4.0	4.0	0.5		1.0	1.5	0.0	2.0	3.5	0.5	0.8	0.0	1.0
1900 Lugger Herring Fishery (salted)	GH1900S		0.5	2.0	0.0	0.3	0.5	1.5	1.0	1.0	0.0		30.0	2.0	1.5	1.0	0.5	1.5	0.0	4.0	4.0	0.5		1.5	1.0	1.5	2.0	3.5	2.0	2.0	0.0	0.0
1900 Cutter Deep Sea Fishery - industrial	CDS1900I		0.8	0.5	2.0	0.3	0.5	1.5	1.0	2.0	2.0		13.0	2.0	1.0	2.0	1.0	0.0	0.0	3.5	2.0	0.5		3.0	0.0	1.5	1.8	3.5	1.8	1.0	0.0	1.0
1900 Cutter Deep Sea Fishery - flatfish	CDSH1900F		1.0	0.5	2.0	0.3	0.5	1.5	1.0	2.0	2.0		13.0	2.0	0.0	3.0	1.0	0.0	0.0	3.5	2.0	1.5		3.0	0.0	1.5	1.8	3.5	0.5	0.5	0.0	1.0
1900 Coastal Fishery - shrimps	CF1900S		1.0	2.0	1.0	0.3	0.0	0.5	1.0	1.0	2.0		1.0	0.0	1.0	1.0	1.0	1.0	0.0	1.0	2.0	1.0		2.5	0.8	1.5	1.5	3.5	0.5	2.0	0.0	1.0
1900 Coastal Fishery - mussels	CF1900M		1.0	0.0	1.0	0.3	0.0	0.0	1.0	0.0	2.0		1.0	0.0	1.0	0.5	1.0	1.0	0.0	1.0	0.0	1.5		1.0	1.0	1.5	1.5	3.5	1.0	2.0	0.0	0.5
1900 Coastal Fishery - oysters	CF1900O		1.0	0.0	1.0	0.3	0.0	1.0	1.0	0.0	2.0		1.0	2.0	0.5	0.5	1.0	1.0	0.0	3.5	1.5	2.0		2.0	1.0	2.0	1.5	3.0	0.5	0.5	0.0	0.5
1900 Coastal Fishery - hydrozoids	CF1900H		1.0	2.0	1.0	0.3	0.0	1.0	1.0	1.0	0.5		1.0	2.0	0.5	0.5	1.0	2.0	0.0	1.0	4.0	1.0		1.0	1.0	1.5	1.5	3.0	1.0	2.0	0.0	1.0
1900 Coastal Fishery - estuary	CF1900R		1.0	2.0	1.5	0.3	0.0	0.8	1.0	2.0	2.0		1.0	0.5	0.5	3.0	0.8	0.5	0.0	1.0	3.0	0.5		3.0	0.0	1.5	1.5	3.0	2.0	2.0	0.0	0.0
1900 Coastal Fishery -flatfish	CF1900F		2.0	2.0	1.3	0.3	0.0	0.8	1.0	1.5	2.0		1.0	0.0	0.0	3.0	0.5	0.5	0.3	0.5	2.0	1.0		2.5	0.3	2.5	1.5	3.0	1.8	1.0	0.0	1.0
1925 Deep Sea Pelagic Fishery	GHS25H		0.0	0.0	0.0	0.5	1.0	2.0	1.0	2.0	0.5		12.0	2.3	2.0	2.0	1.0	0.5	0.0	4.0	4.0	0.5		1.0	1.3	0.0	2.0	3.8	1.3	2.0	0.0	0.5
1925 Deep Sea Demersal Fishery	GHS25D		0.0	0.0	0.0	0.5	1.0	2.0	1.0	2.0	0.5		12.0	2.3	1.0	2.0	1.0	0.0	0.0	4.0	3.5	0.5		1.0	1.3	0.0	2.0	3.8	0.8	0.5	0.0	1.5
1925 Lugger Herring Fishery (salted)	GH25S		0.5	0.0	0.0	0.5	1.0	1.5	1.0	1.0	0.0		30.0	2.0	1.5	1.0	0.5	1.5	0.0	4.0	2.0	0.5		1.5	0.8	1.5	2.0	3.5	2.0	2.0	0.0	0.0
1925 Cutter Deep Sea Fishery - industrial	CDS25I		0.8	0.5	1.0	0.5	0.5	2.0	1.0	2.0	2.0		6.0	2.0	1.0	2.0	1.0	0.0	0.0	3.5	2.5	0.0		2.5	0.8	1.5	1.8	3.5	2.0	1.0	0.0	1.0
1925 Cutter Deep Sea Fishery - flatfish	CDSH25F		1.0	0.5	1.0	0.5	0.5	2.0	1.0	2.0	2.0		6.0	2.0	0.0	2.5	1.0	0.0	0.0	3.5	2.5	1.3		2.5	0.8	1.5	1.8	3.5	0.5	0.5	0.0	1.0
1925 Coastal Fishery - shrimps	CF25S		2.0	1.0	1.0	0.5	0.5	0.5	1.0	1.5	2.0		1.0	0.3	1.0	1.0	1.0	0.0	0.0	1.0	1.0	1.5		2.8	0.5	1.5	1.5	3.5	0.5	1.5	0.0	1.0
1925 Coastal Fishery - mussels	CF25M		1.0	0.0	1.0	0.5	0.0	0.0	1.0	0.0	2.0		1.0	1.0	1.0	0.5	1.0	1.0	0.0	3.0	2.0	1.5		1.5	0.8	1.5	1.5	3.5	0.5	2.0	0.0	1.0
1925 Coastal Fishery - oysters	CF25O		1.0	0.0	1.0	0.5	0.0	1.0	1.0	0.0	2.0		1.0	2.0	0.5	0.5	1.0	1.0	0.0	3.5	0.0	2.0		2.0	0.8	2.0	1.5	3.5	0.5	0.5	0.0	0.5
1925 Coastal Fishery - hydrozoids	CF25H		1.0	0.0	1.0	0.5	0.0	1.0	1.0	1.5	0.5		1.0	2.0	0.5	0.5	1.0	2.0	0.0	3.5	4.0	1.0		1.0	0.8	1.5	1.5	3.5	1.0	2.0	0.0	1.0
1925 Coastal Fishery - estuary	CF25R		2.0	1.0	1.5	0.5	0.5	1.5	1.0	1.8	2.0		4.0	0.5	0.5	2.5	0.8	0.3	0.0	1.5	3.0	0.0		3.0	0.0	1.5	1.5	3.5	2.0	1.0	0.0	0.5
1925 Coastal Fishery -flatfish	CF25F		2.0	1.0	1.3	0.5	0.5	1.5	1.0	1.8	2.0		6.0	0.3	0.0	2.5	0.8	0.5	0.0	0.5	3.0	1.0		2.5	0.0	1.5	1.5	3.5	1.5	1.0	0.0	1.0
1950 Deep Sea Pelagic Fishery	GHS50H		0.0	0.0	0.0	0.5	1.0	2.0	1.0	2.0	0.0		13.1	2.3	2.0	2.0	1.0	1.0	0.0	4.0	4.0	0.0		1.0	1.5	0.0	2.0	3.5	1.5	2.0	0.0	0.5
1950 Deep Sea Demersal Fishery	GHS50D		0.0	0.0	0.0	0.5	1.0	2.0	1.0	2.0	0.0		13.2	2.3	1.5	2.0	1.0	1.0	0.0	4.0	4.0	0.5		1.0	1.5	0.0	2.0	3.5	0.8	0.5	0.0	2.0
1950 Lugger Herring Fishery (salted)	GH50S		0.5	0.0	0.0	0.5	1.0	2.0	1.0	1.0	0.0		26.5	2.0	1.5	1.0	0.8	1.5	0.0	4.0	0.5	1.0		1.5	0.9	0.5	2.0	3.5	1.5	2.0	0.0	0.3
1950 Lugger Herring Fishery (fresh)	GH50F		0.5	0.0	0.0	0.5	1.0	2.0	1.0	0.0	0.0		26.5	2.0	0.0	2.0	0.8	1.5	0.0	4.0	0.5	1.0		1.5	0.9	0.5	2.0	3.5	1.5	2.0	0.0	0.3
1950 Cutter Deep Sea Fishery - industrial	CDS50I		0.8	0.5	0.1	0.5	1.0	2.0	1.0	2.0	0.0		7.0	2.0	0.5	0.5	1.0	0.0	0.0	3.0	2.0	0.0		2.3	1.0	0.5	1.8	3.5	2.0	0.5	0.0	0.5
1950 Cutter Deep Sea Fishery - flatfish	CDS50F		1.0	0.5	0.1	0.5	1.0	2.0	1.0	2.0	1.0		11.0	1.5	0.5	2.5	0.9	0.0	0.2	3.0	0.5	1.0		2.3	1.0	0.5	1.8	3.5	0.5	0.5	0.0	2.0
1950 Coastal Fishery - shrimps	CF50S		1.5	0.1	1.0	0.5	1.0	1.0	1.0	1.8	1.5		1.0	0.5	1.5	1.0	1.0	0.0	0.0	2.0	2.0	1.5		2.8	0.5	0.5	1.5	3.5	0.5	1.0	0.0	2.0
1950 Coastal Fishery - mussels	CF50M		1.0	0.5	0.5	0.5	1.0	0.5	1.0	1.0	1.0		1.0	1.0	1.0	0.5	1.0	1.0	0.0	3.0	1.0	1.3		1.5	1.0	0.5	1.5	3.5	1.5	1.8	0.0	0.8
1975 Deep Sea Pelagic Fishery	GHS75H		0.0	0.0	0.0	1.0	1.0	2.0	1.0	2.0	0.0		110.0	3.0	2.0	2.0	1.0	1.5	0.0	4.0	0.5	0.0		1.0	2.0	0.0	2.0	3.5	1.5	1.5	1.5	0.5
1975 Deep Sea Demersal Fishery	GHS75D		0.0	0.0	0.0	1.0	1.0	2.0	1.0	2.0	0.0		110.0	3.0	2.0	2.5	1.0	1.5	0.0	4.0	0.5	0.5		1.0	2.0	0.0	2.0	3.5	1.0	1.3	1.5	2.0
1975 Lugger Herring Fishery (fresh)	GH75F		0.0	0.0	0.0	1.0	1.0	2.0	1.0	2.0	0.0		15.0	3.0	0.0	2.0	0.8	1.5	0.0	4.0	0.0	1.5		1.5	1.5	0.5	2.0	3.5	2.0	1.5	1.5	0.3
1975 Cutter Deep Sea Fishery - industrial	CDS75I		0.8	0.0	0.0	1.0	1.0	2.0	1.0	2.0	0.0		15.0	2.0	0.5	0.5	1.0	0.0	0.0	4.0	0.3	0.0		1.0	1.0	0.5	2.0	3.5	2.0	0.5	1.5	0.5
1975 Cutter Deep Sea Fishery - flatfish	CDS75F		1.0	0.0	0.0	1.0	1.0	2.0	1.0	2.0	0.3		15.0	2.5	0.0	2.0	0.8	0.0	0.2	4.0	0.3	1.0		1.0	1.0	0.5	2.0	3.5	1.5	1.3	1.5	2.0
1975 Coastal Fishery - shrimps	CF75S		1.5	0.1	0.2	1.0	1.0	1.0	1.0	2.0	1.0		1.0	0.8	1.0	1.0	1.0	1.0	0.0	2.5	0.5	1.3		2.0	0.8	0.5	1.8	3.5	1.5	1.5	1.5	2.0
1975 Coastal Fishery - mussels	CF75M		1.0	0.5	0.2	1.0	1.0	1.0	1.0	2.0	0.5		1.0	1.0	0.5	0.5	1.0	1.5	0.0	4.0	2.0	0.8		2.0	0.8	0.5	1.8	3.5	4.0	1.8	1.5	0.0
1997 Deep Sea Pelagic Fishery	GHS97H		0.0	0.0	0.0	1.0	1.0	2.0	1.5	2.0	0.0		110.0	3.0	2.0	2.0	1.0	1.5	0.0	4.0	0.0	0.0		1.0	2.0	0.0	2.0	4.0	2.0	1.8	1.5	0.5
1997 Deep Sea Demersal Fishery	GHS97D		0.0	0.0	0.0	1.0	1.0	2.0	1.5	2.0	0.0		110.0	3.0	1.5	2.0	1.0	1.5	0.0	4.0	0.0	0.5		1.0	2.0	0.0	2.0	4.0	1.5	1.5	1.5	2.0
1997 Cutter Deep Sea Fishery - flatfish	CDS97F		1.0	0.0	0.0	1.0	1.0	2.0	1.0	2.0	0.3		15.0	2.5	1.0	2.0	0.8	0.0	0.0	4.0	0.1	1.0		0.5	1.0	0.5	2.0	4.0	1.8	1.5	1.5	2.0
1997 Coastal Fishery - shrimps	CF97S		1.0	0.0	0.2	1.0	1.0	1.0	1.0	2.0	0.5		1.0	0.8	1.5	1.0	1.0	1.0	0.0	2.5	0.3	1.3		1.5	1.0	0.5	2.0	4.0	1.8	1.8	1.5	2.0
1997Coastal Fishery - mussels	CF97M		0.5	0.0	0.2	1.0	1.0	1.0	1.0	2.0	0.5		1.0	1.0	1.0	0.5	1.0	1.5	0.0	4.0	0.3	0.8		1.5	1.0	0.5	2.0	4.0	4.0	1.9	1.5	0.0
Good	G		2.0	2.0	2.0	2.0	2.0	0.0	2.0	2.0	4.0		0.5	0.0	2.0	3.0	0.0	0.0	0.0	0.0	0.0	0.0		3.0	2.0	2.0	4.0	4.0	4.0	4.0	0.0	0.0
Bad	B		0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0		30.0	3.0	0.0	0.0	1.0	1.0	1.0	2.0	2.0	2.0		0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.0	2.0

