Reports from the FSS mini-symposia
2004 – 2005

by

Edward A. Codling & Ciaran J. Kelly (Editors)

Marine Institute, Fisheries Science Services, Galway Technology Park,
Parkmore, Galway
# Table of contents

### Acknowledgements

1. Introduction

2. FSS mini-symposium 2004 – ‘Modelling the dynamics of fisheries’
   - Introduction to FSS mini-symposium 2004
   - Paper 1, 2004: Modelling marine fish and fisheries dynamics: a new paradigm, by Ciaran Kelly
   - Paper 2, 2004: The Irish Sea cod recovery plan: some lessons learned, by Edward Codling and Ciaran Kelly
   - Paper 4, 2004: Simulation of fishery management using total allowable catch, by Marco Kienzle
   - Paper 5, 2004: Sound: the essential navigation cue for young reef fishes to find their way home, by Stephen Simpson
   - Paper 6, 2004: Discarding by the demersal fishery around Ireland, by Lisa Borges et al.
   - Paper 7, 2004: How consistent are the replicated age estimates of experienced blue whiting age-readers?, by Gavin Power et al.

3. Discussion from FSS Symposium 2004

4. Appendix – List of participants

### 3. FSS mini-symposium 2005 – ‘Bridging the gap: the science and management of fisheries’

- Introduction to FSS mini-symposium 2005
- Paper 1, 2005: ‘Cheap and dirty’ fisheries science part I, by Ciaran Kelly and Edward Codling
- Paper 2, 2005: ‘Cheap and dirty’ fisheries science part II: an example of fisheries management using simple indicators, by Edward Codling and Ciaran Kelly
- Paper 3, 2005: Simple stochastic models for fish and fisheries, by Jon Pitchford
- Paper 4, 2005: From ecology to fisheries management: Celtic Sea herring, by Leonie Dransfeld
- Paper 5, 2005: Technical conservation measures in the Irish Nephrops fishery, by Emmet Jackson and Dominic Rihan
- Paper 6, 2005: The role of gear technology and selectivity in the advisory process, by Dominic Rihan
- Paper 7, 2005: The management of shellfisheries in Ireland, by Oliver Tully

Discussion from FSS Symposium 2005

Appendix – List of participants
Acknowledgements

We are grateful to all those who participated in the two symposia (a full list is given in the Appendix) and we are especially grateful to those who made presentations at either or both meetings and to those who wrote summary papers for this publication. Obviously the success of a symposium is entirely dependent on those who participate in it, and it is a reflection of those who attended that the quality of the papers and discussion ideas presented here is so high.

We are particularly grateful to Geraldine Kane for helping to organise both symposia and to the management and staff of the Harbour Hotel, Galway for facilitating both meetings in such a professional manner.
1. Introduction
FSS mini-symposia 2004 – 2005

The mini-symposia documented in this publication were meetings organised in October 2004 and August 2005 by the ‘Modelling and Simulation’ team in Fisheries Science Services (FSS) of the Marine Institute, Ireland. Both symposia took place at the Harbour Hotel in Galway, Ireland. Each meeting consisted of a number of presentations (given as talks or posters) followed by a round-table informal discussion session. The two meetings were attended by participants from FSS and the Marine Institute, BIM (Bord Iascaigh Mara – Irish Sea Fisheries Board), FRS (Fisheries Research Services) Aberdeen, and universities in both Ireland and the UK. The Appendix contains a full list of the participants at each meeting, while contact details for those who gave presentations are given at the start of each summary paper.

The meetings aimed to bring together academics, fisheries scientists and those involved in fisheries management, in order to exchange ideas and allow discussion of the current research and issues being faced. The meetings were well attended by academics and fisheries scientists but unfortunately there was no participation from those directly involved with fisheries management in Ireland (DCMNR). However, a number of collaborations were formed through contacts and discussion made at the two meetings, and work is ongoing on several projects involving FSS and others. The symposia fulfilled a useful purpose in allowing face-to-face discussion of the most important research and issues currently affecting fisheries in Ireland and elsewhere. It is anticipated that similar meetings will be arranged in the future and it is hoped that the positive outcomes of the first two FSS symposia can be built upon and further collaboration can be maintained with all participants.

Reports

This publication is arranged into two main sections reporting the FSS symposia in 2004 and 2005. In each section, we present summary papers for every presentation made at the symposium, followed by a brief review of the main points arising from the discussion session. Although the summary papers have not been formally peer-reviewed, they still serve as a useful record of the current state of fisheries research and policy in Ireland and elsewhere. Many diverse topics were covered over the two meetings ranging from studies into discarding of catches, new simulation methods for analysing fisheries management strategies, experimental work on fish acoustics or technical gear trials, stochastic fisheries modelling, and discussion of various new management frameworks. Several authors made presentations at both the 2004 and 2005 symposia, and in those cases where the two presentations are similar in style and content we have included one summary paper only in the 2005 section of the publication.
2. FSS mini-symposium 2004 – ‘Modelling the dynamics of fisheries’
Introduction to FSS mini-symposium 2004

The 2004 FSS mini-symposium was advertised with the following objectives:

‘The mini-symposium is intended to promote debate on modelling work that underpins strategies for sustainable fisheries. Key to this aim is the use of theoretical and simulation models to gain an insight into the population dynamics of fish stocks due to the interactions between fish, fishery and environment. This meeting aims to bring together fisheries scientists working on theoretical modelling and/or involved in more ‘hands on’ approaches, in order to exchange ideas and allow discussion of the current issues being faced.’

The meeting was attended by representatives from the Marine Institute and BIM (Ireland), FRS (UK) and various universities in the UK and Ireland (see Appendix). Although the theme of the meeting was ‘modelling of fisheries’, a diverse range of presentations was encouraged relating to all aspects that may affect the formulation, use of and interpretation of fishery models.

The first paper presented here (Paper 1, 2004 by Ciaran Kelly) describes the context in which theoretical models of fishery dynamics are used in the ICES (International Council for the Exploration of the Sea) community, and how these models are used to give advice on which annual quotas in the EU are based. There is currently a state of great change in the way fisheries science and management is undertaken and the paper discusses a new paradigm that has been suggested. Paper 2, 2004 by Edward Codling and Ciaran Kelly introduces a new stochastic simulation tool that has been used to illustrate the failure of the Irish Sea cod recovery plan. The paper demonstrates how a much more drastic cut in fishing mortality than has been so far produced in the recovery plan is required if the stock is to have any chance of recovering in the short to medium term.

The next papers are more theoretical in style and illustrate how theoretical modelling can be used to gain an insight into how complex systems may function in the real world. Paper 3, 2004 by Paul Baxter et al presents a simple stochastic (probability based) model for the processes involved in larval development and fish recruitment. Results from simulations of the model under particular scenarios are used to illustrate how it is important to consider the variability and random nature of larval growth if we are going to properly understand how these processes occur in reality. The next paper (Paper 4, 2004 by Marco Kienzle) describes a simulation of a fishery managed using only information about the age structure of the virtual population. The results presented show that, at least in theory, it should be possible to manage fisheries using much less data than is currently required (a typical ICES assessment requires much more data than just the age structure of the population). The use of an ‘indicator’ as a proxy of the state of the stock could be highly useful for stocks where there is little data available or where this data is expensive to collect. Further discussion of ‘indicator management’ is given in Papers 1 and 2 from the 2005 symposium and in the discussion section from 2005.
The remaining papers from the 2004 symposium do not involve theoretical modelling directly, but the ideas they raise are important nonetheless. It is a key point to remember that any theoretical model needs to be validated against evidence gathered from observations or experiments in the real world, and once validated, the results of a model are only as good as the data that are used with it. Paper 5, 2004 by Stephen Simpson details experiments completed in the field looking at the behaviour of early life-stage tropical reef fish. There is strong evidence from the experimental results that settlement-stage reef fish are attracted to reef noise. The paper then discusses the implications of this in relation to future modelling work and possible management strategies. The next paper (Paper 6, 2004 by Lisa Borges et al) presents data collected from discard sampling trips completed around Ireland over the last 10 years. The data clearly shows that different fleets have different discarding strategies even when using the same gear in a similar area. Discards are often ignored or assumed to be constant across all fleets when using simple fishery models or completing stock assessments and the paper demonstrates that this may be a significant problem. The final paper (Paper 7, 2004 by Gavin Power et al) makes a similar point about the quality (or lack of quality) in data used in typical fisheries models or assessments. Comparisons between age readers show significant discrepancies in the aging of blue whiting, which could clearly cause problems in the typical age-based assessments completed by ICES.

The final part of this section is a report of the round-table discussion session completed at the 2004 symposium. The discussion session consisted of a number of questions being raised by the chairman. The group discussed each question in turn and a record of responses and comments was kept. The report presented here is a summary of the main responses and comments and acts as a record of the general consensus of the group. The responses and comments presented should not be taken as indicative of the personal opinion of any of the individual participants.
1. Introduction

Since the inception of the EU Common Fisheries Policy, managers have set out to control the exploitation of marine fish resources by limiting the quantities removed each year (TAC: Total Allowable Catch). This policy requires matching the TAC to the stock size each year. For this to be effective, scientists must be able to determine the stock size each year, and the TAC must be enforced. There are a number of flaws to the system, which have at least contributed to the poor state of many fish stocks in the EU zone.

However, hubris is slowly giving way to a new paradigm, which embraces the real complexity of natural marine systems. This new approach does not set out to determine the state of any population in a given year; rather the approach is based on incorporating real knowledge of stock and fisheries dynamics in a stochastic framework. The aim is to establish long-term harvest strategies based on targets and thresholds, which are designed to maintain fish stocks with buffer levels. This strategy brings with it stability in harvest levels, which avoids the ‘gold rush’ and the commercial pressure to harvest at maximum level.

This paper will review briefly the current ‘life cycle’ of the TAC (principal management instrument of the common fisheries policy), and introduce the models, which are used to support this. The framework of the new approach will be explained, and areas where research needs to be focused will be highlighted.

2. The System

The current system of fisheries management in the EU is comprised of fishermen, scientists and fisheries managers who interact on a national and international scale (Fig. 1).
Scientists interact with fishermen on a national level and collect fisheries dependent and independent data. This is analysed by national scientists at an international meeting (ICES). ICES then provide information on stock status and short-term catch options to fisheries managers. The fisheries managers debate the scientific advice amongst other considerations, and TAC’s and other measures are agreed and written into legislation. National bodies enforce the legislation. The life cycle of the TAC is given in Fig. 2.

Figure 1. Fisheries management interactions in the EU.

3. Background
The basis of the science used to assess fish stocks has remained unchanged since the late 19th Century. Catches are still related to populations according to Baranov’s equation. Since this, the development of basic population models has been to make them more complex, through ‘virtual population analyses’ (VPA) and stock-recruitment functions, to statistical catch at age models (where the model is fit to observations with assumptions reducing the number of parameters). The complexity pertains not just to age structured models but also through the development of production or biomass dynamic models and mark recapture techniques.
(such as Jolly-Seber). Modelling this complexity, all in a deterministic state, gives us the current 'state of the art' in fisheries science (as used routinely in the EU). A good analogy would be a modern highly developed internal combustion engine, state of the art- yes, but still based on technology over a hundred years old.

The current system of fisheries management has grown up around such models and the TAC system is dependent on such a view of the fisheries 'world'. These models offer the impression that biological production can be enumerated precisely. Yet even back in the 1970’s the voices of dissent were beginning to be heard. People realised that the assumption of equilibrium, which was fundamental to concepts such as maximum sustainable yield (MSY), was poorly based. Worse still that without such equilibrium, what was being 'sold' as MSY was unattainable, and the consequences of trying to achieve it would likely lead to overexploitation. Nonetheless the fisheries assessment and management system still (to a great extent) views the world from a deterministic viewpoint. And the requirement to enumerate productivity keeps an army of scientists in gainful employment, and a cohort of fisheries managers comfortable in the illusion that they are only exploiting a 'sustainable' proportion of what they really 'have'.

To view a biological system from a deterministic perspective, to add great complexity to a system that ignores uncertainty, and to blithely rely on un-testable assumptions, characterises much of the mistakes that have been made in fisheries assessment and advice in the past. More recently Fisheries scientists have struggled to 'explain' the retrospective inconsistencies in their assessments, while much of this comes from poor fisheries dependent data coupled with inappropriate assumptions, some also comes from unaccounted uncertainty in the system.

4. How can we make things better?

Whatever its shortcomings, the current system has at least not prevented overexploitation. It would be easy to just blame the fisheries managers and their failure to tackle serious issues such as overcapacity. However science is not innocent in the situation in which we now find ourselves. The current annual cycle of setting TAC’s perpetuates the belief that science is able to measure and predict the state of stocks annually. It is the belief that we know what the stocks can sustain that leads to a ‘gold-rush’ mentality to exploit the resource at maximum level. The truth is we don’t know with enough certainty what the maximum sustainable yields of fish stocks are, and even if we did, we couldn’t measure exploitation with enough precision to avoid exceeding MSY. We need to move away from the deterministic view of the ‘world’, to accept we cannot exploit at MSY without a high risk of overexploitation. This could be done by taking a longer-term view of exploitation. The move to long-term advice brings with it the tacit admission that we cannot precisely enumerate biological productivity, and thus maximum harvest rates annually. The move to long-term advice should be based on stochastic simulations of stock productivity. From these we can develop and test Harvest Control Rules (HCRs). The introduction of such an approach should engender stability in the fishing industry, which is good for fishermen and ultimately good for fish stocks. However stock levels must be
rebuilt above thresholds defined in the HCR’s to effect their introduction. In the short term this will necessitate some fishing restrictions, and this will be difficult for managers to implement.

5. Two schools of thought

The move to long-term advice is one element of the new paradigm. The second is the appreciation that marine organisms do not live in isolation. There are differing views on how to incorporate external ecosystem of environmental factors into single species advice. One school of thought seeks to incorporate each factor as a new parameter, building ever more complex models (‘splitters’). The other school seeks not to model those factors that cannot be distinguished from background ‘noise’ and are without hypotheses, and incorporates their effects as extra ‘noise’ (‘lumpers’). Either approach is incorporated by the framework composed by the ICES WGMG in 2003 (Fig. 3).

![Figure 3. The ICES simulation framework for evaluating management strategies.](image)

6. Where do we go from here?

The EU has espoused a desire to embrace long-term advice in the review of the CFP, and this is now incorporated in the memorandum of understanding with ICES. ICES has confounded itself with a lack of direction between this and the incorporation of ecosystem effects, and multi-species advice. However there is nothing like a crisis to create an opportunity. At this point there is a need for academia to join the debate, National fisheries institutes need to work more closely with both Maths and Biological departments to broaden perspectives. There is inertia in the system as it currently stands (Fig.1), and given this it is probably best changed from without.
Paper 2, 2004: The Irish Sea cod recovery plan: some lessons learned

Edward Codling¹,* & Ciaran Kelly¹

¹Fisheries Science Services (FSS), Marine Institute, Galway, Ireland.

*Presented as a talk by Edward Codling at FSS mini-symposium 2004. Email: edd.codling@marine.ie

Note – this paper has subsequently been substantially rewritten and submitted to ICES Journal of Marine Science.

1. Introduction

Cod (Gadus morhua) in the Irish Sea has been actively fished and exploited for hundreds of years, and is considered a ‘unit stock’. The Irish Sea is towards the southern limit of the distribution of cod in the North Atlantic. The warmer waters mean that Irish Sea cod is rapid growing, fully maturing by the age of 3. Spawning occurs in aggregations and the stock is routinely targeted at this time, suffering high mortality due to fishing.

Even though the direct fishery for cod is in trouble because of low stock sizes, the value of the catch in 2002 was about €9 million, and the stock is important as a target and a bycatch in several fisheries. The stock is managed by the EU under the common fisheries policy (CFP), through annually determined total allowable catches (TAC) with minimum landing size and mesh size restrictions. Irish Sea cod is caught by four EU Nations (Ireland, UK, France, Belgium), comprising 32 fleets. The main catches by gear are: mid-water trawls – 64%, Nephrops trawls – 22% (bycatch), beam trawls – 7%, other gears – 7%.

Figure 1. Irish Sea cod stock history from 1968 – 2002. The stock is currently at historically low levels and is considered to be ‘overexploited’ and in a state of ‘recruitment impairment’.
Since 1989, the Irish Sea cod stock has been in a state of continual decline (Fig.1) and is currently well below precautionary limits ($B_{lim} = 6000$ tonnes), and is considered to be in a state of 'recruitment impairment' (Fig. 2).

Figure 2. Plot of numbers of recruits against spawning stock biomass (SSB) for Irish Sea cod between 1968 and 2003. Assessment data is compared to simulated values that have been generated using the simulation algorithm described in Section 4. Note that for large values of SSB, the historical data shows that recruitment can be high or low, but at small levels of SSB (most recent years), the recruitment is always low.

2. Irish Sea cod recovery plan

Closed areas were introduced in 2000 in an attempt to recover stocks to $B_{pa} = 10,000$ tonnes. The closed areas were based on the putative spawning grounds at peak spawning time (14 February to 30 April), see Fig. 3. The closed areas applied to all fishing activities, excepting derogations for Nephrops and beam trawlers (29% of total fleet), which were permitted to fish in defined ‘boxes’. There were no effort restrictions outside the closed areas so it was likely that there was displacement of fishing activities to the areas outside the closed area. No compensation was given to fishing fleets so there was no incentive to ‘tie up’.
3. Stock projection simulation

Simulations of the stock dynamics of Irish Sea cod under different harvesting strategies (different levels of F) have been completed using an in-house simulation tool (F-PRESS).

3.1 Simulation structure and data requirements

The F-PRESS simulation uses simple population dynamics:

- Single species;
- Age structured;
- Exponential decay used for mortality;
- Total mortality consists of fishing mortality, F, and natural mortality, M;
- Discrete time process (stepwise in years);
- Simple recruitment function;
- Stock is projected forwards from a set date for a given number of years.

Standard ICES format data is used in the simulation:

- Population numbers at age;
- Weight at age;
- Maturity at age;
- Fishing mortality at age, F;
- Natural mortality at age, M;
- Proportion of F and M before spawning (if applicable);
- CV values for each of the above (a measure of variability).
3.2 Stochasticity

In any forward projection of a fishery, the high levels of uncertainty in the system must be accounted for. This uncertainty can be due to natural or environmental variability (e.g. fluctuations in recruitment), operational variability (e.g. changes in effort levels, catchability, etc), or observational errors.

Deterministic modellers usually either i) ignore this variability; or ii) attempt to account for it with increasingly complex models. However, we explicitly include this inherent variability by including a stochastic (random) model. Random noise is added to the initial population numbers (to account for observational errors in 1st year only), to the stock characteristics (e.g. weight at age, maturity at age, natural mortality at age to account for natural variability), and to the fishery characteristics (fishing mortality at age every year to account for operational variability). This added random noise consists of ‘Gaussian white noise’ – this may sound complicated, but is actually very simple and consists of randomly redrawing parameter values from a Normal distribution with specified CV value (any negative values are rejected and redrawn).

Recruitment is typically highly variable and can be modelled using various stochastic methods (Bootstrap, Ricker map, Segmented regression etc). The simplest realistic model is to use a Ricker map with added ‘white noise’ – this is far more realistic than a simple deterministic fit, see Figure 2.

3.3 Using the simulation

The simulation can be used to i) make a ‘hindcast’ (e.g. project forward from 1999-2003 and then compare to real data); or ii) make a forecast (e.g. project forward from 2003-2007). Due to the (realistic) random variability, the results of a single projection only represent ‘typical’ behaviour. If a number of projections (e.g. 100 or 1000) are run then general trends and confidence limits can be observed.

4. Hindcast projections

4.1 What if’ scenarios

The following hindcast projections are completed using the simulation with a stochastic Ricker recruit function. The hindcasts are completed from 1999-2003, and two different ‘F multiplier’ (F_mult) scalars are applied from 2000-2003 to represent two different scenarios (no action or a closed area). The projections completed with 100 iterations each. Results are given in Figs. 4 and 5.
Figure 4. Time series plot for projections using $F_{mult} = 1$ (representing no management action). The plot shows that with fishing mortality unchanged, we can expect SSB and catch to continue to decline.

Figure 5. Time series plot for projections using $F_{mult} = 0.36$ (representing a closed area where all mid-water trawls are removed from the fishery). The plot shows that with fishing mortality significantly reduced, we can expect SSB and catch to recover.
The levels of catch after 5 years are similar even though there is 64% less fishing effort in the second scenario (catches are similar because more fish are in the population):

- $F_{\text{mult}} = 1$, sum of 5 yrs mean catch weights = 16,253 t,
- $F_{\text{mult}} = 0.36$, sum of 5 yrs mean catch weights = 15,864 t

The total landings over the 5 year projection period are almost unchanged, but SSB is in a much healthier state with $F_{\text{mult}} = 0.36$.

4.2 What actually happened?
Using estimated landings, average fishing mortality over 2000-2003 is on average at 85% of 1999 levels, i.e. $F_{\text{mult}} = 0.85$, although there are some problems with recent under-reporting of landings and recent fishing mortality may actually be higher.

As shown in Fig. 6, although there is some doubt over the data reliability in the results of NSWG 2003 / 2004, our stochastic projection (using data available to NSWG 2003) could be a useful tool to help with stock assessment.
4.3 More ‘what if’ scenarios

Further simulations were completed to investigate the effect of changing the F multiplier on the stock. Hindcasts were completed from 1999-2003 with an F multiplier applied from 2000-2003. Each projection completed for a value of \( F_{\text{mult}} \) from 0 to 1.2, with 1000 iterations for each F multiplier.

In the first scenario (Fig. 7), observed recruitment values were used in the simulation, although this may not be realistic as recruitment may be SSB dependent and simulation SSB values may vary from actual values.

![Graph](image)

**Figure 7.** Plot of final SSB level in 2003 against \( F_{\text{mult}} \) used in the projection. It is clear that current fishing mortality is too high and a very low level is required to recover the stock above \( B_{pa} = 10,000 \) tonnes. There is a low level of variability in results as all projections use the same recruitment data for each year.

Assuming the recruit values are known:
- \( F_{\text{mult}} = 0.225 \) gives a 95% probability of the stock reaching \( B_{pa} \) after 5 years (1999-2003).
- \( F_{\text{mult}} = 0.85 \) (actual average value) gives a less than 5% chance of reaching \( B_{pa} \) after 5 years.

In the second scenario (Fig. 8), we assume that we don’t know the recruit values. The same simulations were completed but using random Ricker recruitment values for each year rather than the observed data. This will obviously introduce more uncertainty into results, but this is realistic, as in reality managers don’t know recruit values 5 years ahead!
These results demonstrate that it is not possible to give a 95% probability of reaching $B_{pa}$ after 5 years even with zero fishing! Similar simulations could be used by managers to assess levels of medium term risk under different recovery plan scenarios (different $F$ multipliers).

5. Future management strategies

Forecast projection results (2002 – 2006) show similar results to the hindcasts presented above. The main conclusion of the simulations is that stock recovery will only happen with a drastic cut in fishing mortality - half hearted efforts are as ineffective as doing nothing. The total landings over a 5-year period need not be substantially reduced with reduced fishing mortality – larger stocks will produce higher catches for similar levels of effort.

It is clear that the recovery plan has not produced the required stock recovery. In our opinion, contributory problems for this failure include:

- No effort control outside closed areas;
- Derogation syndrome (closed areas are not truly ‘closed’);
- ‘I’m only a bycatch’ syndrome (closing the directed fishery may not be enough if there is large bycatch of cod from other fisheries);
- Being frank about what we can measure (landings, effort data etc) – we can’t judge if the recovery plan is working if there is high uncertainty in SSB estimates;
− Industry misconceptions – the industry should be consulted when designing the recovery plan and the high levels of uncertainty should be made clear.

In our opinion, the following measures are required for there to be any chance of the Irish Sea cod stock recovering:

− Fishing mortality to be drastically cut:
  o Overall effort in Irish Sea needs to be cut – closed areas are ineffective as effort is increased outside;
  o Derogation fleets (Nephrops) to use TCMs to reduce bycatch;
  o If greater numbers are allowed to reach large sizes (5 yrs +) the SSB will increase much more rapidly.

− Long term management plan needed rather than short term ‘reactionism’;

− Precautionary approach is required – reverse burden of proof, i.e. if the stock is not clearly in a healthy state then no fishing should be allowed;

− Accept that there are irreducible system uncertainties and manage with this in mind rather than ignoring these;

− Plan and scientific advice need to be fully implemented and acted on – no half hearted efforts or political ‘watering down’;

− Industry collaboration:
  o Plan designed with industry and benefits made clear;
  o Compensation payments for non-fishing / decommissioning.
1. Introduction

Many commercial species (e.g. cod, haddock) progress from the hatched egg through a larval stage before evolving into juvenile and eventually adult fish. For an individual larva, growth and recruitment into the adult population is fraught with peril. Even under favourable conditions only a tiny minority of larvae survive to adulthood (Chambers and Trippel, 1997). There are physiological limits on how fast an individual can grow, imposed by factors such as gut size and metabolism (leading to capped-rate models.). When growth models are deterministic an ordinary differential equation (ODE) can be derived (Cushing and Horwood, 1994). However, larvae exist in a highly stochastic and patchy environment and possess only limited locomotory and sensory ability. There is a need to extend these ODE models to incorporate stochasticity. Incorporating stochasticity into capped-rate models is not trivial and intuitively plausible approaches based on stochastic differential equations can lead to misleading result (see James et al., 2005 for a full discussion). We propose simple models based on queuing theory.

2. Queuing models

We assume the larva takes $T$ time units to digest an item of zooplankton, and can accommodate a maximum of $N$ items of zooplankton in its stomach. The Larva grows at its physiological maximum rate whenever there is zooplankton in its stomach, and grows at a rate zero otherwise. For simplicity, we ignore the effects of predation, and assume zero metabolic costs. We seek to estimate the time taken for a larva to reach a critical mass.

2.1 Poisson (random) zooplankton encounters

If prey items are encountered as a Poisson process with mean rate $\lambda$ per unit time, $X$, the total number of prey items encountered in $U$ units of time, follows a Poisson distribution with probability function:

$$f(x) = \frac{\exp(-\lambda U)(\lambda U)^x}{x!}, \quad (x = 0, 1, \ldots). \quad (1)$$

It follows that the time between encounters, $T$, follows an exponential distribution. The expectation and variance of $T$ are given by:
\[ E(T) = \frac{1}{\lambda}, \quad \text{Var}(T) = \frac{1}{\lambda^2}. \]  

(2)

2.2 Pareto (patchy) zooplankton encounters

Zooplankton populations are known to exhibit spatial patchiness at length scales ranging from tens of kilometres to less than 10 metres (Tokarev et al., 1998). The Poisson process model implicitly assumes that zooplankton are homogeneously distributed across the ocean. By generalising to a Pareto encounters process we can incorporate patchiness into the model. The Pareto process has parameters \( \lambda \) (mean zooplankton encounter rate) and \( r \) (variability in zooplankton encounter rate).

The Poisson process model can be extended by allowing \( \lambda \) to be a random variable, whilst assuming that \( X|\lambda \) is Poisson (as in Eqn. 1). A plausible distribution for \( \lambda \) is the gamma distribution with parameters \( r, \alpha > 0 \). It can be shown that \( X \) then follows a negative binomial distribution with probability function:

\[ f(x) = \binom{r + x - 1}{x} \left( \frac{\alpha}{\alpha + U} \right)^r \left( \frac{U}{\alpha + U} \right)^x, \]  

(3)

for \( x = 0, 1, \ldots \) It follows that the time between encounters, \( T \), has cumulative distribution function:

\[ F(t) = 1 - \left( \frac{\alpha}{\alpha + t} \right)^r, \quad (t > 0), \]  

(4)

which defines a Pareto distribution. Setting \( \alpha = (r - 1)/\lambda \), it may be shown that:

\[ E(T) = \frac{1}{\lambda}, \quad r > 1, \]  

\[ \text{Var}(T) = \frac{r}{\lambda^2(r - 2)}, \quad r > 2. \]  

(6)

From Eqns. 2 and 5, \( E(T) \) is now the same as for the ordinary Poisson process model. As \( r \downarrow 2 \), \( \text{Var}(T) \rightarrow \infty \), giving rise to a patchy encounters process. As \( r \rightarrow \infty \), \( \text{Var}(T) \downarrow 1/\lambda^2 \) (i.e. the same as for the Poisson model).