APPENDIX 7 (PART 3): ORIGINAL DATA USED FOR THE UK NORTH SEA FISHERIES IN THE FIVE 'RAPFISH' FIELDS. SOURCES OF SCORES ARE DOCUMENTED IN ANNEX TABLE X AND DISCUSSED IN THE TEXT.

Fishery	Abbreviation	Ecological	Exploitation status	Recruitment variability	Change in trophic level	Migratory range	Range collapse	Size of fish caught	Catch before maturity	Discarded by-catch	Species caught	Primary production	Economic	Profitability	Fisheries in GDP	Average Income	Management Regime	Other income	Sector employment	Ownership	Market	Subsidy
1910 Herring English-drift net	EN10HD		0.0	1.0	0.0	2.0	0.0	0.0	0.0	0.0	0.0	1.0		0.0	0.0	3.0	1.0	2.0	1.0	1.0	2.0	1.0
1910 Herring-Scottish drift net	SC10HD		0.0	1.0	0.0	2.0	0.0	0.0	0.0	0.0	0.0	1.0		0.0	1.0	3.0	0.0	2.0	2.0	0.0	2.0	1.0
1910 Haddock-English trawl	EN10HAT		1.0	2.0	0.0	2.0	2.0	0.0	1.0	1.0	1.0	1.0		1.0	0.0	3.0	1.0	2.0	2.0	1.0	1.0	1.0
1910 Haddock-Scottish line	SC10HAL		1.0	2.0	0.0	2.0	2.0	0.0	0.0	0.0	0.0	1.0		1.5	0.0	2.0	0.0	2.0	0.0	0.0	0.0	1.0
1910 Cod-English trawl	EN10CT		1.0	1.0	0.0	2.0	2.0	0.0	1.0	1.0	1.0	1.0		1.0	0.0	3.0	1.0	2.0	2.0	1.0	1.0	1.0
1910 Cod Scottish line	SC10CL		1.0	1.0	0.0	2.0	2.0	0.0	0.0	0.0	0.0	1.0		1.5	0.0	2.0	0.0	2.0	0.0	0.0	0.0	1.0
1910 Plaice English trawl	EN10PT		2.0	1.0	2.0	2.0	2.0	2.0	1.0	1.0	1.0	1.0		1.0	0.0	3.0	1.0	2.0	2.0	1.0	1.0	1.0
1910 Plaice Scottish line	SC10PL		2.0	1.0	2.0	2.0	2.0	2.0	0.0	0.0	0.0	1.0		1.5	0.0	2.0	0.0	2.0	0.0	0.0	0.0	1.0
1910 Haddock Scottish trawl	SC10HAT		1.0	2.0	0.0	2.0	2.0	0.0	1.0	1.0	1.0	1.0		1.0	0.5	3.0	0.0	2.0	1.0	0.0	1.0	1.0
1910 Cod Scottish trawl	SC10CT		1.0	1.0	0.0	2.0	2.0	0.0	1.0	1.0	1.0	1.0		1.0	0.5	3.0	0.0	2.0	1.0	0.0	1.0	1.0
1910 Plaice Scottish trawl	SC10PT		2.0	1.0	2.0	2.0	2.0	2.0	1.0	1.0	1.0	1.0		1.0	0.5	3.0	0.0	2.0	1.0	0.0	1.0	1.0
1990 Herring-English pel.trawl+purse seine	EN90HTS		2.0	1.0	0.0	2.0	0.0	0.0	0.0	1.0	1.0	3.0		1.0	0.0	3.0	2.0	2.0	1.0	0.0	1.5	2.0
1990 Herring-Scottish pel.trawl+purse seine	SC90HTS		2.0	1.0	0.0	2.0	0.0	0.0	0.0	1.0	1.0	3.0		1.0	0.0	3.0	2.0	2.0	1.0	0.0	1.5	2.0
1990 Haddock-English trawl+seine	EN90HATS		3.0	2.0	0.0	2.0	0.0	0.0	1.0	2.0	1.0	3.0		1.0	0.0	3.0	2.0	2.0	0.0	0.0	1.0	2.0
1990 Haddock-Scottish trawl+seine	SC90HATS		3.0	2.0	0.0	2.0	0.0	0.0	1.0	2.0	1.0	3.0		1.0	0.0	3.0	2.0	2.0	0.0	0.0	1.0	2.0
1990 Cod-English trawl+seine	EN90CTS		3.0	1.0	0.0	2.0	0.0	0.0	1.0	2.0	1.0	3.0		1.0	0.0	3.0	2.0	2.0	1.0	0.0	1.0	2.0
1990 Cod--Scottish trawl+seine	SC90CTS		3.0	1.0	0.0	2.0	0.0	0.0	1.0	2.0	1.0	3.0		1.0	0.0	3.0	2.0	2.0	1.0	0.0	1.0	2.0
1990 Plaice-English beam trawl	EN90PBT		2.0	1.0	1.0	2.0	0.0	1.0	2.0	2.0	2.0	3.0		1.0	0.0	3.0	2.0	2.0	1.0	0.0	1.0	2.0
1990 Plaice-Scottish beam trawl	SC90PBT		2.0	1.0	1.0	2.0	0.0	1.0	2.0	2.0	2.0	3.0		1.0	0.0	3.0	2.0	2.0	1.0	0.0	1.0	2.0
1955 Herring-English -drift	EN55HD		3.0	1.0	1.0	2.0	0.0	1.0	0.0	0.0	0.0	1.0		3.0	0.0	3.0	0.0	2.5	1.0	1.0	1.5	2.0
1955 Herring-Scottish-drift	SC55HD		3.0	1.0	1.0	2.0	0.0	1.0	0.0	0.0	0.0	1.0		3.0	0.0	3.0	0.0	2.5	1.0	0.0	1.5	2.0
1955 Haddock-English trawl+seine	EN55HATS		1.0	2.0	0.0	2.0	0.0	0.0	2.0	1.0	1.0	1.0		1.5	0.0	3.0	0.0	2.5	1.0	1.0	1.0	2.0
1955 Haddock-Scottish trawl+seine	SC55HATS		1.0	2.0	0.0	2.0	0.0	0.0	1.0	1.0	1.0	1.0		1.5	0.0	3.0	0.0	2.5	1.0	0.0	1.0	2.0
1955 Cod-English trawl+seine	EN55CTS		1.0	1.0	0.0	2.0	0.0	0.0	0.0	1.0	1.0	1.0		1.5	0.0	3.0	0.0	2.5	2.0	1.0	1.0	2.0
1955 Cod--Scottish trawl+seine	SC55CTS		1.0	1.0	0.0	2.0	0.0	0.0	0.0	1.0	1.0	1.0		1.5	0.0	3.0	0.0	2.5	2.0	0.0	1.0	2.0
1955 Plaice-English trawl+seine	EN55PTS		1.0	1.0	1.0	2.0	0.0	1.0	0.0	1.0	1.0	1.0		1.5	0.0	3.0	0.0	2.5	0.0	1.0	1.0	2.0
1955 Plaice-Scottish trawl+seine	SC55PTS		1.0	1.0	1.0	2.0	0.0	1.0	0.0	1.0	1.0	1.0		1.5	0.0	3.0	0.0	2.5	0.0	0.0	1.0	2.0
Good	G		0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.0		5.0	2.0	4.0	2.0	2.0	0.0	0.0	0.0	0.0
Bad	B		3.0	2.0	2.0	2.0	2.0	2.0	2.0	2.0	2.0	0.0		0.0	0.0	0.0	0.0	0.0	3.0	2.0	2.0	2.0

APPENDIX 7 (PART 4): ORIGINAL DATA USED FOR THE UK NORTH SEA FISHERIES IN THE FIVE 'RAPFISH' FIELDS.

Fishery	Abbreviation	**Social**	Socialization of fishing	Fishing community growth	Fishing sector	Environmental knowledge	Education level	Conflict status	Fisher influence	Fishing income	Kin participation	**Technological**	Trip length	Landing sites	Pre-sale processing	Onboard Harvest Handling	Gear	Selective gear	FADS	Vessel size	Catching power	Gear side effects	**Ethical**	Adjacency and reliance	Alternatives	Equity in entry to Fishery	Just management	Influences – ethical formation	Mitigation – habitat destruction	Mitigation – ecosystem depletion	Illegal fishing	Discards & wastes
1910 Herring English-drift net	EN10HD		0.0	2.0	1.0	0.5	0.0	0.0	0.0	1.0	0.0		2.0	1.0	2.0	1.0	0.0	1.0	0.0	4.0	4.0	0.0		3.0	1.0	1.0	0.0	2.0	2.0	1.0	0.0	0.0
1910 Herring-Scottish drift net	SC10HD		2.0	2.0	2.0	0.5	0.0	0.0	0.0	1.0	0.0		2.0	1.0	2.0	1.0	0.0	1.0	0.0	4.0	4.0	0.0		3.0	0.0	1.0	0.5	2.0	2.0	1.0	0.0	0.0
1910 Haddock-English trawl	EN10HAT		0.0	2.0	2.0	0.5	0.0	1.0	0.0	1.0	0.0		8.0	1.0	1.0	1.0	1.0	0.0	0.0	4.0	4.0	1.0		3.0	1.0	1.0	0.0	2.0	1.0	1.0	0.0	1.0
1910 Haddock-Scottish line	SC10HAL		1.5	0.0	0.0	0.5	0.0	0.0	0.5	1.0	0.5		5.0	1.0	1.0	1.0	0.0	1.0	0.5	4.0	3.5	0.0		3.0	0.0	1.0	0.5	2.0	2.0	2.0	0.0	0.0
1910 Cod-English trawl	EN10CT		0.0	2.0	2.0	0.5	0.0	1.0	0.0	1.0	0.0		8.0	1.0	1.0	1.0	1.0	0.0	0.0	4.0	4.0	1.0		3.0	1.0	1.0	0.0	2.0	1.0	1.0	0.0	1.0
1910 Cod Scottish line	SC10CL		1.5	0.0	0.0	0.5	0.0	0.0	0.5	1.0	0.5		5.0	1.0	1.0	1.0	0.0	1.0	0.5	4.0	3.5	0.0		3.0	0.0	1.0	0.5	2.0	2.0	2.0	0.0	0.0
1910 Plaice English trawl	EN10PT		0.0	2.0	2.0	0.5	0.0	1.0	0.0	1.0	0.0		8.0	1.0	1.0	1.0	1.0	0.0	0.0	4.0	4.0	1.0		3.0	1.0	1.0	0.0	2.0	1.0	1.0	0.0	1.0
1910 Plaice Scottish line	SC10PL		1.5	0.0	0.0	0.5	0.0	0.0	0.5	1.0	0.5		5.0	1.0	1.0	1.0	0.0	1.0	0.5	4.0	3.5	0.0		3.0	0.0	1.0	0.5	2.0	2.0	2.0	0.0	0.0
1910 Haddock Scottish trawl	SC10HAT		0.0	2.0	1.0	0.5	0.0	1.0	0.0	1.0	0.0		5.0	2.0	1.0	1.0	1.0	0.0	0.0	4.0	4.0	1.0		3.0	0.0	1.0	0.0	2.0	1.0	1.0	0.0	1.0
1910 Cod Scottish trawl	SC10CT		0.0	2.0	1.0	0.5	0.0	1.0	0.0	1.0	0.0		5.0	2.0	1.0	1.0	1.0	0.0	0.0	4.0	4.0	1.0		3.0	0.0	1.0	0.0	2.0	1.0	1.0	0.0	1.0
1910 Plaice Scottish trawl	SC10PT		0.0	2.0	1.0	0.5	0.0	1.0	0.0	1.0	0.0		5.0	2.0	1.0	1.0	1.0	0.0	0.0	4.0	4.0	1.0		3.0	0.0	1.0	0.0	2.0	1.0	1.0	0.0	1.0
1990 Herring-English pel.trawl+purse seine	EN90HTS		0.0	0.0	0.0	2.0	1.0	2.0	1.0	1.0	0.0		7.0	1.0	1.0	1.0	1.0	1.0	0.0	4.0	2.0	0.0		3.0	2.0	1.0	2.0	2.0	2.0	3.0	2.0	2.0
1990 Herring-Scottish pel.trawl+purse seine	SC90HTS		0.5	0.0	0.0	2.0	1.0	2.0	1.0	1.0	1.0		5.0	1.0	1.0	1.0	1.0	1.0	0.0	4.0	2.0	0.0		3.0	1.0	1.0	2.0	2.0	2.0	3.0	2.0	2.0
1990 Haddock-English trawl+seine	EN90HATS		0.0	0.0	0.0	2.0	1.0	2.0	1.0	1.0	0.0		7.0	1.0	1.0	1.0	1.0	0.5	0.0	4.0	2.0	1.0		3.0	2.0	1.0	2.0	2.0	1.5	3.0	2.0	2.0
1990 Haddock-Scottish trawl+seine	SC90HATS		0.5	0.0	0.0	2.0	1.0	2.0	1.0	1.0	1.0		5.0	1.0	1.0	1.0	1.0	0.5	0.0	4.0	2.0	1.0		3.0	1.0	1.0	2.0	2.0	1.5	3.0	2.0	2.0
1990 Cod-English trawl+seine	EN90CTS		0.0	0.0	0.0	2.0	1.0	2.0	1.0	1.0	0.0		7.0	1.0	1.0	1.0	1.0	0.5	0.0	4.0	2.0	1.0		3.0	2.0	1.0	2.0	2.0	1.5	3.0	2.0	2.0
1990 Cod--Scottish trawl+seine	SC90CTS		0.5	0.0	0.0	2.0	1.0	2.0	1.0	1.0	1.0		5.0	1.0	1.0	1.0	1.0	0.5	0.0	4.0	2.0	1.0		3.0	1.0	1.0	2.0	2.0	1.5	3.0	2.0	2.0
1990 Plaice-English beam trawl	EN90PBT		0.0	0.0	0.0	2.0	1.0	2.0	1.0	1.0	0.0		7.0	1.0	1.0	1.0	1.0	0.5	0.0	4.0	2.0	1.0		3.0	2.0	1.0	2.0	2.0	1.0	3.0	2.0	2.0
1990 Plaice-Scottish beam trawl	SC90PBT		0.5	0.0	0.0	2.0	1.0	2.0	1.0	1.0	1.0		5.0	1.0	1.0	1.0	1.0	0.5	0.0	4.0	2.0	1.0		3.0	1.0	1.0	2.0	2.0	1.0	3.0	2.0	2.0
1955 Herring-English -drift	EN55HD		0.0	0.0	0.0	1.0	1.0	1.0	0.5	1.0	0.0		3.0	1.0	1.5	1.0	0.0	1.0	0.0	3.5	3.0	0.0		3.0	0.5	0.0	0.0	1.0	2.0	0.0	0.0	0.0
1955 Herring-Scottish-drift	SC55HD		0.5	0.0	0.0	1.0	1.0	1.0	0.5	1.0	0.5		3.0	1.0	1.5	1.0	0.0	1.0	0.0	3.5	3.0	0.0		3.0	0.5	0.0	0.0	1.0	2.0	0.0	0.0	0.0
1955 Haddock-English trawl+seine	EN55HATS		0.0	1.0	1.0	1.0	1.0	1.0	0.5	1.0	0.0		8.0	1.0	1.0	1.0	1.0	0.5	0.0	3.5	4.0	1.0		3.0	0.5	0.5	0.0	1.0	1.0	1.0	0.0	1.0
1955 Haddock-Scottish trawl+seine	SC55HATS		0.5	1.0	1.0	1.0	1.0	1.0	0.5	1.0	0.5		5.0	1.0	1.0	1.0	1.0	0.5	0.0	3.5	4.0	1.0		3.0	0.5	0.5	0.0	1.0	1.0	1.0	0.0	1.0
1955 Cod-English trawl+seine	EN55CTS		0.0	1.0	1.0	1.0	1.0	1.0	0.5	1.0	0.0		8.0	1.0	1.0	1.0	1.0	0.5	0.0	3.5	4.0	1.0		3.0	0.5	0.5	0.0	1.0	1.0	1.0	0.0	1.0
1955 Cod--Scottish trawl+seine	SC55CTS		0.5	1.0	1.0	1.0	1.0	1.0	0.5	1.0	0.5		5.0	1.0	1.0	1.0	1.0	0.5	0.0	3.5	4.0	1.0		3.0	0.5	0.5	0.0	1.0	1.0	1.0	0.0	1.0
1955 Plaice-English trawl+seine	EN55PTS		0.0	0.0	0.0	1.0	1.0	1.0	0.5	1.0	0.0		8.0	1.0	1.0	1.0	1.0	0.5	0.0	3.5	4.0	1.0		3.0	0.5	0.5	0.0	1.0	1.0	1.0	0.0	1.0
1955 Plaice-Scottish trawl+seine	SC55PTS		0.5	0.0	0.0	1.0	1.0	1.0	0.5	1.0	0.5		5.0	1.0	1.0	1.0	1.0	0.5	0.0	3.5	4.0	1.0		3.0	0.5	0.5	0.0	1.0	1.0	1.0	0.0	1.0
Good	G		2.0	2.0	2.0	2.0	2.0	0.0	2.0	2.0	4.0		0.5	0.0	2.0	3.0	0.0	0.0	0.0	0.0	0.0	0.0		3.0	2.0	2.0	4.0	4.0	4.0	4.0	0.0	0.0
Bad	B		0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0		30.0	3.0	0.0	0.0	1.0	1.0	1.0	2.0	2.0	2.0		0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.0	2.0

APPENDIX 8: RESULTS OF THE MDS RAPFISH ORDINATION ON THE FIVE EVALUATION FIELDS FOR THE GERMAN AND UNITED KINGDOM FISHERIES. VALUES FOR SUSTAINABILITY AXES ONLY, AS PERCENTAGE OF THE BEST POSSIBLE.