3. Simulations

Unfortunately, the probability density function (PDF) of time to critical mass is not straightforward to obtain explicitly for either the Poisson or Pareto encounters process. With Poisson encounters, the model is described by an \( M / D / 1 / N \) queue (see Brun and Garcia, 2001 for a discussion). For Pareto encounters, the model is a queue \( G / D / 1 / N \) (for which few results are available in the literature). To develop an understanding of the model we run a series of simulations. We set \( \lambda = 0 \) and define time to critical mass to be the time taken to digest 1000 items of zooplankton. PDFs are based on simulation of 1000 larvae.
In Fig. 1 we consider the effect of stomach size by setting \(N = 2\) (solid curve), \(N = 8\) (dashed curve), \(N = 32\) (dotted curve) and \(N = 128\) (dotdash curve). We consider digestion times \(T = 0.5\) (i.e. \(T < 1/\lambda\)), \(T = 2\) (i.e. \(T \approx 1/\lambda\)), and \(T = 8\) (i.e. \(T > 1/\lambda\)). We compare the Pareto encounters process with \(r = 2.1\) (shown in Figs. 1(b), (d) and (f) on the right) with the Poisson encounters process (shown in Figs. 1(a), (c) and (e) on the left).

**Figure 1:** Comparing M / D / 1 / N (left figures) with G / D / 1 / N models (right figures): (a) (b) physiologically-limited regime \((T > 1/\lambda)\); (c) (d) food-limited regime \((T < 1/\lambda)\); (e) (f) balanced regime \((T \approx 1/\lambda)\).
In a physiologically-limited regime, time to critical mass is dominated by the time to digest an item of zooplankton, $T$. Contrasting Figs. 1(a) and (b), for a Poisson encounters process, stomach size has no influence on time to critical mass. For a Pareto encounters process a small stomach size greatly increases time to critical mass.

In a food-limited regime, time to critical mass is dominated by mean zooplankton encounter rate, $\lambda$. Contrasting Figs. 1(c) and (d), patchiness increases the variability in the time to critical mass.

In a balanced regime, time to critical mass is constrained below by the time to digest an item of zooplankton, $T$. Having a large stomach size is a distinct advantage. Contrasting Figs. 1(e) and (f), patchiness increases the variability in the time to critical mass and the average time to critical mass. Having a small stomach is a greater disadvantage in a patchy environment.

4. Conclusions and further work

We are developing a simple yet flexible model of larval growth that encapsulates the random nature of the process, and yet respects the physiological limits on growth. The model generalises in a straightforward manner to allow for the patchiness of zooplankton. Early results show that, except in a food-limited regime, patchiness increases time to critical mass. Small stomach sizes are generally a disadvantage (and more so in a patchy environment). For a more extensive discussion of results, see James et al. (2005).

We seek to incorporate metabolic costs and predation into the model (perhaps using the framework of the Cushing and Horwood, 1994). We are particularly interested in using larval stomach contents data and otolith growth data to set the parameters of the model realistically. We intend to search for an analytical solution that removes the need for simulation (perhaps using the approach of Truscott and Gilligan, 2003).

References
1. Introduction

Current fisheries science literature often criticises single species stock assessment methods, arguing that more complex biological processes should be taken into account. The present state of fisheries worldwide, with 17% of the stocks being over-exploited (Anonymous, 2005), and dramatic collapses such as that of the cod stock off the east coast of Newfoundland, is certainly an incentive for a shift in the scientific approach of assessing a stock.

Although the picture worldwide is gloomy, some stocks are presently exploited within safe biological limits e.g. North Sea herring (ICES, 2005). Therefore this suggests that current methods of assessment capture well enough the dynamic of these stocks, for management purposes. The VPA based methods used to study stocks in European waters, essentially estimate the rates of fishing mortality assuming that the number of surviving fishes from a cohort decreases with time according to a negative exponential function of the mortality rates. This fundamental population dynamic process has a statistical description known as the exponential probability distribution function (Cowan, 1998). It relates the probability of surviving up to a certain age with fishing mortality rates: at a low fishing rate the average age is high because individuals have a better chance of reaching an old age, while at high rates of mortality the chance to reach an old age is reduced as is the mean age of the cohort. Therefore fisheries management could be summarized as the monitoring of the older age group of a fish population and adjusting of the exploitation rate in order to maintain them in sufficient proportion in the population.

This hypothesis was tested using a computer simulation in order to investigate if a harvest control rule based on the monitoring of the oldest age classes of a population has the capacity to determine a sustainable level of exploitation of a wild population. The simulation was designed as a qualitative investigation of the behaviour of a fishery, when managed using Total Allowable Catches (TAC).

2. Methods

2.1 Simulated population dynamics

The simulated population has been structured into 10 age groups. Yearly time steps were used to simulate the dynamic of the fishery. The different biological and fishing processes, which are detailed below, were
applied at each time step following this sequence: natural mortality, reproduction and fishing mortality.

Natural mortality
The survival of individuals between 2 successive age groups depends on natural and fishing mortality. Natural mortality is assumed to claim every year the life of a constant fraction of individuals at age. This fraction is fixed to 18% which in terms of yearly rate of natural mortality is equal to $M = 0.2$.

Reproduction
The abundance of individuals in the first age group ($N_1$) is determined at every time step by a density dependent function of the fecundity at age ($F_a$)

$$N_1 = \sum_{a=10}^{a=10} \frac{F_a \times N_a}{a=10} \left( \sum_{a=1}^{10} N_a^{0.25} \right)$$

where $N_a$ is the abundance at age $a$. Only fishes from age groups 5 and above are assumed to contribute to the new generation with a fecundity that increases with age (Table 1). In Eq. 1, the density dependence to the total number of fishes in the population, raised at the power 0.25, was set empirically in order for reproduction to compensate mortality at high population abundance level assumed to represent the carrying capacity of the environment.

<table>
<thead>
<tr>
<th>Age group</th>
<th>1</th>
<th>2</th>
<th>3</th>
<th>4</th>
<th>5</th>
<th>6</th>
<th>7</th>
<th>8</th>
<th>9</th>
<th>10</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fecundity</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>4</td>
<td>8</td>
<td>12</td>
<td>16</td>
<td>20</td>
<td>24</td>
</tr>
</tbody>
</table>

*Table 1: Fecundity at age.*

Somatic growth
The weight at age was assumed to be fixed. They were calculated using the equation of Von Bertalanffy

$$W_a = 0.01 \times [L_\infty \times (1 - \exp[-k \times (a - t_0)])]^{3}$$

where

- $W_a$ is the weight at age $a$.
- $L_\infty$ is the asymptotic length ($L_\infty = 10$).
- $k$ is the parameter of growth velocity ($k = 0.5 \text{ years}^{-1}$)

---


*The fraction ($f$) of individuals dying from natural causes is related to their yearly rate of mortality ($M$) by $f = 1 - \exp(-M)$ (Hillborn and Walters, 1992).*
$t_0$ is an adjustment parameter ($t_0=0$ years)

**Fishing mortality**

The amount of fishes killed by the fishing fleet is determined by the total allowable catch (TAC). Harvest is assumed to be age-selective with the first 2 age groups never being caught by the fishing gear while the last 5 are always (Table 2). The distribution of the catch at age is calculated by taking selectively a fixed fraction (1%) of the abundance of the number at age in the population. The sum of the weight of this fraction is compared to the TAC:

- if it equals or exceed the TAC, the age distribution of the catch is determined by this percentage of the population
- otherwise the process is repeated until a particular fraction of the population is found to have a total weight equal or superior to the TAC

Such an algorithm allows one to determine an age distribution for the TAC that is removed from the population at every time step.

<table>
<thead>
<tr>
<th>Age group</th>
<th>1</th>
<th>2</th>
<th>3</th>
<th>4</th>
<th>5</th>
<th>6</th>
<th>7</th>
<th>8</th>
<th>9</th>
<th>10</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fecundity</td>
<td>0</td>
<td>0</td>
<td>0.2</td>
<td>0.5</td>
<td>0.8</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
</tr>
</tbody>
</table>

*Table 2: Selectivity of the fishing gear.*

### 2.2 Harvest control rule

The rule used to modify the TAC assumes that a sample (1%) of the population is available to determine its age distribution. If the number of fishes of an age $a$ is below the threshold value of 25 individuals, the risk level for the stock is declared to have the value of $n = 10 - a$. The harvest control rule states that if the risk level is null, the TAC for the following time step can be increased by 10%, while if the risk level is of value $n$ then the TAC is decreased by $n \times 10\%$.

The total abundance of fishes requires some iterations to reach a plateau. The fishery exploitation is simulated to start only after this “carrying capacity” value is reached *i.e.* at the thousandth iteration.

### 3 Results

#### 3.1 Beginning with a cautious exploitation

In this particular run, the harvest control rule is initiated with a very small TAC (TAC=200). Once the exploitation starts, at the thousandth iteration, the population abundance drops at about a 1/4 of its un-exploited level (Fig. 1). The landings start by rising to a maximum level around $1e+05$ (arbitrary units). Following this, the TAC diminishes consistently in a dumped oscillation fashion to converge to a stable exploitation level of about $1.5e+04$. The harvest control rules allows for a sustainable exploitation of the simulated fish population.
3.2 Beginning with over-exploitation

In this run, the harvest control rule is initiated with a very large TAC (TAC=1e+08). The starting TAC is so large that it collapses the population (Fig. 2). The harvest control rules delays any further catches to until the stock is rebuilt. When the exploitation resumes, starting this time from a very small TAC as in the previous case, the exploitation of the population converge towards a stable state with final population abundance oscillating around a quarter of their pre-exploitation level and landings in weight being in average 1.5e+04 arbitrary units.

Figure 1: Variation of the abundance of individuals (grey line - left hand side axis) and landings in weight (black line - right hand side scale in arbitrary units) from a population exploited at first with a very small TAC which is subsequently modified using the harvest control rule.
3.3 Comparison of the two strategies

Fig. 3 superimposes the trajectory of population abundance under these 2 strategies of exploitation. Both strategies end up at the same abundance level but a cautious exploitation of the population allows one to quickly reach a stable level of exploitation.
4 Conclusion

The results from this simulation of a fishery exploitation using TAC reveals that monitoring the age structure of the population, as an indication of the survival of the individuals, is sufficient to achieve sustainable exploitation and to avoid a collapse of the stock. The level of exploited achieved by the harvest control rule described here are probably far from optimal. The purpose of this simulation was not to optimise the harvest but determine whether a robust method of management exists.

The current methods used for stock assessment are much more complex than the simple simulation presented here because they investigate many more different aspects of “real world” populations. But in essence, they estimate mortality of the individuals and alter the TAC in consequence. It is important to realise that mortality or probability to reach a particular age is a relative concept independent of the absolute number of fishes or total
landings. In matter of fact, it is very difficult to obtain absolute estimate of abundance of marine animals. Therefore it is essential to develop management methods that don’t rely on such measurements.

The results of these simulations shows that starting the exploitation of a stock with a low TAC level is a strategy that allows for a more efficient harvest: it avoids the collapse of the stock and allows one to reach a stable rate of exploitation much faster. The critical problem with over-exploitation is that once the stock is collapsed, it takes many years to rebuild itself (if it does at all). Therefore management decisions that increase fishing mortality or produce a disappearance of the older age classes should be taken with great caution.

References
Paper 5, 2004: Sound: the essential navigation cue for young reef fishes to find their way home

Stephen D. Simpson¹,*

¹University of Edinburgh, UK

*Adapted by Edward Codling from a talk presented by Stephen Simpson at FSS mini-symposium 2004. Email: s.simpson@ed.ac.uk

1. Introduction

Coral reef fish are quite different to fish typically exploited in temperate oceans. However, when reef fish stocks were initially exploited and managed as fisheries, traditional temperate models and management methods were used, needless to say, without success. The difference between temperate and reef fisheries has much to do with the particular bipartite life-history strategies of most reef fish (which consist of a pelagic larval stage and a localised adult stage on the reef). The pelagic early life history is not well understood (the ‘black box’) and we must increase our knowledge of this important stage if reef fish stocks are to be successfully managed. In this paper I will discuss the life-history of typical reef fish and in particular, their sensitivity to and dependence on underwater sound.

2. The life history of a coral reef fish

Figure 1. The bipartite life history of a typical coral reef fish. The adult fish spends its life close to the reef but larvae develop in pelagic areas away
from the reef before returning to a suitable reef environment. This pelagic stage is not well understood and has been described as the ‘black box’ in our knowledge of these fish.

2.1 Riding on the High Seas…

Early studies and models of the pelagic stage assumed that reef fish larvae were simply passive and were transported by currents. However, local self-recruitment (larvae returning to the reef where they were spawned and originally left) was observed which could not be explained. It was suggested that highly complex eddy and current structures could result in this self-recruitment effect but the more plausible explanation was that the fish larvae had quite well-developed locomotory abilities. Recent experiments (both in labs and in situ) have shown that reef fish larvae are very strong swimmers. For example, temperate fish (Clupeiformes & Gadiformes) are weak swimmers when in the larval stage and can swim at 1-2 cms⁻¹, which is equivalent to 1-5 BL s⁻¹. However, typical coral reef fish larvae have been observed to cruise at 14 BL s⁻¹ and the record observed speed is 34 BL s⁻¹. Reef fish larvae can sustain these speeds over several days – typically cruise speeds have been observed for 3½ days but the record is 12 days sustained swimming. This means that a typical coral reef fish larva can swim an average of 40km in only a few days, with a record of 140km.