Fishery	Ecological	Economic	Social	Technological	Ethical
1880 Deep Sea Demersal Fishery	45.9	17.2	39.0	40.7	39.9
1880 Deep Sea Demersal Fishery	63.2	33.3	24.8	57.6	54.0
1880Cutter Deep Sea fishery - industrial	58.9	36.6	34.4	48.8	50.3
1880 Cutter Deep Sea fishery - flatfish	38.0	33.2	34.4	38.3	43.3
1880 Coastal fishery - shrimps	69.9	44.0	24.8	51.4	52.4
1880 Coastal fishery - oysters	63.5	42.8	36.6	41.4	35.1
1880 Coastal fishery - hydrozoids	79.5	55.1	18.0	70.9	55.1
1880 Coastal fishery - estuary	47.0	39.6	35.1	54.8	51.9
1880 Coastal fishery -flatfish	38.2	49.1	36.6	23.2	50.3
1880 Deep Sea Demersal Fishery	41.1	25.0	39.7	45.6	37.0
1900 Lugger Herring Fishery (salted)	52.2	34.1	29.0	57.6	55.3
1900 Cutter Deep Sea fishery - industrial	46.9	29.3	41.4	49.9	46.3
1900 Cutter Deep Sea fishery - flatfish	37.1	26.1	42.1	35.3	33.3
1900 Coastal fishery - shrimps	62.6	36.9	32.7	52.0	46.0
1900 Coastal fishery - mussels	76.3	39.4	32.6	51.5	49.6
1900 Coastal fishery - oysters	62.1	37.6	33.5	41.2	37.4
1900 Coastal fishery - hydrozoids	56.1	35.7	29.1	53.5	47.4
1900 Coastal fishery - estuary	42.6	31.5	35.3	50.3	51.8
1900 Coastal fishery -flatfish	33.6	39.8	36.2	31.6	48.2
1925 Deep Sea Pelagic Fishery	42.7	33.1	44.6	53.6	52.3
1925 Deep Sea Demersal Fishery	40.0	29.0	44.6	47.0	36.0
1925 Lugger Herring Fishery (salted)	47.9	31.5	38.7	58.1	54.7
1925 Cutter Deep Sea fishery - industrial	40.3	30.3	43.5	51.2	48.8
1925 Cutter Deep Sea fishery - flatfish	35.7	32.7	44.1	38.6	39.1
1925 Coastal fishery - shrimps	49.5	31.8	43.4	42.6	42.6
1925 Coastal fishery - mussels	57.0	42.5	34.3	47.8	45.5
1925 Coastal fishery - oysters	37.0	31.5	35.1	42.4	36.7
1925 Coastal fishery - hydrozoids	45.5	22.8	36.5	51.4	47.8
1925 Coastal fishery - estuary	35.4	31.0	44.9	51.8	47.5
1925 Coastal fishery -flatfish	32.5	33.9	45.1	42.0	43.3
1950 Deep Sea Pelagic Fishery	44.0	32.2	42.3	60.2	53.7
1950 Deep Sea Demersal Fishery	41.1	19.6	42.3	53.7	35.3
1950 Lugger Herring Fishery (salted)	48.8	33.4	38.6	57.9	50.5
1950 Lugger Herring Fishery (fresh)	48.8	31.9	33.7	55.2	50.5
1950 Cutter Deep Sea fishery - industrial	39.3	30.8	42.6	49.1	46.0
1950 Cutter Deep Sea fishery - flatfish	35.3	27.9	45.8	43.3	36.3
1950 Coastal fishery - shrimps	39.9	33.6	45.1	43.5	36.0
1950 Coastal fishery - mussels	61.3	43.0	40.4	50.5	47.4
1975 Deep Sea Pelagic Fishery	38.4	16.4	43.9	78.3	52.9
1975 Deep Sea Demersal Fishery	49.9	24.0	43.9	75.5	44.0
1975 Lugger Herring Fishery (fresh)	47.6	20.7	43.9	52.3	53.6
1975 Cutter Deep Sea fishery - industrial	41.7	28.7	45.5	51.7	46.8
1975 Cutter Deep Sea fishery - flatfish	37.3	28.6	46.8	38.6	45.4
1975 Coastal fishery - shrimps	43.8	31.5	48.1	51.8	45.2
1975 Coastal fishery - mussels	72.6	34.5	44.2	51.4	66.5
1997 Deep Sea Pelagic Fishery	58.6	28.6	53.1	79.8	59.4
1997 Deep Sea Demersal Fishery	50.3	19.9	53.1	74.5	50.0
1997 Cutter Deep Sea fishery - flatfish	47.3	26.8	46.8	50.2	50.1
1997 Coastal fishery - shrimps	63.0	27.2	45.5	54.7	50.5
1997Coastal fishery - mussels	73.1	30.5	44.6	56.3	71.0
1910 Herring English-drift net	75.6	38.0	11.3	63.8	42.7
1910 Herring-Scottish drift net	75.6	50.7	7.5	63.8	41.0
1910 Haddock-English trawl	49.6	36.6	10.0	43.3	36.6
1910 Haddock-Scottish line	59.7	38.7	27.2	42.7	45.7
1910 Cod-English trawl	51.8	36.6	10.0	43.3	36.6
1910 Cod Scottish line	61.7	38.7	27.2	42.7	45.7
1910 Plaice English trawl	24.9	36.6	10.0	43.3	36.6
1910 Plaice Scottish line	34.7	38.7	27.2	42.7	45.7
1910 Haddock Scottish trawl	49.6	43.6	14.6	43.3	33.1
1910 Cod Scottish trawl	51.6	43.6	14.6	43.3	33.1
1910 Plaice Scottish trawl	24.9	43.6	14.6	43.3	33.1
1990 Herr-English pel.trawl+purse seine	65.7	39.0	42.2	55.0	63.2
1990 Herr-Scottish pel.trawl+purse seine	65.7	39.0	47.6	54.9	60.5
1990 Haddock-English trawl+seine	57.6	42.1	42.2	48.8	59.2
1990 Haddock-Scottish trawl+seine	57.6	42.1	47.6	48.6	56.8
1990 Cod-English trawl+seine	57.2	40.1	42.2	48.8	59.2
1990 Cod--Scottish trawl+seine	57.2	40.1	47.6	48.6	56.8
1990 Plaice-English beam trawl	39.2	40.1	42.2	48.8	55.1
1990 Plaice-Scottish beam trawl	39.2	40.1	47.6	48.6	52.4
1955 Herring-English -drift	52.9	25.0	33.2	60.8	29.0
1955 Herring-Scottish-drift	52.9	26.1	36.2	60.8	29.0
1955 Haddock-English trawl+seine	57.3	30.6	30.7	46.9	31.7
1955 Haddock-Scottish trawl+seine	59.1	32.2	33.8	46.5	31.7
1955 Cod-English trawl+seine	63.7	28.1	30.7	46.9	31.7
1955 Cod--Scottish trawl+seine	63.7	28.7	33.8	46.5	31.7
1955 Plaice-English trawl+seine	54.1	30.4	33.2	46.9	31.7
1955 Plaice-Scottish trawl+seine	54.1	33.5	36.2	46.5	31.7

APPENDIX 9: SCORES USED FOR FISHERIES IN THE CODE OF CONDUCT 'INTENTIONS' RAPFISH FIELD.

Fishery	**Management**	formal ref points	fleet capacity	small scale fisheries consider	biodiversity impacts allowed	restoring depleted stocks	human impacts on habitat	fish gear mandated to avoid impacts	explicit ecosystem linkages	explicit environmental influences	**Framework**	removals accounted for	compatible management measures	long term objectives stated	stakeholders identified & considered	open & tranparent processees	timely & complete statisitics	social, econom & instit factors	**Precaution**	PP enshrined in legislation	uncertainty quantified & used	stock-specific target reference points	stock-specific limit reference points	contingeny plans - env	contingeny plans - fishing	continuous management review	no-take areas sufficient & working	restrict fishing if spp threatened
GOM Inshore		1	0	3	1	1	3	0	1	1		2.5	2	2	4	3	4	2.3		1.5	2	2	1.5	1.5	2.5	3	2.5	2
GOM Lobster		2	3	4	3	2	3	2	0.5	2.3		3	2	2	4	4	4	3		2	3	3	3	1.3	3	3	2.5	2
GOM Offshore commercial trawl		1.8	2.5	2.5	1	2	2	0	1	1		2.5	2	2	3.5	3	4	1		2	2.5	2.5	3	1.5	3	3	5	2
N.Sea-Eng Herring(trawl/purse seine)		2	1.5	2	0	0.5	1.5	1	0	0		2	1.3	1.5	2	0	2	1		0	3	3	1	1	1	3	0.5	0.3
N.Sea-Eng Plaice(beam trawl)		2	1.5	1	0	0.5	1.5	0	0	0		1.5	1	1.5	1.5	0	2.5	1		0	3	3	1	1	0	3	2.5	0.5
N.Sea-Eng Haddock(trawl/seine)		2	2.5	1	0.5	1	1.5	0	0	0		2.5	1.3	1.5	1.5	0	2.5	2		0	3	3	1	1	0	3	0	0.3
N.Sea-Eng Cod(trawl/seine)		2	2.5	1	0.5	0.8	1.5	0	0	0		2.5	1.3	1.5	1.5	0	2.5	1		0	3	3	1	1	0	3	0	0.3
N.Sea-Ger Deep sea herring pelagic		2	3	1	2	2	3	1.5	2	2		3	3	2	4	2.5	4.5	1		2	2.5	3	3	1	3	3	0	1
N.Sea-Ger Deep sea demersal		2	3	1	2	1.5	3	2	2	2		3	3	2	4	2.5	4.5	1		2	2.5	1	3	1	0	3	0	1
N.Sea-Ger Deep Sea flatfish		2	3	1	2	1	3	1.5	2	2		3	3	2	3	2.5	4.5	2		2	2.5	3	3	1	0	3	3	1
N.Sea-Ger Coastal Shrimp		1	2	2	1.5	1.5	3	1.5	3	2		1.5	2	2	4	2.5	3	2.5		1	1	2	2	1	1.5	3	1.5	0
N.Sea-Ger Coastal Mussel		1	2	2	2	0.5	3	0	2.5	2		1.5	2	1.5	4	2.5	3	2.5		1	3	2	2	1	1	3	0	0
GOOD		2	3	4	3	2	3	2	3	3		3	2	2	4	4	5	4		2	3	3	3	3	3	3	5	3
BAD		0	0	0	0	0	0	0	0	0		0	0	0	0	0	0	0		0	0	0	0	0	0	0	0	0

	Stocks, Fleetts & Gear	reducing excess fleet cap	harmful methods	non-sp by-catch minimised	discards minimised	gear minimises ghost fishing	Safe levels of juv and spawners fishing	rebuilding depleted stocks	**Social & Economic**	conflict minimised	local needs	cost-effective evaluation of change	social impact evaluated	cost recovery for res & MCS	**MCS**	observer effectiveness	inspection effectiveness	vessel monitoring effectiveness	illegal non-flag fishing	effectiveness in stopping illegal fishing
shore		0	0	2.5	1	0	0.25	1		0	1	1	1.5	0		0	1	1	0	0.5
obster		1	1	4	2	2	2	2.5		1	2	2	1.5	0		2	4	3	1	4
ffshore commercial trawl		1.5	0	1	1	1	0.5	2		0	0	2	0	0		2.5	3.5	4	0	1
ng Herring(trawl/purse seine)		0	0	1.5	0	2	1	1.5		0	1	1	0	0		2	3	3	2	0
ng Plaice(beam trawl)		2	0	1	0	2	0	1.25		1	1	1	0	0		2	3	3	1	1
ng Haddock(trawl/seine)		1	0	1	0	2	0.5	1		1	1	1	0.5	0		2	3	3	1	1
ng Cod(trawl/seine)		1	0	1	0	2	0.5	0		1	1	1	0	0		2	3	3	1	0
er Deep sea herring pelagic		3	0	3	3	2	3	2.5		1.5	0	1.5	1	0		0	3	4	1.5	0.5
er Deep sea demersal		3	0	3	3	2	2	2		1.5	0	1.5	1	0		0	3	4	1.5	0.5
er Deep Sea flatfish		3	0	3	2	2	1.5	1		2	0	1.5	1	0		0	3	4	1.5	1
er Coastal Shrimp		2	0	2.5	1.5	2	1.5	1.5		2	1	2	2	0		0	2	2	0	4
er Coastal Mussel		2	0	1	0	2	2	2.5		2	1.5	2	2	0		0	2	2	0	4
		3	2	5	3	2	3	3		2	2	2	2	3		4	4	4	0	4
		0	0	0	0	0	0	0		0	0	0	0	0		0	0	0	4	0

Appendix 10: Results of the MDS Rapfish ordination on the six evaluation fields for the Gulf of Maine, German and United Kingdom fisheries. Values for sustainability axes only, as percentage of the best possible.