2.2 You can swim, but you still need to navigate…

Experimental observations on swimming ability show that coral reef fish larvae are well-developed swimmers. However, this does not explain how they are able to return to coral reefs after days or weeks away in the open sea in their pelagic stage. Several sensory cues have been suggested as possible mechanisms for larval orientation (smell, sound, visual etc). At the larval stage most fish have only limited sensory capabilities. A hierarchy of senses (increasing downwards) for coral reef fish larvae is given below:

- Visual cues (limited);
- Chemosensory cues:
  - coral types;
  - suitable habitats;
  - conspecifics (also sound);
- Tidal fronts:
  - temperature / salinity;
  - chemical heterogeneity;
- Sound.

2.3 Why investigate sound?

Recent modelling work has demonstrated that even weak swimming larvae can increase their chances of survival if they can adapt their movement to take into account current-independent cues. The most likely current-independent cue is sound and those species that are able to respond to reef noise and adapt their behaviour at the larval stage would be at an evolutionary advantage.

Coral reefs are extremely noisy places – aside from the noise of wind and waves, snapping shrimps, nocturnal zooplanktonivorous fish, and fish choruses (on daily and seasonal cycles) all contribute to reef noise. In fact,
there is recent evidence that peaks in noise levels on the reef correspond to periods of reef fish larvae settlement, suggesting a direct link between reef sound and recruitment.

Temperate regions are noisy too - non-biological noise (wave action, wind & rain noise) is of the order of 100 DB (< 25 KHz), while invertebrates such as spiny lobsters (*Palinuridae*) and *Diadema* and many species of fish all make a lot of underwater noise.

### 2.4 How fish hear

Fish respond to sound through the octavolateralis (otolith and lateral line) system and through their swim bladders, both of which are sensitive to sound vibrations in the water. The swim bladder is also used to create vibrations (‘sounds’) in the water and many fish can be classed as specialists: grunts, drums, croakers, etc.

### 3. The importance of sound for larval coral reef fishes

#### 3.1 Are larval coral reef fishes attracted by the noises of reefs at settlement?

Experimental studies were completed in the field where pre-recorded reef noise (Fig. 2) was played on a continuous loop using a UW-30 speaker. Noise was emitted at 104 dB (re 1 µPa at 1 m) and two light traps were placed to catch settling reef fish, one with sound, one without (Light pool = 50 m radius, Sound pool = 130 m radius), see Fig. 3.

Figure 2. Frequency pattern of pre-recorded noise played in light traps in field experiments. The pattern is typical of the noise emanating from a coral reef and peaks can be seen corresponding to a fish chorus and shrimp snapping.
Figure 3. Experimental set up in the field to test settling reef fish sensitivity to sound. Two identical light traps were set up where only one had an underwater speaker playing pre-recorded reef noises.

37 paired samples were made (16 Nov, 11 Dec, 10 Jan) and 94 species (21 families) of fish were caught. In total, 40,225 fish were caught: 89% were Pomacentridae, 6% Apogonids, and 5% were from the other 19 families, see Fig. 4. It is clear that the settling reef fish were attracted more to the light trap that was also playing reef noise.

Figure 4. Bar chart showing percentage of fish caught in each light trap for the main families of fish caught. The black bar corresponds to the light trap with sound, the white bar to the light trap without sound. Clearly, for all families, the greatest catch was from the light trap with sound suggesting that settling reef fish are attracted to reef sound.
3.2 Can larval coral reef fishes use this attraction to navigate to reefs where they then settle?

A similar field experiment was set up to test whether reef fish are more likely to settle on a noisy ‘patch reef’ (a small artificial reef), see Fig. 5.

Figure 5. Experimental set up in the field to test whether settling reef fish are more likely to settle on a noisy reef. Two identical ‘patch reefs’ were set up where only one had an underwater speaker playing pre-recorded reef noises.

57 species of fish (17 families) were caught from (and were assumed to have settled on) the ‘patch reefs’ 17 families. In total, 1,113 fish and 868 recruits were caught, of which 80% were *apogonids* and 15% *pomacentrids*. As with the earlier experiment, it is clear from Fig. 6 that in the majority of species, the reef fish were more likely to settle on the patch reef that had the associated reef noise playing.

Figure 6. Bar chart showing percentage of fish caught from each patch reef for the main families of fish caught. The black bar corresponds to the patch...
reef with sound, the white bar to the patch reef without sound. Clearly, for all fish in the majority of families (and particularly for juveniles of all families), the greatest catch was from the patch reef with sound.

3.3 Do fishes work alone or in groups to navigate to reefs?
Experiments have been completed in the lab using choice chambers. No result was found when fish were tested individually, but fish in groups generally moved towards sound, which suggests some form of ‘voting’ or ‘consortium’ behaviour? However, the exact mechanics of group interactions is very difficult to study experimentally.

3.4 How far from reefs can the larvae cue in from?
Experimental field studies suggest that reef fish larvae can respond to reef noises at distances of 1km (or possibly further), although different species may respond at different distances. Experimental work is ongoing.

3.5 Do larval reef fishes have a memory for sounds?
Laboratory experiments were completed to test for directional response & conditioning. The results are shown in Fig. 7.

<table>
<thead>
<tr>
<th>Treatment</th>
<th>During conditioning</th>
<th>During testing</th>
<th>Predictions if conditioning of cues on larvae is occurring</th>
<th>Actual findings</th>
<th>Hypothesised explanation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Natural</td>
<td>a</td>
<td>a,A</td>
<td>Response seen</td>
<td>Positive response (p&lt;0.05)</td>
<td>In innate attraction</td>
</tr>
<tr>
<td>Cue (A)</td>
<td>a</td>
<td>b,A</td>
<td>No response</td>
<td>Positive response (p&lt;0.05)</td>
<td>Innate attraction</td>
</tr>
<tr>
<td>Artificial</td>
<td>b</td>
<td>a,B</td>
<td>No response</td>
<td>Negative response (p&lt;0.1)</td>
<td>Innate aversion</td>
</tr>
<tr>
<td>Cue (B)</td>
<td>b</td>
<td>b,B</td>
<td>Response seen</td>
<td>Positive response (p&lt;0.05)</td>
<td>Conditioned response</td>
</tr>
<tr>
<td>No (C)</td>
<td>c</td>
<td>a,C</td>
<td>No response</td>
<td>No response</td>
<td>Control</td>
</tr>
<tr>
<td>Treatment</td>
<td>c</td>
<td>c,C</td>
<td>No response</td>
<td>No response</td>
<td>Control</td>
</tr>
</tbody>
</table>

Figure 7. Table of conditioning and response of reef fishes to sound in lab experiments. The results suggest that larval reef fish may have a ‘memory’ and can become conditioned for particular sounds.

3.6 Could the attraction be an ‘imprinted’ behaviour?
Embryonic studies completed at the University of Kentucky on *Amphiprion ephippium* (saddle anemonefish) show that reef fish embryos can respond to noise they are subjected to, see Fig. 8. This suggests that the development of ‘hearing’ may start even before the fish is hatched. This could potentially allow for ‘imprinting’ of particular noises (i.e. reef sounds), which the hatched larval fish may then respond to.
Figure 8. Plot showing the responses of embryonic reef fish to different sounds. The lines and associated numbers correspond to the days in the embryonic stage (i.e. the line labelled 9 refers to the responses of a 9 day old embryo). Clearly, as the embryo ages it responds to a wider range of sounds at different sound levels. The frequency ranges for particular reef noises are marked showing that these all lie within the range that very young embryonic reef fish can respond to.

4. The road ahead...

A number of further investigations are planned to study reef fish and sound in the field, in labs, and modelling work.

Field work:
- Compare with temperate water fishes;
- Use recordings of different reef habitats;
- Identify sources of sound cues - key species?
- Study hearing in larval fish across families;
- Larger scale artificial reef study.

Lab work:
- Attempt to imprint fish with artificial sounds;
- Look for 'windows of opportunity' in learning;
- Study environmental effects on learning;
- Study parental / nutritional effects on learning.

Modelling work:
- Look at behaviour of groups vs. individuals;
− Predict the evolutionary forces acting on sensory development;
− Simulate human perturbations in ‘real’ arenas;
− Build behaviour-based dispersal models to improve fisheries management.

There are a number of important applications of the results gained from this study into coral reef fish response to sound. Aside from the intrinsic scientific value of studying the life history and behaviour of reef fish, it may be possible to use these results to control fish behaviour. For example, if artificial reef sounds are used, it may be possible to ‘direct’ larval reef fish into areas that require re-stocking or into areas designated as marine protected areas (MPAs).

Further studies are also required to determine the impacts of anthropogenic noise pollution on the marine environment. Clearly, if fish are highly sensitive to sound and use aural cues to navigate at important life-stages then we may expect adverse effects from noise pollution. There is a huge amount of anthropogenic noise in the oceans and this is increasing on a yearly basis:

− Supertankers (< 500 Hz, 205 dB) and container ships (~160-170 dB);
− Airguns (250 dB) and drilling rigs;
− Offshore wind turbines (< 500 Hz, 130 dB);
− Active sonar systems (< 500 Hz, > 230 dB; cf. earthquakes 240 dB);
− Plus pingers, ringers, loudspeakers, explosives, dredges;
− Dynamite fishing.

In fact, ambient noise in the world’s oceans has doubled since 1950 (a 20 dB increase) and this increase is likely to be affecting the whole marine ecosystem.
Paper 6, 2004: Discarding by the demersal fishery around Ireland

Lisa Borges¹*, Emer Rogan¹ & Rick Officer²

¹Aquaculture and Fisheries Development Centre, Department of Zoology, Ecology and Plant Science, University College Cork, Cork, Ireland.
²Fisheries Science Services (FSS), Marine Institute, Galway, Ireland.

*Adapted by Edward Codling from a poster presented by Lisa Borges at FSS symposia 2004. Email: lisa.borges@wur.nl

1. Methodology

This paper presents the first estimation of discarding levels by the Irish demersal fishery (demersal = bottom feeding fish). The analysis is based on the Irish discard programme: an on-board observers voluntary sampling scheme aimed at estimating discards rates of the demersal fisheries. The programme started in 1993 and, until 2002, 225 trips were sampled, corresponding to 2189 sampled tows. Environmental information (sea state, weather conditions and depth), tow location, gear type and species landings composition are recorded for each haul. A sample of discards (40 kg box) is randomly sampled from each haul and all fish species are identified, counted and measured. The total weight of discards per haul is estimated by subtracting the landing from the total catch estimated by eye by the skipper.

2. Fleets

Fleets are arbitrarily defined using ICES division, fishing ground, and gear used. 10 fleets are considered:

a) Beam trawlers (VIIa)
b) Scottish Seine (Vla, VIIa, VIIb, VIIg, VIIj)
c) Otter trawlers (Rockall bank, VIIb)
d) Otter trawlers (Stanton bank, Vla)
e) Otter trawlers (West of Achill, VIIb)
f) Otter trawlers (Aran Isles, VIIb)
g) Otter trawlers (Porcupine bank, VIIc)
h) Otter trawlers (VIIj)
i) Otter trawlers (Smalls, VIIg)
j) Otter trawlers (VIIa)

See Fig.1 for ICES division and fishing grounds.
3. Results: percentage of catch discarded

The following bar charts display the estimated percentage of the total catch that is discarded in the years 1993 – 2002 for the 10 defined fleets. A blank entry does not necessarily mean zero discarding and can represent insufficient data.

a) Beam trawlers

b) Scottish Seine
c) Otter trawlers (Rockall bank)

d) Otter trawlers (Stanton b.)

e) Otter trawlers (West of Achill)

f) Otter trawlers (Aran Isles)

g) Otter trawlers (Porcupine bank)

h) Otter trawlers (VIIj)
4. Results: percentage of species discarded

11 main fish species were identified in the discard samples:

- *Merlanogrammus aeglefinus* (haddock)
- *Merlangius merlangus* (whiting)
- *Helicolenus dactylopterus* (bluemouth)
- *Scyliorhinus spp.* (spotted dogfishes)
- *Eutrigla gumardus* (grey gurnard)
- *Trachurus trachurus* (horse mackerel)
- *Phycis phycis* (greater forkbeard)
- *Argentina sphyraena* (lesser argentine)
- *Lepidorhombus whiffiagonis* (megrim)
- *Pleuronectes platessa* (plaice)
- *Limanda limanda* (dab)
- Others

The percentage of each species in the discard samples for each fleet is given below. It is clear that even fleets fishing in similar areas with the same gear discard different proportions of each species.

a) Beam trawlers:

- 25% *Lepidorhombus whiffiagonis* (megrim)
- 23% *Limanda limanda* (dab)
- 21% *Pleuronectes platessa* (plaice)
- 20% *Scyliorhinus spp.* (spotted dogfishes)
- 11% *Merlangius merlangus* (whiting)

b) Scottish Seine

- 40% *Lepidorhombus whiffiagonis* (megrim)
- 32% *Merlangius merlangus* (whiting)
- 15% *Merlanogrammus aeglefinus* (haddock)
- 13% *Eutrigla gumardus* (grey gurnard)

c) Otter trawlers (Rockall bank, VIb)
- 63% *Merlanogrammus aeglefinus* (haddock)
  - 22% Others
  - 15% *Helicolenus dactylopterus* (bluemouth)

d) Otter trawlers (Stanton bank, VIa)
  - 51% Others
  - 20% *Merlanogrammus aeglefinus* (haddock)
  - 18% *Scyliorhinus spp.* (spotted dogfishes)
  - 11% *Merlangius merlangus* (whiting)

e) Otter trawlers (West of Achill, VIIb)
  - 42% Others
  - 16% *Eutrigla gumardus* (grey gurnard)
  - 15% *Helicolenus dactylopterus* (bluemouth)
  - 15% *Trachurus trachurus* (horse mackerel)
  - 12% *Scyliorhinus spp.* (spotted dogfishes)

f) Otter trawlers (Aran Isles, VIIb)
  - 37% Others
  - 27% *Merlangius merlangus* (whiting)
  - 20% *Merlanogrammus aeglefinus* (haddock)
  - 16% *Scyliorhinus spp.* (spotted dogfishes)

g) Otter trawlers (Porcupine bank, VIIc)
  - 53% Others
  - 16% *Helicolenus dactylopterus* (bluemouth)
  - 16% *Argentina sphyraena* (lesser argentine)
  - 15% *Phycis phycis* (greater forkbeard)

h) Otter trawlers (VIIj)
  - 52% Others
  - 13% *Lepidorhombus whiffiagonis* (megrim)
  - 12% *Trachurus trachurus* (horse mackerel)
  - 12% *Scyliorhinus spp.* (spotted dogfishes)
  - 11% *Helicolenus dactylopterus* (bluemouth)

i) Otter trawlers (Smalls, VIIg)
  - 57% Others
  - 43% *Merlangius merlangus* (whiting)

j) Otter trawlers (VIIa)
  - 35% Others
  - 32% *Merlangius merlangus* (whiting)
  - 22% *Merlanogrammus aeglefinus* (haddock)
  - 11% *Eutrigla gumardus* (grey gurnard)
5. Results: discard data on haddock, whiting and dogfish

Fleet based annual data (1993 – 2002) on the discard patterns for haddock, whiting and dogfish were also presented in the poster (not presented here). These results illustrate that discard patterns actually change over time within fleets and there is evidence that discarding of these species is increasing.

6. Conclusions

Up to 1/3 of the annual catch of demersal fish in the waters around Ireland is discarded. There is evidence that discarding is also increasing in recent years. 65% of total discards by weight were made up of three species: haddock, whiting and dogfish. There are high levels of juvenile discards for these (and other) species.

Each of the defined fleets had different discard patterns and species composition and population components were different in the discard samples from each fleet (even fleets fishing in similar areas with the same gear). High discarding of juvenile dogfish in the Porcupine and Smalls banks suggests that this may be a (previously unknown) dogfish nursery ground.
Paper 7, 2004: How consistent are the replicated age estimates of experienced blue whiting age-readers?

Gavin R. Power¹,*, Ciaran J. Kelly², Pauline A. King¹, David McGrath¹, Eugene Mullins², & Ole Gullaksen³

¹Commercial Fisheries Research Group, Life Sciences Dept., Galway - Mayo Institute of Technology.
²Fisheries Science Services (FSS), Marine Institute, Galway, Ireland.
³Institute of Marine Research, Bergen, Norway.

*Presented as a poster by Gavin Power at FSS mini-symposium 2004.
Email: nivagp@hotmail.com

1. Introduction

Effective fisheries management relies upon the most up to date and reliable scientific data available on the aquatic resource of interest. The current management of blue whiting, *Micromesistius poutassou* (Risso, 1810), in the North East Atlantic is largely dependant upon the results of annual age-based assessments. These assessments give fishery managers valuable information on potential changing trends in the commercial exploitation, age composition and stock recruitment dynamics of blue whiting. Age estimation of blue whiting by intact “reading” of sagittal otoliths, using stereo-microscopy, is considered to be the most commonly accepted method in use today by fisheries scientists in the northern hemisphere.

In age determination studies the term “precision” is used to describe “agreement” or variability between readings of the same specimen by the same or different age-readers (Kimura et al. 1991). The term “accuracy” is reserved to describe a comparison of ages generated by readers with the "true" age for specimens of known age (Kimura et al. 1991). Only by mark – recapture studies or use of known-age fish can all age classes in a population be validated and accuracy proven (Beamish & McFarlane 1983). In the absence of a known-age reference collection, ageing consistency is the best that can be achieved (Campana et al. 1995).

Results of recent international acoustic surveys indicate some differences in the age estimation of blue whiting between several nations. In light of these observations and the importance of a standardised approach to international stock assessment of this important species, this study has come about. This investigation sets out to determine the consistency of age determinations both within-reader and between selected international readers, for blue whiting, using an established graphical and statistical approach as used by (Campana et al. 1995). Results are discussed in view of the importance of reliable biological data to fish stock assessment and modelling of growth and yield in fisheries science.

2. Materials & Methods

A standard set of otoliths, N = 299, removed from adult blue whiting taken from the west of Ireland spring spawning area (I.C.E.S. areas VIa & VIIc, March 2003) were chosen for this analysis. These otoliths were
selected as they had been previously validated as part of a Marine Institute Ireland quality control blue whiting age-reading exercise and therefore provided un-biased age estimates for the purpose of the present study. Due to the commercial source of the sample, the otoliths of younger age groups were absent.

The otolith sample was circulated to two fish ageing laboratories in Ireland, i.e. one academic and one governmental lab and one fish-ageing laboratory in Norway. Three otolith “readers” were asked to carry out two independent age estimates of each otolith in the circulated sample. All age-readers were considered to be experienced in ageing blue whiting and the method of age estimation employed, i.e. microscope magnification and light intensity levels, were standardised as far as was possible. The three age-readers are henceforth referred to as age-reader A, age-reader B and age-reader C. One of the age-readers has considerable experience and has contributed to the training of one of the remaining age-readers. Both these age-readers have largely been responsible for the training of the one last remaining age-reader. Replicated age determination data was analysed for both within-reader and between-reader precision, using a combination of age frequency tables, age bias plots, Coefficient of Variation plots, standard regression and nonparametric statistical analysis.

3. Results

For the analysis of intra-reader precision, age frequency plots were constructed for each of the three pair wise age comparisons between age-readers. Such tables form a central component of many paired age comparisons but they are not particularly well suited to the detection of age differences between readers (Campana et al. 1995). Age difference plots on the other hand highlight major systematic differences between two sets of age readings. In age difference plots the difference between the two age readers, in years, is plotted as a function of one of the sets of ages. These plots were constructed for the three pair-wise age comparisons and highlighted under-ageing in comparisons with age-reader C, with no obvious under or over-ageing was evident in comparisons between age-readers A & B. Linear regression analysis was carried out on each of the three pair wise age comparisons between age-readers. A slope other than 1 suggests inconsistency in the interpretation of annuli by one of the readers, an intercept other than 0 suggests systematic differences between the two readers, perhaps due to different interpretation of the first annulus. Results show that readers A & B were least inconsistent and no significant difference was found between their paired age comparisons (p = 0.08) Both statistically significant inconsistency and systematic differences were found in comparisons between readers A & C and B & C (p = 0.001, p = 0.001 respectively). Results from the nonparametric one sample sign test for bias also highlighted significant differences in age estimates between readers A & C and B & C (p = 0.001, p = 0.001 respectively). Results of this test found no significant bias in comparisons between age-readers A & B (p = 0.63).

Intra-reader precision analysis was carried out using similar tests of precision and bias. Statistical inconsistency was observed within all age-readers using regression analysis (p = 0.001), with systematic differences in age estimates also observed for age-reader A. B & C (p = 0.001, p = 0.001, p = 0.004). The nonparametric one sample sign test for found similar results
with significant bias existing between the replicated age estimates of each of the three readers. Both age bias graphs and plots of coefficient of variation, (Chang, 1982) and percentage agreement were constructed using pair-wise data for inter reader precision analysis. Age bias graphs plot the age reading of one age-reader against that of a second age-reader and are interpreted through reference to a 1:1 equivalence line. Graphical analysis using age bias plots highlighted liner under-ageing in comparisons between age-reader A & C and age-readers B & C (CV = 14.4%, CV = 14.3%). This under-ageing by age-reader C became apparent at fish age 4, and appeared linear in relationship with older fish under-aged by approximately one year. No graphical differences were evident in comparisons between age-readers A & B (CV = 8.3%) using age bias plots. Lastly, plots of percentage agreement and CV highlighted a marked drop in % agreement and increase in CV at fish age 4 for comparisons with age reader C. Graphical plots of comparisons between age-readers A & B exhibited only gradual increases in CV, and gradual decreases in % agreement over increasing fish age.

4. Discussion

The combination of graphical and statistical approach to analysis was sufficient to both identify bias and imprecision and to describe the nature of the observed results when comparing pair-wise age estimates both between and within age-readers. In general, the precision of the blue whiting age-readers tested in this study was found to be poor. Inter-reader comparative analysis was complicated by an under-ageing bias detected in one of the age-readers in this study. This observed relative bias led to the identification of significant imprecision in both regression analysis and in results of the 1-sample sign test.

The combination of the age bias plots, regression analysis and 1-sample sign test enabled the detection of age-reader C in consistently under-ageing fish in comparisons with age-readers A & B. This identified relative bias may be due to the misinterpretation of ‘annuli’ in the otolith structure by age-reader C. Furthermore, age bias plots identified the under-ageing by age-reader C as following a linear relationship, with under-ageing only becoming apparent at fish age 4 or above. No significant imprecision was identified between age-readers A & B.

The results of intra-reader precision analysis showed that the reproducibility of estimates by all three readers was poor, even with high percentage agreement. Significant differences were found between replicated estimates, within all three age-readers, by both regression analysis and use of the 1-sample sign test for bias. However a problem exists in analysis using the 1-sample sign test as tied ranks are not included in the analysis, using instead only the ratio of positive and negative ranks in the computation of the test result. Therefore, the 1-sample sign test may have been over-sensitive to limited age differences as a realistic test for bias. In future analysis, a test that includes tied ranks may prove more robust, such as the One sample Wilcoxon test (Conover, 1998), a test which has been used in such analyses as that of Eltink et al. 2000.

The findings of this study indicate that problems exist in precision of age estimates between international age-readers on a limited scale. An age-reading workshop may provide valuable information on the broader scope of the problem in the future. Our results indicate that, differences in age
estimation of over one year between fish ageing laboratories, is evident. This potential ‘ageing error’ has worrying implications for the international stock assessment of blue whiting. The trend of above average recruitment in recent years is believed to be maintaining current levels of Spawning Stock Biomass (Anon., ICES CM 2006). With such a strong dependence on recruitment, under-ageing or over-ageing emerging year classes, even by one year, may pose a real threat to the reliable monitoring and assessment of blue whiting population dynamics. Under-ageing may incorrectly inflate the numbers of a recruiting year class giving an overstated estimate of recruitment success. Over-ageing may lead to the artificial strengthening of numbers in older year classes leading to possible overestimation of reproductive potential of the combined stock.

The modelling of such basic biological parameters such as growth may be greatly affected by relative ‘ageing errors’ such as those identified in the present study. The under-ageing of selected year classes will result in higher values of K indicating that the stock is faster growing, with maximum size reached quicker by adult blue whiting. Over-ageing selected year classes will result in lower computed values of K, and a reduction in the estimated rate at which maximum size is reached for blue whiting. Yield per recruit and surplus production models are similarly subject to overestimation or underestimation due to ‘ageing error’ in the biological data used in computation.

This study indicates that standardisation of criteria and protocols for blue whiting age reading are now necessary for the international scientific community. Such an approach will ensure the provision of reliable and sound biological data, necessary for both annual in-situ annual assessment of recruitment and spawning stock biomass and for the forecasting of total allowable catches, necessary to ensure the rational exploitation of this important stock.

References
Discussion from FSS Symposium 2004

The discussion session of the meeting took the form of open questions being presented to the floor by the chairman. The chairman then left the discussion to run its course, noting down the various relevant group responses. The group responses to each open question are given below. Please note that responses and comments are given in the form of a general consensus. The responses and comments presented should not be taken as indicative of the personal opinion of any of the individual participants.

Do we really need a ‘paradigm shift’ towards multi-species / ecosystem models and associated management strategies?

Initially the group raised a number of questions in relation to what exactly is meant by the possible paradigm shift. Is it really a new approach? Do we have a choice at all if ICES are changing their approach? Can we actually do it? The main concern seemed to be what is actually meant by the ‘ecosystem approach’ – there doesn’t seem to be clear consensus and explanation in either the literature or in management policy. The group considered whether the ecosystem approach implies different things for modelling and management – perhaps the modelling is okay but we need a paradigm shift in management! Do we actually need to model the ecosystem or should the managers simply take into account ecosystem considerations using advice from current models? There was some concern raised that perhaps managers / politicians / scientists may be pursuing their own agendas by pushing for the ecosystem approach.

If we assume that we need a change in the way fisheries models are used then the group raised several concerns. Although there was agreement that current single-species models are perhaps over-simple and new models other than VPA are needed, there was much concern that the wrong approach may be taken. The group agreed that large complex ecosystem models with many parameters are unlikely to be any more useful than current models, but would be far harder to understand and rely on much more data. Too many parameters results in over-parameterisation and may introduce more errors into the model. Rather than aiming to account for all variables with highly parameterised complex models, it was suggested we should be aiming for a far simpler approach where we accept that we can’t account for all uncertainty and errors. We should try and adopt an approach that includes the key features of the system and reproduces the qualitative ecosystem behaviour without being too complex (stochastic models would be appropriate here).

The group agreed that, with ecosystem modelling and management, it would be much harder to decide on and monitor key measures for the quality of the ecosystem. This problem would be confounded with the baseline problem (shifting ecosystem state) that is thought to be occurring in fisheries systems. It was also thought that there would be many problems in actually identifying the ecosystems and their habitats, along with their key features. All the group agreed that if there is a paradigm shift in fisheries then industry and stakeholders need to be kept involved and informed of any implications right through from the modelling process to the management process.
Is there any value left in deterministic modelling?

Many of the earlier presentations suggested that we should be using stochastic models rather than deterministic models so this was an interesting question to raise. There was general agreement that deterministic models are easier to understand and analyse than stochastic models. They can be used as a useful indicator or to produce a target point value. However, the group agreed that there is maybe a presentation problem with the results of deterministic models – people perhaps assume that the results are exactly what to expect rather than simply an indicator. The limitations of deterministic models are perhaps not explained clearly to managers. Stochastic models allow a level of risk to be incorporated into advice given to managers and they are then able to act upon this rather than just using a point value from a deterministic model.