Fishery	management	framework	precaution	social & econ	stocks, fleet & gear	MCS
GOM Inshore	43.3	82.2	62.4	37.9	23.9	32.6
GOM Lobster	92.1	91.0	77.3	56.3	60.8	77.6
GOM Offshore commercial trawl	58.8	78.6	80.1	22.8	35.9	76.9
N.Sea-UK Herring(trawl/purse seine)	40.8	59.8	54.6	28.5	28.1	61.1
N.Sea-UK Plaice(beam trawl)	34.5	56.2	55.6	33.1	29.1	62.5
N.Sea-UK Haddock(trawl/seine)	43.7	60.7	51.4	36.7	28.4	62.5
N.Sea-UK Cod(trawl/seine)	41.9	59.3	51.4	33.1	21.2	62.9
N.Sea-Ger Deep sea herring pelagic	74.5	91.7	73.4	32.9	67.7	64.1
N.Sea-Ger Deep sea demersal	74.0	91.7	58.2	32.9	60.3	64.1
N.Sea-Ger Deep Sea flatfish	67.8	88.9	69.2	28.0	51.6	63.7
N.Sea-Ger Coastal Shrimp	67.3	78.3	54.9	58.0	45.2	53.1
N.Sea-Ger Coastal Mussel	54.5	73.8	58.0	60.9	41.1	53.1

APPENDIX 11: SOURCES OF INFORMATION FOR GULF OF MAINE, GERMAN AND UNITED KINGDOM FISHERIES

Gulf of Maine Fisheries

Acheson, J.M. and R.S. Steneck. 1997. Bust then boom in the Maine lobster industry: perspectives of fishers and biologists. N. Am. J. Fish. Mgt. 17: 826-847.

Alexander, A.B., H.F. Moore, and W.C. Kendall. 1914. Report on the otter trawl fishery. Report of the United States Commissioner of Fisheries for the Fiscal Year Ending June 30, 1914, Appendix VI: 97 p.

Clark, S.H. 1998. Status of the fishery resources off the Northeastern United States. NOAA Technical Memorandum NMFS-NE-115: 149 p.

Cobb, J.E. 1899. The lobster fishery of Maine. Bull. U.S. Fish Comm., 19: 241-265.

Collins, J.W. 1887. The gill-net cod fishery. In Goode, G.B. (ed.) The fisheries and fishery industry of the United States. United States Commission of Fish and Fisheries, Washington, pp. 226-233.

Collins, J.W. 1892. Statistical review of the coast fisheries of the United States. U.S. Comm. Fish and Fisheries, Part XVI, Report of the Commisioner For 1888, pp. 271-378.

Dewar, M.E. 1983. Industry in trouble: the federal government and the New England fisheries. Temple Univ. Press, Philadelphia: 252 p.

Earll, R.E. 1887. The herring fishery of the United States. In Goode, G.B. (ed.) The fisheries and fishery industry of the United States. United States Commission of Fish and Fisheries, Washington, pp. 419-439.

Fiedler, R.H. 1933. Fishery indusrties of the United States, 1931. Rept. U.S. Comm. Fisheries, 1932: pp. 97-440.

Goode, G.B. 1887. The swordfish fishery. In Goode, G.B. (ed.) The fisheries and fishery industry of the United States. United States Commission of Fish and Fisheries, Washington, pp. 317-326.

Goode, G.B. and J.W. Collins. 1887a. The fresh halibut fishery. In Goode, G.B. (ed.) The fisheries and fishery industry of the United States. United States Commission of Fish and Fisheries, Washington, pp. 3-89.

Goode, G.B. and J.W. Collins. 1887b. The George's bank cod fishery.In Goode, G.B. (ed.) The fisheries and fishery industry of the United States. United States Commission of Fish and Fisheries, Washington, pp. 187-198.

Goode, G.B. and J.W. Collins. 1887c. The hake fishery. In Goode, G.B. (ed.) The fisheries and fishery industry of the United States. United States Commission of Fish and Fisheries, Washington, pp. 241-243.

Goode, G.B. and J.W. Collins. 1887d. The mackerel fishery of the United States.In Goode, G.B. (ed.) The fisheries and fishery industry of the United States. United States Commission of Fish and Fisheries, Washington, pp. 247-313.

Hall, A. 1898. The herring fishery of the Passamaquoddy Bay region, Maine. U.S. Comm. Fish and Fisheries Report of the Commissioner. Government Printing Office, Washington: pp. 443-487.

Higgins, E. 1933. Progress in biological inquiries, 1931. Rept. U.S. Comm. Fisheries, 1932: pp. 441-529.

Hoy, D.L. and J.T. Gharrett. 1971. North Atlantic fisheries. In S. Shapiro (ed.). Our changing fisheries. United States Government Printing Office, Washington: pp. 168-183.

Moore, H.F. 1898. Observations upon the herring and herring fisheries of the northeast coast, with special reference to the vicinity of Passamaquoddy Bay. U.S. Comm. Fish and Fisheries Report of the Commissioner. Government Printing Office, Washington: pp. 387-442.

Pearson, J.C. (Editor). 1972. The fish and fisheries of colonial America: a documentary history of fishing resources of the United States and Canada, part II. The New England states. Dept of Interior. U.S. fish and wildlife service, Washington: pp. 178- 678

Sette, O.E. 1932. Outlook for the mackerel fishery in 1932. U.S. Dept. of Commerce Fish. Circ. 10: 25 pp.

Sette, O.E. and A.W.H. Needler. 1934. Statistics of the mackerel fishery off the east coast of North America 1804 to 1930. U.S. Dept. of Commerce Investig. Rept. 19: 48 pp.

German Fisheries

Anon. 1925. Berichte der deutschen wissenschaftlichen Kommission für Meeresforschung - Arbeiten der Jahre 1919-1923. Deutsche wissenschaftliche Kommission fr Meeresforschung, Berlin .

Bahr, K. 1954. Sportlicher Thunfischfang und die Entwicklung des Makrelenangelns seit 50 Jahren. Schriftenreihe des Verbands deutscher Sportfischer e.V. 11:1-23.

Bartz, F. 1964. Die grossen Fischraeume der Welt - Versuch einer regionalen Darstellung der Fischerei-Wirtschaft der Erde. Franz Steiner Verlag GmbH, Wiesbaden.

BFA. 1997. Jahresbericht 1997 - Annual Report 1997. Bundesforschungsanstal für Fischerei,

Blanke, W. 1956. Die Seefischerei Nordwest-Europas - Struktur und Probleme. Verlag - Kroegers Buchdruckerei, Hamburg-Blankenese.

BML. 1951. Jahresbericht über die deutsche Fischerei 1950. Bundesministerium für Ernaehrung, Landwirtschaft und Forsten,

BML. 1976. Jahresbericht ueber die deutsche Fischwirtschaft 1975/76. Bundesministerium für Ernaehrung, Landwirtschaft und Forsten, Report No.ISSN 0075-2851.

BML. 1998. Jahresbericht über die deutsche Fischwirtschaft 1998. Bundesministerium r Ernaehrung, Landwirtschaft und Forsten, Report No.ISSN 0075-2851.

Camphuysen, C. J., B. Calvo, J. Durnik, K. Ensor, A. Follestad, R. W. Furness, S. Garthe, G. Leaper, H. Skov, M. L. Tasker and C. J. N. Winter. 1995. Consumption of discards by seabirds in the North Sea. NIOZ, Rapport Report No.1995 -?

Coull, J. R. 1972. The Fisheries of Europe - an economic geography., G.Bells & Sons, LTD, London.

Cushing, D. H. 1988. The provident sea. Cambridge University Press, Cambridge.

Dankers, N., C. J. Smit and M. Scholl. 1992. Present and Future Conservation of the Wadden Sea - Proceedings of the 7th international Wadden Sea Symposium, Ameland 1990. DLO - Institute for Forestry and Nature Research -Texel, Report No.20 -1992.

Dethlefsen, V. 1973. Zur Frage des Fischvorkommens im Duennsaeureverklappungsgebiet nordwestlich Helgolands. Arch.FischWiss. 24:65-75.

Dittmer, R. 1902. Die deutsche Hochsee-, See- und Kuestenfischerei im 19. Jahrhundert und bis zum Jahre 1902. i.A. des deutschen Seefischereivereins,

Eichelbaum, E. 1913. Das Seemoos und seine Fischerei - Die Seemoosfischerei an der Kueste von Ostfriesland. Abhandlungen des deutschen Seefischerei-Vereins 12:94.

Eichelbaum, E. 1922. Herings-Schleppnetzfischerei. Abhandlungen des deutschen Seefischerei-Vereins 13:

Erdmann, W. 1937. Die deutsche grosse Heringsfischerei in den Jahren 1931 -1934. Berichte der deutschen wissenschaftlichen Kommission für Meeresforschung 8:23.

Goeben, H. 1974. Die Zukunftsaspekte der Kutterfischerei aus der Sicht des Landes Niedersachsen. Arbeiten des deutschen Fischerei-Verbandes 17:101 -111.

Hauke, M. 1974. Die Bedeutung der Kutterfischerei für die Seefischereimaerkte der Bundesrepublik Deutschland. Arbeiten des deutschen Fischerei-Verbandes 17:126 - 133.

Hempel, G. 1977. Fischerei in marinen Oekosystemen - Fishing in marine ecosystems. Verh.Dtsch. Zool. Ges. 67-85.

Hempel, G. 1978. Fisch frisst Fisch - Nahrungsketten und Fangerträge in der Nordsee. Umschau in Wissenschaft und Technik 78:271-277.