Deterministic models can be used as the basis of a stochastic model (e.g. adding noise to a deterministic function) but some models are intrinsically stochastic and such an approach could be misleading. However, this approach would allow well understood deterministic models to be developed further, rather than starting afresh. There is perhaps a general assumption that a deterministic model would give the same result as the mean average of a stochastic model but this is not always the case and could be a dangerous assumption to make.

How can we develop useful models where data is scarce?

The general group consensus was that the current VPA suite of models will always be susceptible to problems if data is scarce. If VPA models are used then it may be possible to compare to similar stocks or areas using meta-analysis. However, a better approach may be to become less reliant on VPA style models and develop robust models that are not sensitive to scarcity of large amounts of data. One possibility would be to look at index methods (such as the oldest age fish in catches) that do not need much data. The group agreed that there is a fear of being honest about our uncertainty with most of the models used, even if full data is available. A model that gave advice as an assessment of risk rather than a point value would be more suitable when the data is scarce. If models are data hungry then uncertainty may increase as each data subset will have associated levels of uncertainty. This fits with the idea of developing robust models that are reliant on less data such as index methods.

How do we get fisheries managers to consider ideas for new approaches?

The group conceded that it may be difficult to overcome the current inertia associated with the status quo approach, particularly as there is some suspicion that there are certain agendas being pursued in the development of both the management and the science of fisheries. However, it was agreed that for any new approaches are to be accepted, we need consistent and rigorous scientific analysis to show why any new approach may be better. Any benefits / implications of any new approach should be clearly communicated to both industry and management. It was felt that honesty was always the best approach, particularly with regard to limitations in any new approach. It was agreed that any new approaches or models are more likely to be accepted if they are improvements on earlier
models rather than completely new ideas – this may not be a good thing though if the current approach / models are flawed!

The group agreed that we need a proper framework to compare and test models before developing them further. It was felt that current ‘new’ models are usually simply variants of the VPA suite of models – this is understandable if there is no framework to rigorously test a new approach so that it can be accepted by the scientific community (and then by management).

**Should we be concentrating on modelling management strategies rather than the detailed population dynamics of stocks?**

The group agreed that research is needed in both areas and that for any management advice to be useful it must be based on results that use a viable model for the underlying population dynamics. Given this, the group also agreed that the underlying population dynamics model doesn't need to be complex to give good management advice and should be based on data that is already available. One concern raised was the possibility of the ‘shifting baseline’ problem – where the population dynamics are continually changing and long-term management strategies may thus be affected. It was suggested that as many stocks are at low levels, detailed research into their population dynamics may be misleading – instead strategies should be devised to bring the stock back to safe levels where the population dynamics may be more typical. The main conclusion the group raised was that, although in depth research into population dynamics is useful, the final product that managers want from fisheries scientists is advice. Modelling and simulating different management strategies using simple but realistic population models seems the best way to go about producing useful advice.

**How do we get industry more involved in the process of fisheries modelling/assessment?**

Generally the group agreed that there was not enough industry involvement at the level where modelling takes place and from where most advice emerges. This can lead to problems where the industry may be suspicious of any new results or strategies developed. There was a feeling that the industry may not ‘trust’ fisheries scientists in general. This may have come about because of scientists in the past not being honest about the limitations of most of the modelling processes that produced the advice. Fisheries scientists could point to the fact that their models will never work if there is large misreporting and the industry is not correctly regulated. The group agreed that these problems could only be overcome if the industry and scientists move closer together rather than apart. If the industry understands the scientific processes better then they would perhaps be more open to the benefits of different management strategies based on scientific advice. The group all agreed that most of the current problems are due to the ‘gold-rush’ problem of open seas. The solution to this problem would have to include some sort of devolved responsibility where the industry has an interest in preserving the fish stocks. It was suggested that there may be other important interests outside the fishing industry who could be considered ‘stakeholders’ and they should be informed of the science behind the advice given.
3. FSS mini-symposium 2005 – ‘Bridging the gap: the science and management of fisheries’
Introduction to FSS mini-symposium 2005

The 2005 FSS mini-symposium was advertised with the following objectives:

‘The mini-symposium is intended to promote debate on:

i) the current fisheries management system, the data it requires, and the types of assessment and forecasts required to carry it out;

ii) alternative approaches to resource assessment and future stock states, which would underpin a more robust approach to fisheries management;

iii) the processes that are used to plan, evaluate and implement such management strategies in the real world.

In particular, we want to explore the possible reasons for the divergence that can occur between scientific results and advice and management actions. The meeting aims to bring together academics, fisheries scientists, and managers in order to exchange ideas and allow discussion of the current issues being faced in fisheries science and management.’

The meeting was attended by representatives from the Marine Institute and BIM (Ireland) and various universities in the UK and Ireland (see Appendix). Unfortunately, members of DCMNR (the government department responsible for fisheries management in Ireland) declined to attend, so there was no direct representation from those involved with managing Irish fisheries. However, a number of varied and interesting presentations were made relating to all aspects of fisheries management around Ireland and the symposium created a lot of stimulating discussion.

Included in this section are two papers (Paper 3, 2005 by Jon Pitchford and Paper 5, 2005 by Emmet Jackson and Dominic Rihan) that include content that was presented at both the 2004 and 2005 symposia.

The first two papers (Paper 1 and Paper 2, 2005 both by Ciaran Kelly and Edward Codling) go under the joint title of ‘Cheap and dirty’ fisheries science and discuss a possible new approach to fisheries management using indicators (that act as a proxy for the state of the stock) rather than information about ‘absolute’ stock levels. Paper 1, 2005 is a discussion paper that described the current problems with the traditional ‘absolutist’ approach to fisheries science and management as practised by the ICES community. This approach is extremely expensive as huge amounts of data collection is required, but the system breaks down when the data is unreliable (as has happened at recent important ICES assessments). An alternative approach is suggested for stocks where data is unavailable or where data collection is too expensive. This approach uses proxies such as catch per unit effort (CPUE) or age structure (see Paper 4, 2004 by Marco Kienzle in this publication) to manage the fishery. Paper 2, 2005 continues this discussion and gives a simulated example where a stock is successfully managed using only the oldest age proportion of the stock and a simple harvest control rule (HCR).

The third paper (Paper 3, 2005 by Jon Pitchford) also takes the form of a discussion, where the main point raised is that deterministic models are
not suitable for describing and/or analysing highly complex systems that can be extremely variable at the individual level. Examples of stochastic models for fish recruitment and management using marine protected areas (MPAs) are given to illustrate this point. Paper 4, 2005 by Leonie Dransfeld summarises the ecology of Celtic Sea herring. The specific biological characteristics of the stock and the underlying oceanographical features are discussed in relation to how current management strategies have arguably failed. Suggestions for alternative strategies are made that directly account for the stock life history and ecology.

The next paper (Paper 5, 2005 by Emmet Jackson and Dominic Rihan) presents the main results from a series of technical gear trial experiments completed by BIM. The experimental results clearly show that using technical gear modifications can dramatically reduce the bycatch of fish from Nephrops fisheries and the implications for management of these stocks is discussed. Following on from this paper, Paper 6, 2005 by Dominic Rihan summarises how the role of gear technology and selectivity is now being given more consideration by ICES in response to recent changes in how advice is presented. The paper discusses what information is required and who can provide this so that useful advice can be given. The final paper (Paper 7, 2005 by Olliver Tully) considers the management of shellfisheries in Ireland. The new management framework (established by DCMNR in 2005) for shellfisheries in Ireland is discussed and some general points are made contrasting management of shellfisheries to typical ‘offshore’ fisheries.

The final part of this section is a report of the round-table discussion session completed at the 2005 symposium. As with the 2004 symposium, the discussion session consisted of a number of questions being raised by the chairman. The group discussed each question in turn and a record of responses and comments was kept. The report presented here is a summary of the main responses and comments and acts as a record of the general consensus of the group. The responses and comments presented should not be taken as indicative of the personal opinion of any of the individual participants.
Paper 1, 2005: ‘Cheap and dirty’ fisheries science part I

Ciaran Kelly1,* & Edward Codling1

1Fisheries Science Services (FSS), Marine Institute, Galway, Ireland.

*Presented by Ciaran Kelly as a talk at FSS mini-symposium 2005.
Email: Ciaran.Kelly@marine.ie

1. Introduction

“...There are known knowns, there are things we know we know. We also know there are known unknowns; that is to say, we know there are some things we do not know. But there are also unknown unknowns - the ones we don't know we don't know.” Donald Rumsfeld 2002.

Despite the fact that he was ridiculed in the press for this statement, Rumsfeld was making a lot of sense, and referring to process management theory. There is a whole field of science behind managing uncertainty, but very little of it has found its way into fisheries. We are, still to a great extent, stuck in the paradigm of absolutist predictability. This is because fisheries science has, with much hubris, set out to provide managers with answers to questions, which scientists themselves would like to pose, rather than trying to find solutions to fisheries management problems. Fisheries science has not embraced the concept of uncertainty, rather we have tried to “solve it”. That is to say by ignoring the “unknown unknowns”, much of the predictive capacity of existing models is eroded, especially in terms of their usefulness to managers.

2. Justification & Background

There is a natural tendency to want to understand a fisheries system in an absolute sense, for example “How many fish are there now?” or “How productive is a fish stock?”. In some situations there is enough information to answer these questions but, in general, fisheries science does not have a particularly good record with these types of questions. One of the reasons why these questions are notoriously difficult to answer is because they require data that are informative, that is, reasonably precise, accurate and high contrast. High contrast data is where observations are available over a wide range of system states. Many fisheries datasets are inaccurate, imprecise and low in contrast. There is usually limited information about the historical state of a fish stock. Although absolute indicators such as biomass and fishing mortality are appealing they cannot be estimated reliably when only un-informative data are available. An alternative type of understanding of a fishery system is with a relative sense, where only changes to stock status are indicated. For example, a relative question might be, “Is the stock state now different to what is was five years ago? Or “Is the catch per unit effort decreasing?” These questions are more amenable to low contrast data because they are about system change rather than the absolute state.
These are the types of questions that can be answered using theory from process management.

Let's look for a moment at the requirements for the collection of high contrast accurate data for absolute indicators. This is exemplified by the conventional assessment approach that attempts to estimate biomass and fishing mortality rates. I am not saying that this approach does not have a place in the management of fisheries, it certainly does, and will probably be the primary approach for the foreseeable future. Population modelling will be fundamental to management strategy evaluation and constructive in the interpretation of empirical indicators. However, because of the material and opportunity costs, this approach cannot be applied exhaustively. Even now the “absolutist” approach is being undermined by the deterioration of basic landings data. Population modelling cannot, therefore, be the linchpin of fisheries resource assessment for the future.

For example a rough calculation on the cost of servicing the current list of stocks assessed by ICES:

- Stocks on which Advice is given: 113 stocks
- Estimate 75% age based assessments: 85 stocks
- Conservative estimate of 10 metiers (1.25 ICES subdivisions 4 quarters and 2 gears & an average of 30 length classes per stock & 10 age readings per length class: 10 x 30 x 10 x 85 = 255,000 age readings
- Each otolith needs to be reaged: 255,000 x 2 = 510,000 age readings
- FSS audit revealed that it takes 0.21 hours to sample, prepare and read an average otolith, and 0.103 man hours to get a length measurement.
- Assuming 60% overheads on 1590 man hours/year = 636 ageing hours per year
- Therefore it takes approximately 168 man-years to service current ageing requirements
- The same calculation for lengths implies 165 man-years to service the length measurements
- Therefore it takes approximately 333 man-years to provide age and length data.

If you consider there are 19 countries providing data to ICES with approximately 1.2 labs by country this means that there are 22.8 labs, thus 14.6 people per lab would be required to service the current estimated demand for age and length data. This is about the current capacity of most EU labs for fish sampling/ageing techs.

If you assume an average wage of €35k, this means that the cost of providing age/length data to service assessment WG’s is about €11.65 million. In addition, the cost of fuelling the TAC machine by ageing and measuring fish to conduct assessments also requires analytical scientists and administrative personnel. Not to mention assessment surveys, discarding programmes or of course funds required to look at other issues. If we look to other countries where this process is more rationalised (e.g. Australia) the cost of conducting fisheries monitoring, assessments, advice and research is funded at 0.5% of the value of the sector. In 2003 the value of the EU fishing sector was less than €5 billion (by comparison Agriculture is valued at €164 billion). This implies that EU funding for fisheries science
should be less than €24 million. Therefore at a rough calculation we are spending almost 50% of what should be the total allocation for monitoring, assessment, advice and research, on the collection of age and length data for currently assessed stocks. And remember that this does not include any inshore species, nor the cost of any other monitoring or research, such as surveys, discard studies, gear trials etc..

This is unsustainable and we need to think of alternatives for a number of reasons

- The EU cannot support the future cost
- The TAC system which this work underpins does not work anyway
- The number of stocks on which advice is required is more likely to increase

3. Calculating Indicators

Methods such as VPA have dominated the assessment of north Atlantic stocks over the last twenty years but are slowly being replaced with synthetic methods that integrate a wider source of information. There is now recognition that indicators derived directly from length and age based data can be informative within resource assessment.

With few exceptions, model-based estimates of fishing mortality and biomass require the use of iterative non-linear equation solving or optimisation algorithms. These estimates are usually dependent upon additional assumptions, or prior probability distributions, of natural mortality and other key parameters (which are usually not well known). Like the internal combustion engine, many currently used assessment models are highly evolved in complexity, but fundamentally based on a simple model with important assumptions.

Alternatively if you accept that there are “unknown unknowns”, then you realise that, with poor resolution data, you may not be able to predict the future state of the system with the precision required for management. In this case from a management point of view what you are interested in is if things are getting worse, or “out of control”. This is the basic principle behind process management. To track if things are under or out of control, you need an indicator and some carefully defined thresholds. So long as the system is within the thresholds it doesn’t matter how it evolves, but as soon as it becomes out of control you know you need to take corrective action. In fisheries science such indicators would be empirical measures of stock status. These would be calculated using non-iterative algorithms with few parameters. For example, mean length is calculated by weighting the average length of sampled fish by the proportion of commercial catch landed at the port at which they were sampled. These calculations are very simple and there are software packages that can reliably and transparently assist in the analysis. Data required to calculate empirical indicators are, or should be, stored in a controlled environment, such as on a SQL relational database server. Algorithms used to calculate the indicators should be coded as SQL queries or SQL stored procedures where the calculations are directly linked to the data tables. In cases where this would create un-maintainable code (such as calculating a percentile) then the indicators should be calculated within procedures or functions written using the scripting language of a
statistical or software development package (e.g. SAS, S+, R, C or Basic). In such a situation, “select” queries should be used to extract the raw data from the database and “update” queries used to store the results of the calculations back in the database (to be eventually extracted by another select query).

4. Interpreting indicators

The simplest resource assessment will be a chart of an empirical indicator associated with expert interpretation and judgement. There is nothing wrong with this approach except that the reference model is often implicit (what model is the data being compared to?) and often there are no quantitative trigger points. Usually these indicator charts are associated with a qualitative description of the pattern. Once the indicators are presented, experts can sometimes have exceptional insight into the processes that are responsible for those patterns. For example, a simultaneous decrease in mean length with a decrease in catch or CPUE can be evidence of growth over-fishing. There should be development of analytical tools that enable rapid visualisation of indicators and patterns between indicators so that the expert’s time is spent examining patterns rather than extracting, transforming and loading data. The greatest challenge when using expert judgement is the conversion of an implicit analysis to an explicit trigger value. Various statistical rules can be applied or a consultative process can be undertaken to define that trigger value.

Age and length datasets are extremely useful for resource assessment. Age composition data provides a unique demographic snapshot of a population and, based upon the relative strength of younger age-classes, is capable of predicting future exploitable biomass. The age-distribution of catch can be used to measure total mortality and thus used to estimate fishing mortality. Furthermore, indicators such as maximum age provide a metric of the suitability of a species for commercial and recreational harvesting. Length-composition data can be used for similar purposes as age-composition data but because of the highly variable and asymptotic relationship between age and length, the demographic information content within length-composition data is compromised. Large numbers of fish-age estimates from the catch (or independent surveys) will always be considered international best practice within the assessment and management of higher valued fish stocks. That said, there may not be the need for a superior demographic indicator, but simply an indicator that can detect substantial changes to population structure. For this latter purpose, length-based information may be sufficient and will certainty form the basis of a superior stock status indicator in comparison to a questionable indicator of abundance. It is notable that many invertebrate fisheries are equally as well assessed and managed on the basis of length-data as finfish are on age-data. Pragmatic issues such as costs are likely to dominate the sampling decisions for most species.

Even catch data could be used as an indicator under special circumstances (e.g. where catch is proportional to effort, i.e. where there is no excess capacity). In this case a smoother could be applied to the catch data (e.g. running mean). Thresholds could be defined arbitrarily. In this case risk is based on the threat to economic yield, and the assumption is
that catches within the thresholds are sustainable, and that corrective action
taken when the system is "out of control" is sufficient to return the system
within the thresholds. A more developed example is presented in the next
paper, where the proportion of the stock over a certain age is used as a
proxy for exploitation. The stock is managed according to a harvest control
rule that adjusts the TAC based on this proportion.

These are still simplistic examples where the definition of “out of
control” is based on a simple application of $|X_t| > h$ (h = decision interval),
this is the equivalent of a Shewhart chart, which is a common diagnostic used
in process control. The procedure can be made more robust by using a
running mean. Another approach commonly used in process control is
CUSUM. This is similar to a Shewhart chart with tolerances but is more
effective at signalling persistent causes because it captures the memory
within a sequence of indicator values:

$$\Phi^+_t = \max(\Phi^+_{t-1} + X_t, -K)$$
$$\Phi^-_t = \min(\Phi^-_{t-1} + X_t, +K)$$
$$\sum \Phi^+_t > h$$, where K is the tolerance level.

There is of course a trade-off between the decision interval and the
usefulness of the procedure at identifying real events. Too wide and it has no
sensitivity, too narrow and it is triggered by uncertainty in the data. The more
robust a test the more reliable it is at not signalling a situation that is not
anomalous. This trade-off between the sensitivity and robustness of a test is
very well developed in health medicine, but does not appear to be used in
fisheries science. In health medicine ROC’s (Receiver-Operator Curve) are
used as a diagnostic tool to examine the objective performance of a test.
These curves are simply a plot of the probability of a true positive result and
a true negative result over a range of decision intervals:

Sensitivity $P(T+) = P(T+)/P(T+) + P(F-))$
Robustness $P(T-) = P(T-)/P(T-) + P(F+))$

The performance of the indicator is then given by the area under the curve.
This approach integrates out the decision interval (which could be
considered as a nuisance parameter)

5. Conclusions

Fisheries science has historically informed and shaped the current
fisheries management approach in the north Atlantic. From a management
point of view the system does not perform well, not only because of poor
enforcement, but also because it does not cope well with uncertainty,
especially unknown-unknowns. Applying the absolutist approach
exhaustively to all stocks is untenable financially, so there is a need to find
cheaper methods that perform as well from a management point of view.
Such an approach could be developed from process control. This approach
would not completely replace traditional stock assessment, but would allow a
rationalisation of the cost of the current system while fulfilling the
requirement for stock monitoring and advice provision. Detailed
assessments could then be applied on a priority basis depending on the
stock value (either economically or socially).
Paper 2, 2005: ‘Cheap and dirty’ fisheries science part II: an example of fisheries management using simple indicators

Edward Codling1,* & Ciaran Kelly1

1Fisheries Science Services (FSS), Marine Institute, Galway, Ireland.

*Presented by Edward Codling as a talk at FSS mini-symposium 2005.
Email: Edd.Codling@marine.ie

1. Introduction

The current system of fisheries science and management in the EU / ICES area is failing and alternative strategies are required. Many stocks are at historically low levels and considered to be overexploited. The possible reasons for the failure of the ‘ICES system’ are varied. In general, the ‘ICES system’ of fisheries science consists of data collection, stock assessments and advice, followed by political negotiations and setting of TAC (total allowable catch) quotas.

Data collection is completed at national levels and consists of ship surveys, port sampling, analysis of logbook data, and aging of fish samples. Much of this work (e.g. in aging) may be repeated across the EU in different national labs, although rationalisation of this process is planned. There are significant problems with the reliability of this data – recent comparisons of aging techniques between labs suggest that there can be wide discrepancies. There are also the significant problems of ‘black’ or misreported landings and unknown discarding, which will jeopardise stock assessments unless sufficient information is gathered.

Assessments are completed at annual Working Groups using data collected from the national labs (and as such, are sensitive to any errors in the data). Most assessments are based on standard VPA (virtual population analysis) models which are themselves based on theoretical work from the start of the 20th century (i.e. Baranov, 1918; Beverton & Holt, 1957). Although the VPA models may include stochastic elements to account for system uncertainty, the results are usually presented as deterministic short-term forecasts only – suggesting that we can accurately predict the stock levels for the immediate future. However, this is not the case and all assessment forecasts will incorporate some level of uncertainty and risk. Recent important assessments (e.g. WGNSDS (ICES, 2005)) have been rejected due to unreliable data arising from misreporting of landings. This raises a serious problem, as ICES advice will carry less weight if assessments have not been completed.

EU politicians annually use the ICES advice arising from assessments to set TACs. However, although scientific factors are considered, this usually amounts to little more than ‘political horse-trading’. The outcome is short-term annual quotas that are obviously not ideal for fisherman as they are not able to plan ahead with any surety. A long-term approach based on assessing the level of risk to stock, balanced against stakeholder interests...
would make more sense but this is, as yet, just an ideal to aim for and not a reality.

The result of these different elements is:

- Huge expense to support ICES fisheries science;
- Many important stocks still over-fished;
- Several stocks need urgent recovery action;
- Data is no longer considered reliable;
- Rejected assessments.

Clearly the system is not working even with the huge expense involved with each element and we should not carry on like this.

2. Using Indicators

2.1 The ‘absolutist’ approach

The traditional (what we term ‘absolutist’) ICES approach consists of scientists attempting to collect as much data about the stock as possible. These data requirements are based on VPA models that attempt to replicate all (important) stock dynamics. An implicit assumption of this approach is that we can ‘control’ stock dynamics by micro-management of TAC (or effort). The ‘absolutist’ approach requires a large data set: ‘at age’ data on population numbers, F, weight, maturity, etc; plus time series data: recruitment history, catch history, etc. However, the assessment (VPA) models will usually break down or give meaningless results if any part of the data set is missing or unreliable. This is not uncommon as witnessed by recent failed assessments for important stocks (ICES, 2005). However, it is clear that this approach has not worked and alternatives should be considered.

2.2 An alternative approach using indicators

In contrast to the ‘absolutist’ approach, the alternative we suggest here implicitly means that we should accept we can’t know everything about a stock. Instead we choose one (or more) indicator(s) that are proxies for the condition of the stock, e.g. proportion of older age fish in the population, recruitment, Catch per unit effort (CPUE), etc, etc.

A good indicator should be easy (and cheap) to measure, would ideally be fishery independent (although this isn’t absolutely necessary), and most importantly it should be able to trace the dynamics of the stock and act as a proxy for the ‘health’ of the stock.

An indicator on its own will just give a relative measure of the stock status. It is also necessary to set up and implement a well-defined management strategy or harvest control rule HCR based on indicator level and management objectives. The HCR should contain explicit instructions for required actions when the indicator is at different levels. Clearly different stocks will require different HCRs (and possibly different indicators). Any HCR should be set up only after considering the full knowledge we have of the stock dynamics, and after simulations have been completed to assess the levels of risk of various HCR strategies.
The idea of this approach is to ‘take a step back’ and put our efforts into correctly and efficiently managing the stock, rather than spending all resources on endlessly collecting data (which may not even be useful). It should be stressed that compared to ‘well managed’ ICES stocks, the indicator approach may not actually improve management but it will be significantly cheaper than the current approach.

3. Virtual fishery

Simulations were completed to test the ‘indicator approach’ against a standard model of ICES management. The simulations were completed using the F-PRESS simulation tool.

3.1 F-PRESS simulation tool

The F-PRESS simulation tool is a stochastic, single species, non-spatial, age structured stock and fishery simulation. Separate operating, assessment and management models are included so that it is easy to compare e.g. different assessment methods or different HCRs.

For each management strategy / scenario tested, 100 iterations were completed in each simulation run, where each simulation run lasted for 50 virtual years so that long-term average statistics could be collected and compared.

3.2 Virtual stock

The simulations use a fictional virtual stock based on a ‘recovered’ Irish Sea cod stock (SSB starts off much higher than in reality but all other stock characteristics are the same). Thus, the stock dynamics comprise of a fast growing fish, with ages 0 (recruits) to 7+ (plus group). We use a density dependent recruitment function based on a stochastic (randomised) Ricker function.

3.3 Simulated assessment

We use two different models for the simulated assessment. The first model simulates the ‘standard’ ICES approach – we simply assume a noisy (randomised, CV = 0.1) and possibly biased estimate of SSB is used in the HCR (we assume no ‘assessment feedback’). In reality, this estimate of SSB would require all the usual at age and historical data and for assessment software to be run, and we are only aiming to replicate the qualitative errors in ICES assessments.

The second model simulates the indicator approach. A noisy (randomised, CV = 0.1) estimate of the proportion (p) of the population 5 years or older is used in the HCR. In reality this data may only require sampling of landings (port sampling), although this would assume that landings are representative samples of the total population (i.e. this wouldn’t work if young fish were more likely to be discarded – discard data would be required in this case).
3.4 Simulated management

Various HCRs are used to simulate the management process:

Control:
- \( \text{TAC} = 10,000 \) every year (no management).

‘ICES’ HCR 1:
- If \( \text{SSB} > B_{\text{ref}} \), \( \text{TAC}^* = 10,000 \);
- If \( \text{SSB} < B_{\text{ref}} \), \( \text{TAC}^* \) reduced by factor \( \text{SSB} / B_{\text{ref}} \).

‘ICES’ HCR 2:
- If \( \text{SSB} > B_{\text{ref}} \), \( \text{TAC}^* = 10,000 \);
- If \( \text{SSB} < B_{\text{ref}} \), \( \text{TAC}^* \) reduced to 0;

Indicator approach:
- If \( p > 0.1 \), \( \text{TAC}^* = 10,000 \);
- If \( p < 0.1 \), \( \text{TAC}^* \) reduced by factor \( p / 0.1 \);

\((p \) is proportion of 5 years or older in population).\)

* In all simulations the change in TAC from year to year is limited to 30% - this acts to ‘smooth’ out TAC changes from the HCR.

Note that \( B_{\text{ref}} \) in the above HCRS has no real meaning and is simply a reference point SSB for managing the stock. The value of 0.1 used in the indicator HCR was found to be a sensible choice for the stock after evaluating several simulation runs. If this approach was adopted in reality, then this figure would need to be calculated from biological knowledge of the particular stock and from simulation results.

4. Simulation results

Figs. 1 & 2 show time series plots from a single projection (100 projections are run and results averaged to give final statistics) using indicator and ‘ICES’ assessment (10% bias, \( B_{\text{ref}} = 10,000 \)) respectively. Both plots illustrate recruitment driven dynamics but only the indicator HCR is sensitive enough to trace the history of the stock SSB level, while the ‘ICES’ HCR results in stable catches but may not be sensitive enough to sudden drops in the SSB level.
Figure 1. Time series plot for indicator HCR. It is clear from the plots of SSB and catch that the HCR traces the level of SSB, although average catch may be affected.