Hempel, G. and D. Sahrhage. 1959. Zur Berechnung des Anteils nicht angelandeter und untermassiger Fische im Gesamtfang. Arch.FischWiss. 10:58-67.

Kohl, H. 1954. Die Standortverteilung der Hochseefischerei der deutschen Hochseefischerei und ihre natürlichen Bedingungen. Deutscher Verlag der Wissenschaften, Berlin.

Lasch, R. 1974. Der Frischfischmarkt-Entwicklung und Bedeutung für die Kutterfischerei. Arbeiten des deutschen Fischerei-Verbandes 112-125.

Lenz, W. 1992. Die Ueberfischung der Nordsee - ein historischer Ueberblick des Konfliktes zwischen Politik und Wissenschaft. Pages 87- 108 In: W. Lenz and B. Watermann (eds.) Historisch-meereskundliches Jahrbuch. Dietrich Reimer Verlag, Berlin/Hamburg.

Lindeboom, H. J. and S. J. d. Groot. 1998. IMPACT II - The effects of different types of fisheries on the North Sea and Irish Sea benthic ecosystems. NIOZ-Rapport 1998-1, RIVO-DLO -C003/98, NIOZ-Rapport 1998-1 Report No.C003/98.

Lissner, H. 1927. Die grosse Heringsfischerei in Deutschland. Berichte der deutschen wissenschaftlichen Kommission für Meeresforschung 3:

Meseck, G. 1972. Die Fischwirtschaft in der erweiterten Europaeischen Wirtschaftsgemeinschaft. Schriftenreihe der Schleswig-Holsteinischen Fischwirtschaft 11:9-18.

Meyer, H. A., K. Moebius, G. Karsten and V. Hensen. 1878. Jahresbericht der Commission zur wissenschaftlichen Untersuchung der deutschen Meere in Kiel für die Jahre 1874, 1875, 1876. Koenglich Preussischen Ministeriums für die landwirtschaftlichen Angelegenheiten,

Meyer-Waarden, P. F. 1957. Krebs- und Muscheltiere, 2.Teil: Muscheltiere. Arbeiten des deutschen Fischerei-Verbandes 9:1-68.

Nellen, W. 1978. Das Wattenmeer: Ökologische Kostbarkeit und Goldgrube für den Fischer. Umschau in Wissenschaft und Technik 78:163-169.

Nellen, W. 1979. Fischereiliche Bedeutung des Wattenmeeres. Arbeiten des Deutschen Fischerei-Verbandes (Probleme der Koexistenz von Fischerei und anderen Wirtschaftszweigen bei der Nutzung des Deutschen Festlandsockels der Nordsee) 27:1-16.

Nerdel, W. 1974. Die Kutterfischerei in der Nordsee, die Entwicklung in den letzten 15 Jahren und die Aussichten für die Zukunft. Arbeiten des deutschen Fischerei-Verbandes 17:93 - 100.

Pauly, D., V. Christensen, J. Dalsgaard, R. Froese and F. T. (Jr.). 1998. Fishing down marine food webs. Science 279:860-863.

Reitzenstein, F. 1913. Das Seemoos und seine Fischerei - Das Seemoos und das Korallenmoos, ihre Fischerei, ihre Lebensweise- und Wachstumsverhaeltnisse, speziell an der Küste von Schleswig-Holstein. Abhandlungen des deutschen Seefischerei-Vereins 12:94.

RML. 1925. Jahresbericht ueber die deutsche Fischerei 1925. Reichsministerium für Ernaehrung und Landwirtschaft,

RML. 1939. Jahresbericht über die deutsche Fischerei 1938. Reichsministerium für Ernährung und Landwirtschaft,

Sahrhage, D. 1987. Marine Fischbestandsforschung: ein Teil des Umwelt- und Naturschutzes. Naturwissenschaften 74:305-311.

Schnakenbeck, W. 1928. Die Nordseefischerei. E. Schweizerbart'sche Verlagsbuchhandlung (Erwin Naegele) Gmbh, Stuttgart.

Seefischerei-Verein. 1897. Abhandlungen des deutschen Seefischerei-Vereins. Deutscher Seefischerei-Verein, Berlin.

Tiews, K. 1974. Fischereiressourcen der Nordsee in ihrer Bedeutung für die deutsche Kutterfischerei. Arbeiten des deutschen Fischerei-Verbandes 17:1-19.

Tiews, K. 1990. 35-Jahres-Tend (1954-1988) der Häufigkeit von 25 Fisch- und Krebstierbestaenden an der deutschen Nordseekueste. Archiv für Fischerei Wissenschaften 40:39-48.

Tiews, K. and A. Schumacher. 1982. Assessment of brown shrimp stocks (*Crangon crangon*) off the German coast for the period of 1965-1978. Arch. FischWiss. 32:1-11.

Willer, A. 1945. Zoologisches aus der Deutschen Fischereistatistik. Archiv für Hydrobiologie 40:1063-1077.

Wolfff, W. J. 1983. Ecology of the Wadden Sea - Fishes and fisheries/Birds/Marine Mammals/Pollution. A.A.Balkema, Rotterdam.

United Kingdom

Aflalo, F.G. 1904. The Sea Fishing Industry of England and Wales. E. Stanford, London.

Aitkin, A., Mackie, I., & Windsor, M. 1982. Fish Handling and Processing. Ministry of Agriculture, Fisheries, and Food. HMSO, London. pp.28-115.

Anon. Report of the North Sea Roundfish Working Group. ICES Doc. C. M., 1981, G:8

Anon. Report of the Herring Assessment Working Group for the Area South of 62N. ICES Doc. C. M., 1988, 17.

Anon. Report of the North Sea Flatfish Working Group. ICES Doc. C. M., 1988, Assess:9.

Anon. Report of the Herring Assessment Working Group for the Area South of 62N. ICES Doc. C. M., 1990, Asses:14.

Anon. Report of the North Sea Roundfish Working Group. ICES Doc. C. M., 1990, Assess:7.

Anon. Report of the Herring Assessment Working Group for the Area South of 62N. ICES Doc. C. M., 1991, Assess:5.

Anon. Report of the North Sea Flatfish Working Group. ICES Doc. C. M., 1991, Assess:9.

Anon. Report of the North Sea Roundfish Working Group. ICES Doc. C. M., 1991, Assess:4.

Anon. Report of the North Sea Flatfish Working Group. ICES Doc. C. M., 1992, Assess:6.

Anon. Working Group on the Assessment of Demersal stocks in the North Sea and Skagerrak. ICES Doc. C. M., 1993, Assess:5.

Board of Agriculture and Fisheries. Sea Fisheries. Fishery Investigations, 1(1):2.

Board of Trade (Dep. of Labour Statistics). 1915. Great Britain Abstracts of Labour Statistics, 17-19. HMSO, London.

Burd, A. C. 1978. Long-term changes in North Sea herring stocks. Rapp. P. – V. Reun. CIEM, 172: pp.137-153.

Clarkson, L. 1990. British Trade Union and Labour history: A Compendium. The Economic history Society, London. 108p.

Coull, J. 1996. The Sea Fisheries of Scotland: A Historical Geography. John Donald Pub., Edinburgh. pp15-17,142-153.

Coull, J. R. 1972. The Fisheries of Europe. An Economic Geography. G.Bell and Sons Ltd., London.

Daan, N., Hessen, H. J. L., & Pope, J. G. 1994. Changes in the North Sea cod stock during the twentieth century. *ICES Marine Science Symposium*, 198:229-243

Daan, N., Richardson, K., & Pope, J.G. 1996. Changes in the North Sea ecosystem and their causes. *ICES Journal of Marine Science*, 53(6):879-883.

Dyson, J. 1977. Business in Great Waters. London, Angus and Robertson. pp.111-257.

Graham, M. 1956. Sea Fisheries: Their Investigations in the UK. E.Arnold, London.

Graham, M. 1939. Effect of fishing plaice in the North Sea. *Rapp. P.-v. Reun. Cons. int. Explor. Mer*, 110.

Graham, M. 1960-1963. The state of the northern stocks of cod. *Rapp.P.-v.Reun.Cons.int. Explor. Mer. Rapports*, 133-140.

Gray, M. 1978. The Fishing Industries of Scotland 1790-1914: A Study in Regional Adaptation. Oxford University Press, London.

Gulland, J. A. 1971. The Fish Resources of the Ocean. Rome, FAO. 15.

Hardy, A. 1959. The Open Sea II: Fish and Fisheries. Collins Press, London.

Hatcher, A. C. 1977. Producers organizations and developed fisheries management in the UK: Collective and individual quota systems. *Marine Policy*, 21(6):519-533.

Hempel, G. 1978. Synopsis of the symposium on North Sea fish stocks-recent changes and their causes. *Rapp. P.-v. Reun. Cons. Int. Explor. Mer*, 172:445-449.

Holden, M. J. 1978. Long term changes in landings of fish from the North Sea. *Rapp.P.-v. Reun.Cons.int.Explor.Mer*, 11-26.

Holden, M.J. 1981. Long term changes in landings from the North Sea with particular reference to the periods 1930-38 and 1960-69. Symposium on the Changes in North Sea Fish Stocks and their Cause. No. 15.

ICES 1910. Bulletin Statistique des peches maritimes des pays du nord de l'Europe: *Cons.int.Explor.Mer.*, 3-9.

ICES 1953-1955. Bulletin Statistique des peches maritimes des pays du nord de l'Europe: *Cons.int.Explor.Mer.*, 32-43.

Kondrashova, N.G. 1984. Shipboard refidgeration and fish processing equipment. Amerind Pub. Co. Pvt. Ltd., New Delhi. 222 p.

Meyers, R. A., Bridson, J, and Barrowman, N. J. 1995. Summary of worldwide stock and recruitment data. Can. Tech. Rep. Fish. Aquat. Aci., 2024: 274 p.

Ministry of Agriculture, Fisheries, and Food. 1994. Sea Fisheries Statistics:1991 and 1992. HMSO, London. pp. 3-63.

Organisation for European Economic Co-operation. 1960. Fisheries Policies in Western Europe and North America.

Parish,B. B. and Jones, R. 1952. State of the haddock stocks in the north sea: 1946-1950. *Marine Research*, 4.

Pope, J. G. 1978. An evaluation of the stock structure of North Sea cod, haddock, and whiting since 1920, together with a consideration of the impacts of fisheries and predation effects on their biomass and recruitment. ICES J. Mar. Sci, 53:1157-1169.

Posgay, J. A. Status of fisheries and research carried out in subarea 4 in 1971. ICNAF Res. Doc. 72/131. 5 p.

Report of the ICES Advisory Committee on Fishery Management, 1999. *ICES Coop.Res.Rep*, 236(1):234-292.

Ritchie, A. 1960. The Scottish seine net fishery:1921-1957. *Marine Research*, 3.

Rijnsdorp, A.D., Daan, N., Beek, F.A., and Heesen, H. J. L. 1991. Reproductive variability in North Sea plaice, sole, and cod. J. Cons. Int. Explor. Mer. 47:352-375.

Sahrhage, D. & Lundbeck, J. 1992. A History of Fishing. Springer-Verlag, Berlin.

Serchuk, F. M., Kirkegaard, E., & Daan, N. 1996. Status and trends of the major roundfish, flatfish, and pelagic fish stocks in the North Sea: Thirty year overview. *ICES Journal of Marine Science*, 53(6):1130-1145.

Starkley, D. J., & Jamieson A. 1998. Exploiting the Sea. Exeter, University of Exeter Press.

Steele, J. H. 1958. Plant production in the northern North Sea. *Marine Research*, 7.

Steele, J.H. 1961. The environment of a herring fishery. *Marine Research*, 6.

Symes, D. 1991. UK demersal fisheries and the North Sea: Problems in renewable resources management. *Geography*, 76(2):131-142.

Symes, D. 1998. Northern Waters: Management and Practice. Fishing New Books, London. pp.195-246.

The European Commision. The Common Fisheries Policy.

UK Sea Fisheries Statistics 1950. HMSO, London. 1951.

UK Sea Fisheries Statistics 1991 and 1992. HMSO, London. 1994

Varela, R.A., Cruzado, A., and Gabaldon, J.E. 1995. Modelling primary production in the North Sea. *Netherlands Journal of Sea Research*, 33:337-361.

Wimpenny, R. S. 1948-1951. The English plaice fishery in the southern North Sea and English Channel in 1948. *Annales Biologiques CPIEM*, 5-8.

Wren, C. 1988. Industrial Subsidies: The UK Experience. Macmilllan Press, London. pp.123-170.

Appendix 1: Participants, Authors and Guests

Alder, Jackie (Author)
Consultant
Fisheries Centre
University of B.C.
2204 Main Mall
Vancouver, B.C. V6T 1Z4
email: j.alder@fisheries.ubc.ca

Christensen, Villy (Author)
Visiting Scientist
Fisheries Centre
University of B.C.
2204 Main Mall
Vancouver, B.C. V6T 1Z4
email: v.christensen@fisheries.ubc.ca

Guénette, Sylvie (Author)
Post-doctoral Fellow
Fisheries Centre
University of B.C.
2204 Main Mall
Vancouver, B.C. V6T 1Z4
email: s.guenette@fisheries.ubc.ca

Froese, Rainer (Author)
FishBase Project
ICLARM
Los Baños, Phillipines
email: r.froese@cgiar.org

Haggan, Nigel
Sea Around Us Project Coordinator
Fisheries Centre
University of B.C.
2204 Main Mall
Vancouver, B.C. V6T 1Z4
email: n.haggan@fisheries.ubc.ca

Longhurst, Alan (Author)
Place de d'Eglise
46160 Carjac, Lot, France
email: longh@crdi.fr

Maclean, Jay (Guest)
Asian Development Bank
P.O. Box 789, 0980 Manila
Philippines
email: jmaclean@adb.org

Munro, Gordon (Author)
Department of Economics
University of B.C.
#997 - 1873 East Mall
Vancouver, B.C. V6T 1Z1
email: munro@econ.ubc.ca

Nelson, Jay (Guest)
The Pew Charitable Trusts
1700-2005 Market Street
Philadelphia, PA 19102-7017
email: jnelson@pewtrusts.com

Ommer, Rosemary (Guest)
Calgary Institute for the Humanities
The University of Calgary
Calgary, AB T2N 1N4
email: ommer@ucalgary.ca

Pauly, Daniel (Principal Investigator, Sea Around Us Project) (Author)
Fisheries Centre
University of B.C.
2204 Main Mall
Vancouver, B.C. V6T 1Z4
email: d.pauly@fisheries.ubc.ca

Pitcher, Tony (Chair, Sea Around Us Project Steering Committee) (Author)
Director
Fisheries Centre
University of B.C.
2204 Main Mall
Vancouver, B.C. V6T 1Z4
email: t.pitcher@fisheries.ubc.ca

Platt, Trevor (author)
Department of Oceanography
Dalhousie University
Halifax, NS B3H 4J1
email: tplatt@is.dal.ca

Poon, Amy (Student Rapporteur)
Fisheries Centre
University of B.C.
2204 Main Mall
Vancouver, B.C. V6T 1Z4
email: a.poon@fisheries.ubc.ca

Rizzo, Yvette (Student Rapporteur)
Fisheries Centre
University of B.C.
2204 Main Mall
Vancouver, B.C. V6T 1Z4
email: y.rizzo@fisheries.ubc.ca

Ruttan, Lore (Author)
Post-doctoral Fellow
Fisheries Centre
University of B.C.
2204 Main Mall
Vancouver, B.C. V6T 1Z4
email: loreruttan@hotmail.com

Sherman, Kenneth (Author)
National Marine Fisheries Service
Narragansett Laboratory
Naragansett, RI 02992-1199
email: Kenneth.Sherman@noaa.gov

Sathyendranath, Shubba (Author)
Biological Oceanography Division
Beford Institute of Oceanography
Dartmouth, NS B2Y 4A2

Sumaila, Rashid (Author)
Research Associate
Fisheries Centre
University of B.C.
2204 Main Mall
Vancouver, B.C. V6T 1Z4
email: r.sumaila@fisheries.ubc.ca

Tyedmers, Peter (Author)
Resources Management and Environmental Studies
University of B.C.
Vancouver, B.C. V6T 1Z3
email: tyedmers@mntn.net

Walters, Carl (Guest)
Fisheries Centre
University of B.C.
2204 Main Mall
Vancouver, B.C. V6T 1Z4
email: walters@fisheries.com

Watson, Reg (Author)
Research Associate
Fisheries Centre
University of B.C.
2204 Main Mall
Vancouver, B.C. V6T 1Z4
email: r.watson@fisheries.ubc.ca

Zeller, Dirk (Author)
Post-doctoral Fellow
Fisheries Centre
University of B.C.
2204 Main Mall
Vancouver, B.C. V6T 1Z4
email: d.zeller@fisheries.ubc.ca

Appendix 2: Invited Project Evaluators

Alverson, Dayton "Lee"
Natural Resources Consultants Inc.
207-1900 West Nickerson Street
Seattle, WA 98119
email: lschaff@nrccorp.com

Cochrane, Kevern
Fishery Resources Divison
Food and Agriculture Organization of the United Nations
00100 Rome, Italy
email: Kevern.Cochrane@fao.org

Degnbol, Poul
Institute for Fisheries Management & Coastal Community Development
North Sea Center
P.O. Box 104
9850 Hirtshals
Denmark
email: pd@ifm.dk

Fanning, Paul
Marine Fish Division
Fisheries and Oceans Canada
Bedford Institute of Oceanography
Box 1006
Dartmouth, NS B2Y 4A2
email: FanningP@mar.dfo-mpo.gc.ca

Grainger, Richard
Food and Agriculture Organization of the United Nations
00100 Rome, Italy
email: Richard.Grainger@fao.org

APPENDIX 3: EXTERNAL EVALUATORS COMMENTS

Lee Alverson

I like the project. I did not say that before, but now I am changing my direction. You are looking at a broad spectrum of issues and have good leadership. What needs to happen now is to determine how well the project can be improved upon.

The bycatch example was well done but it was a narrow focus which needs to be expanded to other fisheries. You may wish to get hold of Steve Murawsky's work on the New England area. And there are some good papers that relate to the shrimp and trawl fisheries.

Fishing mortality needs to be looked at and quantified. You can look at the work done in Norway. There are some very good estimations of mortalities due to fish passing through nets and very good estimates for ground fish and other species. This will give you an added factor of mortality. However, you need to be careful on how to apply it. The very young fish, which have high mortality, will be less impacted by this mortality. If done well, this data will add credibility to the documentation that you are doing.

I mentioned that you have a fairly wide focus and the only danger is whether you can do all the studies you want as well as your main themes. My concern is not that you should not do these studies. However, some economic matters are more difficult to do and can detract attention from your main objectives.

I have a suggestion that does not relate to your work as such. The impacts this work will have and how you will give out the results need to be carefully evaluated. It is not to the credit of this group or even appropriate that we have not done a good job of managing fisheries. Industry management is not the solution for getting better decisions. NGOs can go to political groups, but fisheries managers and scientists are part of the process and will need to be convinced. I am referring to the way in which the Pew Charitable Trusts will articulate the results to the general public, to decision-makers and to NGOs.

I am glad the project is under way.

Kevern Cochrane

I think this is a very important, relevant and worthwhile study, and congratulate all involved in its development, including the Pew Charitable Trusts.

What makes this project particularly important is its holistic and broad approach in addressing the impact of fisheries on marine ecosystems, the contribution of fisheries to human society, and the means to reconcile the impacts of fisheries and the needs of the human society. Such a holistic and comprehensive look at ecosystems and society is something unique and important. If the project does achieve its aim, it will influence the direction of fishery studies in the future.

I do have some concerns. The methods you have selected are by and large appropriate and we have not been able to come up with any criticisms in the broad approaches you have chosen. The biggest problem you will have to confront is the availability of data, and the assumptions and compromises you will have to make in filling in the gaps. The only defense against criticism in this area is that you fully describe all sources, assumptions and methods used in the compilation of the data used and that you consider the implications of all uncertainties in interpreting your results. Another important set of decisions will be which data not to use when you have a lot of data available for some areas. In these instances, you will need to be very clear as to why you have made those decisions.

The second major methodological issue of concern is whether to go for simple approaches that are not very demanding of data or to go for sophisticated, data intensive methods. The project philosophy is to use simple, robust methods, which can be applied globally, including in the less studied countries and regions of the world.

That is a very commendable and worthwhile attitude and from a FAO perspective, it would mean that the project is more applicable to developing countries. However, you have chosen for your first phase the North Atlantic, probably the most intensively studied marine area in the world. In this region, the project will not be judged in terms of economy in the use of data, but on how well you did the job for the North Atlantic with all the information available. You need to bear that in mind. How you do that is difficult, but I recommend that you use the best data and the best methods available, even if the latter are complex and data-intensive, at least for some of the areas in the North Atlantic. This should not only help to minimize criticism and increase the acceptance of the results, but could also lead to acquiring valuable insights for application in the less data rich areas. The trade-off between use of the most appropriate methods given the data available, and the desire and need for general application of methods is a difficult one, but I do think you will have to do some areas at the level of greatest sophistication and rigor possible. When you move into the less data rich areas, that will be the opportunity to apply more general methods and to refine those.

Another concern is the limited time that is left in this phase in which to accomplish a huge amount of work. You do have a network of consultants also working on the project and we, the reviewers, do not have full details on those and how much time they are dedicating to the project. It is therefore difficult to come up with a valid opinion as to whether you can or cannot achieve your aims in the remaining year - you will be the best judges of that. If you feel that you cannot achieve all the aims, my recommendation is that it would be better to complete 75% of the puzzle well, rather than doing the whole thing in a shoddy way. It can come down to a sacrifice either on locations included or the coverage of themes. Cutting back on either, but doing the remainder well, would be my preferred option.

A final comment on the relevance of policy to this project. You are making assumptions about policy in your project design and implementation, emphasizing some aspects of policy and de-emphasizing others. For example, a major policy assumption is that conservation and the sustainable use of resources is a desirable goal. Given the globally accepted fishery instruments such as the Law of the Sea and the FAO Code of Conduct for Responsible Fisheries, that assumption is easily justified. However, there are others included in the project, such as assuming that maximizing gross energy efficiency, emphasizing the social contribution of fisheries or obtaining maximum gross value are general fisheries objectives, which are not easily validated. Undoubtedly, each of these is likely to be important in some fisheries, but not as important in others. I suggest that you need to interpret your results and express your conclusions within the context of the existing policy, both as written and as actually applied, for each fishery and country. That will require that you have the necessary background on current policies within the fishing nations of the North Atlantic. Obtaining this is not currently included in the project, but it need not be a very big task. You could contract a consultant to compile a review based on available material and this could be a source of reference material for the entire group. I suggest that you do this earlier rather than later, so people can get some ideas of what the different fisheries are working towards as they implement their own work programs.

Good luck.

Poul Degnbol

The project must be evaluated in relation to the six basic questions to be answered as listed in the overview paper.

The need for coherence and clear relation between overall project structure and the basic questions.

The project structure may seem complicated, but there is a backbone of tasks addressing the questions, as follows:

i. Estimation of total kills/removals;

ii. Evaluation of ecological and economic (losses) impacts;
iii. Overall evaluation of status – synthesizing and concluding from i and ii;
iv. Identification of policy and management measures.

Within the project, this flow of tasks is to be fulfilled by a backbone of subprojects, namely:

i. Compilation of catch/efforts statistics and removals estimation;
ii. Ecopath/Ecosim, ecoeconomic analysis;
iii. RAPFISH;
iv. Ecosim.

The definition of units / systems is a background task for all these

Other subprojects provide supplementary inputs to the backbone.

The pattern of subprojects, particularly the backbone projects, does generally fit in well to serve their role within the project. The weakest part is policy and management. Ecosim will provide some information on the biological/technical options. The compliance study will provide some knowledge on policies and agreements. But the reasons why good intentions as stated in policies and agreements are not followed, must be identified if the project is to answer fully question 6 of the overall objectives (see first contribution in this volume. This involves understanding the interests involved and the institutions of policy formulation and fisheries management. This means answering the questions as to which interests are at play, what are the relative powers on various institutions and what are the structural bottlenecks in policy making and management changes. This may not be within reach of the present project but it is nevertheless necessary to keep this endpoint within view.

This understanding of the objectives and the backbone of the project can be used for focusing and prioritization, both of which are needed.

Focusing – The need to evaluate whether subprojects address the specific objectives in relation to overall backbone.

For each subproject, check whether what is to be done actually serves the purpose within the project, which parts of this subproject contribute directly to answering the 6 questions and which parts do not. Some parts of subprojects may be interesting to do, but are outside the overall objectives. Other important components may be missing. For example, the coastal/depth transects must be seen as descriptions of fisheries/stock interactions in time and space. They should be developed to be just that. If other additions are made, they may obscure this purpose and should not be included. The market study is only relevant if it is put in the policy-management context (through green labeling as indicated) and should be modified to ensure that the results are usable in this context. The RAPFISH component is to synthesize and present the status of fisheries as evaluated through other subprojects.

Prioritization.

Subprojects should be prioritized according to their role in relation to the backbone. If it is necessary for time and resource reasons to go from cover-all-mode to case-mode, this should first be done for those projects which are supplementary to the backbone. Again, the policy and management end is the most shaky and could do with prioritization and even addition. Ecosim and the compliance analysis are the most central subprojects here but the compliance analysis could be expanded to include investigations of the reasons for shortcomings – this is however a large task in its own right and may not be doable within time and resource constraints.

Specific advice

Catch data, including the best estimates of landings outside official statistics from the east side of the Atlantic, are best obtained from the times series of 'unallocated landings' given in ICES working group reports. These figures have been estimated year by year by the most knowledgeable biologists in that year and no other source will match this. Moreover, these figures are available for quotation. To understand the background for this data, it will be necessary to go into each year's reports and check the comments. If only the data on unallocated landings (without explanations) are needed, a limited number of reports are required as each contains the last 10 years of data. To get the detailed data (yearly) the best option would be to have a team member working in the ICES library in Copenhagen or in another institution owning the full report set for maybe a week to dig this out.

For fleet disaggregated landings data the best source for EU countries would be the Multi Annual Guidance Program 4 Expert Group Report, published in 1997. Here detailed primary data from each nation were utilized and compiled providing a good overview of European fleets and their landings.

And a small point of irritation – the americo-centric elevation of Chesapeake Bay to a 'Large Marine Ecosystem' of global significance sends very wrong signals to partners and readers in other parts of the world. There may be enough basis for accusations of ethnocentrism in the project already – the most problematic probably being that poor countries can not afford the luxury to establish the knowledge needed to take ecosystem considerations into their fisheries management – and there is no need to add more.

Overall evaluation

Overall I think the project – with some focusing and prioritization - will be able to answer the initial questions. The methods chosen for the backbone of the project are adequate. This is a highly relevant project which will prove useful on a global scale.

Richard Grainger

I said at the beginning that the objectives were good, addressing fundamental questions with several new approaches and some new tricks. I was particularly attracted by the broad based approach applied to this wide geographical area where intensive fishing has indeed taken place for a long time. At the end of the workshop, my views have not changed but I am more enthusiastic about the project.

I particularly welcome the plan to estimate the total fishery removals from this region. It is really important that this is done in a credible way with good documentation and a reproducible methodology. It can be done and much better estimates can be developed. I only plea that it is documented well.

I also hope that standard terminology and concepts are employed, such as nominal catch (live weight equivalent of the landings). It is important to maintain credibility with countries and regional fishery bodies and thus to retain standard concepts. In several of the papers there were references to the non-inclusion of discards in official nominal catch statistics, as if this were a shortcoming. Those statistics were not designed to describe total removals but rather to indicate the contributions of fish to food supply and of fisheries to the national economy (e.g. in national economic accounts). Discard data are an essential supplement to landings data in order to describe the impact of fishing on the resources (fishing mortality), but estimated discards should be kept separate from the landings data as they have no economic value.

The exploration of alternative ecosystem-based management regimes is an interesting one, but this is one area which can probably only be investigated in general terms, because of time constraints. The same applies to the energy consumption study, which has some novel and interesting ideas. These studies could however provide some initial indications and be a good basis for later studies. The methodology for RAPFISH is better established and may provide more conclusive results. In FAO we are particularly interested in using RAPFISH for monitoring implementation of the Code of Conduct and will be anxious to see the results of this study.

With regard to tracking landings, I still think it is important, but I am concerned that the component devoted to profit margin analysis may be too ambitious. Getting data on the costs and benefits for different steps in the utilization chain will be very difficult. I suggest it would be better to scrap the survey relating to individual processors and others and concentrate on obtaining data from some of the industry federations. It will be a much more efficient way of gathering data. Fishery industry federations often have lots of data and you may try to come to some arrangement with them.

I have a note of caution. It struck me that FAO catch data have been mentioned several times as a basis for raising different variables to try to get overall totals for the North Atlantic (e.g. raising effort, fuel consumption and even catch data themselves). There is a need to guard against circularities and spurious relationships arising from this.

When scoring components this morning, I noticed that for most papers I had scored high against the categories concerning contribution to the overall SAUP goals and lower to the categories concerning achievability and the possibility of refining the methodology within the project time frame. My concern is that there is only one year to go and this is a very ambitious project. Given the time frame, the results will be exploratory in many cases, but nonetheless very useful as a basis for further study in the North Atlantic and elsewhere. It is important to be realistic and not to raise expectations that this will be the definitive analysis of fishery impacts on North Atlantic ecosystems.

It is fair to say that FAO has a real interest in seeing this project succeed. At the moment we are in the process of developing a draft international plan of action and mechanisms to improve fishery status and trends reporting. The aim is to get countries to make commitments to gather data, and to exchange information with regional and global fishery bodies in a more systematic way in order to generate more comprehensive and reliable status and trend reports on fisheries at local, national, regional and global levels. We envisage that this plan of action will lead to the development of a partnership arrangement, that will allow partners such as regional fishery bodies and centers of excellence contribute data (on their terms) and benefit from access to other data with the purpose of improving status and trends reporting overall. My real hope is that this project will provide a very good example of how taking a broader perspective and assembling data from many sources can do just this, and thus be a spur for countries to adopt the plan of action.

Paul Fanning

Poul's explanation was extremely well done. I do not have much to add about the specifics of the project. As someone within the project, I have bought into it a long time ago.

In my view there is a need to work at various multiple scales. This applies to both components of the project. We have to decide how much emphasis to put on each component and make sure the things we decide are central to going ahead and not do everything poorly. We need to look at multiple scales within the ecosystems that we have looked at. Some systems we can look at in all complexity, other systems need to be dealt with at a lower and broader scale. Taking the scale to the level of the analysis involved, we have the opportunity to do very detailed analysis in several systems. These should be used where possible, and compared to assumptions for other systems which are done in a more general approach, so they can give validation to simple (not robust) processes. There is a need to work on a variety of levels of sophistication in the level of detail.

Jay Nelson

I agree with reviewers' comments. I do not see any conflicts, but a lot of convergence. The team will take those messages seriously. The concern I have is how much the team can do in a year.

I would like to comment on the misuse of data. The most important thing for the Pew Charitable Trusts is that the report and information that come out of this project are credible solid science that will be accepted within the scientific community at large. If that is not true, it is of less value to us. We want numbers to show to the public. The public cares about the ocean but it does not understand the ocean and does not know what to do. Hopefully the kind of information presented in the report will get enough public interest. If the public is ignorant, policy makers will not act. Once the report is out, it is not our intention to misuse the data but there is always the chance that someone can take information and not use it well. We do give grants to advocacy groups but we cannot control their use of data fully. We are definitely not interested in abuses happening. Obscure data are easier to abuse, but well-grounded data is hard to abuse. I really appreciate the reviewers' comments and the thought and time put into this. It is an honor to be with you. I started up as a scientist and I appreciate the effort you have all put into it. I like to thank Daniel for coordinating this meeting and the project. Thumbs up for the report and a good job.

Lee Alverson

I want to clarify that what I wanted to say was the value in the report lies in how it is articulated from Pew – I did not use the term misuse the data. I also think it is important to give operational definitions in the introduction.

Authors' Index