Figure 2. Time series plot for ‘ICES’ HCR. The HCR results in a stable catch but may not be sensitive to changes in SSB.
Table 1 gives the summary results from different HCRs run using the methods described in Sec.3. The table gives the probability of falling below 6000 tonnes (considered as ‘extinction’ level for this fictional stock) and 10,000 tonnes (considered as ‘critical’ level for this fictional stock) in any particular year of the 50-year projection. The mean catch averaged over all projections and all years is also given. Note that a low average catch can occur because of a HCR reducing catch or because the stock has been overexploited and become extinct. HCRs where the annual TAC is capped (maximum TAC = 10 000 tonnes) gives recruitment driven cyclical stock dynamics. Where there is no limit on TAC then the result is catch driven cyclical stock dynamics.

<table>
<thead>
<tr>
<th>Scheme</th>
<th>(P(&lt; 6000))</th>
<th>(P(&lt; 10 000))</th>
<th>Mean Catch</th>
</tr>
</thead>
<tbody>
<tr>
<td>Control</td>
<td>19.9%</td>
<td>20.9%</td>
<td>8106</td>
</tr>
<tr>
<td>ICES 1</td>
<td>12.9%</td>
<td>14.2%</td>
<td>8791</td>
</tr>
<tr>
<td>((B_{ef} = 10 000, 10% bias))</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>ICES 2</td>
<td>10.8%</td>
<td>11.9%</td>
<td>8905</td>
</tr>
<tr>
<td>((B_{ef} = 10 000, 10% bias))</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Indicator</td>
<td>2.4%</td>
<td>2.8%</td>
<td>8817</td>
</tr>
<tr>
<td>ICES 2*</td>
<td>1.5%</td>
<td>1.8%</td>
<td>9272</td>
</tr>
<tr>
<td>((B_{ef} = 20 000, no bias))</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Table 1. Summary statistics for the various HCR strategies.

From Table 1, it is clear that in any HCR strategy there is a balance between i) risk to stock and ii) catch (both mean catch and stability of catch). It is possible to achieve a low extinction probability and high yield with the ‘ICES’ approach – we do not claim that the indicator approach is ‘better’, simply that it can be ‘as effective’ and will be cheaper. With refinement, optimal HCRs can be found to manage the stock for both assessment methods (ICES and indicator approaches).

These results illustrate our key point that we wished to demonstrate:

Theoretically it is possible to successfully manage a fishery using indicators!
5. Discussion

Our key points are:
− ‘Absolutist’ ICES approach is failing and expensive;
− Indicator approach can deal with limited data;
− Indicator approach is not necessarily more effective than ‘absolutist’ approach (if / when it works) but should be much cheaper.

There are a number of considerations that should be made when interpreting these results. Firstly, we have only shown that the ‘oldest-age proportion’ indicator works with a stock that is initially in a stable stock. Simulations should also be completed with stocks in recovery states. Similarly, we have only considered a single stock with associated dynamics, simulations should also be completed with stocks exhibiting different dynamics (e.g. short-lived stocks, stocks with recruitment peaks etc).

Possible problems with this example of an indicator could include discarding (although this is also a problem in the standard ICES assessment) and physiological changes in stock where the HCR is predetermined and fixed.

Other potential indicators of stock status could include proportion of young-age fish (this would track recruitment peaks / failure), catch per unit effort (although this is never likely to be fishery independent data), and potentially many others.

It seems likely that a combination of simple indicators (with a sensible HCR) is likely to give a good management performance and will still be much cheaper than the current assessment and management approaches. This could be useful for stocks where data is limited or unreliable, or where the stock is not valuable enough for the full data-collection and assessment process. For valuable stocks with good data, the indicator approach could be used as a relative measure to compare to the standard ICES assessment and help inform management (we do not advocate that it should replace the standard ICES assessment for these stocks but it could complement it).

References
Paper 3, 2005: Simple stochastic models for fish and fisheries

Jon W. Pitchford¹⁶

¹University of York, UK

*Presented as talks by Jon Pitchford at FSS mini-symposium 2004 and 2005. Email: jwp5@york.ac.uk

†The research discussed is joint work in collaboration with Paul Baxter (Leeds), John Brindley (Leeds), Edward Codling (UCC / FSS), and Alex James (Canterbury, NZ), together with York graduate students Jenny Dawson, Qiming Lv and Despina Psarra.

1. Introduction

To successfully manage a fishery, we need to accurately assess and safely exploit existing adult populations, and to understand how new recruits enter these populations. Both of these factors are notoriously vulnerable to uncertainty. Stock assessment is an imprecise business dependent on the quality and quantity of data available, scientifically set quotas are subject to political interference and bargaining, and fishers cannot predetermine catches and may be tempted to misreport when in excess of quotas. Recruitment is perhaps even more variable; larval fish suffer enormous mortality and are subject to the vagaries of ocean turbulence, prey patchiness and variability, and predation.

In this short summary, I will argue that any description of fish and fisheries that ignores uncertainty will certainly be wrong, and could possibly be catastrophically wrong. I will explain how reasonably simple stochastic (random) mathematical models can provide important insights into the behaviour of marine systems, and that such models should be used and understood by managers to provide sustainable and profitable fisheries. The models under discussion operate at three scales: the growth of individual fish larvae; the recruitment of populations of fish larvae; and the management of a fishery using traditional quotas, harvest control rules, or marine protected areas. The understanding of stochasticity derived from the individual-based model becomes an input to the population-based model, which in turn becomes a part of the understanding of uncertainty in fishery management at the largest scales.

2. Individual scale - the growth of fish larvae

When modelling the growth of a predator searching for prey, the temptation for a mathematical modeller is to write down an equation of the general form:

\[ \text{rate of growth} = (\text{average rate of prey encounters}) \times (\text{conversion efficiency}) - \text{metabolic costs} \]
One would hope that such an equation, suitably parameterised, would describe the growth of an individual from ‘birth’ (spawning or hatching) to ‘maturity’ (metamorphosis, recruitment, or adulthood). Unfortunately, for fish larvae the above formulation turns out to be inadequate – it will typically predict growth rates that are too small, commonly resulting in predictions of zero recruitment in contradiction to field observations.

The reason the above (deterministic) model fails is basically that the ‘average’ fish larva is dead; only the luckiest larvae survive to recruitment. In other words, one must consider the distribution of growth rates of individuals, anticipating that most recruitment will come from individuals in the upper tail of this distribution.

Pitchford and Brindley (2001), following the approach of Beyer and Nielsen (1996), show how this distribution of growth rates can be derived using very minimal assumptions. Basically individual larvae are treated as SMALL (relative to oceanic turbulence, and relative to the size of their predators), STUPID (their foraging behaviour is visual, based on only very local knowledge, there is limited evidence for long-term memory or complex foraging behaviours) and usually DEAD (there is massive mortality throughout the larval stages). These assumptions allow one to characterise not only the average prey encounter rate \( r \), but also the variance in this encounter rate \( \sigma^2 \) experienced by individuals. Both \( r \) and \( \sigma^2 \) depend on the swimming speeds and perceptive fields of predators and prey, but they are also sensitive to the levels of oceanic turbulence and prey patchiness. Importantly, it is shown that whether or not prey is distributed homogeneously (e.g. patchy versus uniform distribution of zooplankton) has no effect on average encounter rate \( r \), but can have a very significant effect on \( \sigma^2 \). This sensitivity can be quantified using simple equations.

3. Population scale – quantifying recruitment

Armed with individual-based knowledge of \( r \) and \( \sigma^2 \), one can then use these values to compute the statistical distributions for how the size of individuals varies within a large population. The problem can usually be described as a ‘first hitting time problem’ for a stochastic differential equation (SDE). Although SDEs are much more difficult to deal with than their deterministic counterparts, their results are much more realistic, especially when describing statistically rare events such as recruitment. Furthermore, many of the necessary mathematical and computational techniques can be borrowed from recent progress in the field of Mathematical Finance (each larva is a binary option, \( \sigma^2 \) is the market volatility).

Pitchford et al. (2005) considers the simplest possible case of a larva growing at a constant mean rate, with a constant level of stochasticity (i.e. \( r \) and \( \sigma^2 \) are constants). This case can be dealt with analytically, and two significant results emerge. Firstly, compared to a seemingly identical deterministic model, the stochastic model predicts increased recruitment, often by up to an order of magnitude. Stochasticity is especially beneficial in the high mortality, high \( \sigma^2 \) environment in which fish live and die. Secondly, trying to infer the average population growth rate by catching surviving individuals will always produce a biased estimate; the observed growth rate will be larger than \( r \) because it is only the lucky (fast growing) individuals that are ever observed. Again, large mortality and large \( \sigma^2 \) enhance the effect.
This simplest of models can be extended to account for more realistic growth functions, and for stochasticity which changes over the individual’s lifetime (i.e. \( r \) and \( \sigma^2 \) become functions of the organism’s size). Lv and Pitchford (in submission) use the widespread Von Bertalanffy growth model in scenarios where stochasticity increases, is constant, and decreases with body size, motivated by various ecological regimes. James et al. (2005) use queuing theory to implement a rather more complicated physiologically based model for growth and mortality (see Paper 3, 2004 by Paul Baxter et al). Although the mathematics becomes much less tractable, the general conclusion is unchanged: stochasticity benefits recruitment and causes an upward bias in any measure of observed growth rate.

These SDE models present an exciting opportunity to model random processes (growth, death, foraging) at the individual level and then to quantify their consequences at the population level. Current work at York seeks to extend the basic ideas to more detailed models of patchy foraging, and by coupling zooplankton dynamics to larval growth models (see James et al. (2003)). The ideas are also being applied to competitive growth of plants in monocultures. Significantly, the mathematical framework is sufficiently robust that we can use the models in conjunction with real data to infer definite conclusions about the ecological processes driving the system’s behaviour. So far this validation/refinement process has elucidated the onset and nature of competition in plants (Lv and Pitchford, in preparation), and we anticipate that access to similar data from fisheries could lead to major advances in our understanding of the mechanisms governing recruitment.


In light of the stochasticity in growth and recruitment, together with the uncertainty in stock assessment, quota setting, and enforcement, can mathematical models ever tell us anything practical about how best to manage a fishery? Traditional deterministic models (e.g. the single fishery model summarised in Kot (2001)) are elegant and mathematically tractable, but lead to notions such as Maximum Sustainable Yield (MSY) which managers have learned are very dangerous – stocks managed at MSY tend to collapse unpredictably and often violently. I contend that the major flaw in the traditional approaches is not that they lack sufficient detail (spatial information, size-at-age, population age structure, fecundity etc.), nor that they ignore the complex interrelationships within and between species and their environment (i.e. fashionable “ecosystem approaches” to management), but simply that they fail to take into account the variability and uncertainty inherent in any exploited fish stock. When these stochastic processes are taken into account, we arrive at compelling arguments concerning how to manage a fishery sustainably and (potentially) profitably.

The point can be made by considering the simple fishery model (e.g. Kot (2001)) and asking how best it should be exploited. Deterministically, and assuming perfect knowledge of the system, one can and should exploit the system at MSY indefinitely. However, it is well known that any small, temporary increase in fishing above MSY leads to catastrophe; mathematically the system is being exploited perilously close to a point of bifurcation. Now suppose that the system is exploited at (say) 90% of MSY,
allowing fishers 10% leeway in their catch before MSY is exceeded. If fishers always catch within 10% of their new quota, then surely the fishery is safe? Sadly, this is most definitely NOT the case: because the system is always close to bifurcation, small fluctuations year-on-year tend to build up with catastrophic results (mathematically, the presence of an eigenvalue of near unit magnitude causes errors to decay very slowly, meaning that variability is effectively amplified as the sum of a geometric series). In other words, even such a conservative regime is doomed to failure, despite its reduced average annual catch.

As an alternative, the same model fishery could be split into two identical zones, with fishing allowed in one zone but permanently banned from the other (i.e. a marine protected area, MPA, is created) as considered in Pitchford, Codling and Psarra (in submission). Using the simplest and most transparent of assumptions for the model, we can show that deterministically the imposition of a MPA can never increase yield (i.e. the MSY of the smaller fishery + reserve is always less than MSY for the single larger fishery). However, the difference in yield is typically rather small. Therefore in a deterministic world one can argue that, whilst they will not be beneficial, MPAs may not be particularly harmful for fishery profitability.

The situation is completely reversed when one considers realistic (or indeed rather moderate) levels of stochasticity, either in recruitment, catch, or both. Extensive simulation results, across large ranges of realistic parameter values, show that overspill in recruitment generated in the protected area can lead to a sustainable exploited population in the fishery. At levels of exploitation when the single fishery is guaranteed to crash, the MPA system continues to produce a stable and sustainable yield, often very close to those predicted by traditional MSY theory.

To make these basic conclusions more robust, the model was simulated under three “management regimes”: None (simple single fishery model, constant quota); MPA (marine reserve in half of area, same quota); HCR (set quota to target catch, unless population assessed as beneath threshold – quotas adjusted annually). As noted above, None crashes long before exploitation reaches MSY, and tends to result in poor and variable yield even when the fishery does not crash. MPA (and HCR to lesser extent) provide good yields that are stable to overexploitation, and both these strategies reduce catch variability and the probability of fishery collapse.

Our basic conclusion is that both MPA and HCR policies are sustainable and profitable in a random world. The strategies counter uncertainty by buffering it in a protected zone (MPA) or reacting to it (HCR). It is well known that HCR approaches require extensive data capture on an annual basis, are prone to associated errors in data, and are liable to political interference – they are expensive to implement and prone to dilution. Being simpler, MPA approaches potentially avoid these expenses and pitfalls, an addition to possibly providing other benefits (biodiversity, ecosystem services, tourism etc.).

These conclusions are based on simple models and deserve careful scrutiny. Our investigations show the basic results to be robust to changes in: the type and source of stochasticity; small changes in the size of the MPA; the type of recruitment (Beverton-Holt, Ricker, logistic, random) and its location (local or global). In contrast, the imposition of long-term HCRs (with
quotas set for e.g. 3 years after each assessment) fare increasingly badly in a stochastic world. Where our advocacy of MPAs might be more vulnerable to criticism is in its assumptions regarding movement of individuals from MPA to fishery – we require the MPA to be large enough that individuals remain within either the MPA or the fishery except for infrequent (typically annual) movement events. This has consequences for MPA size; a tiny MPA will be entered and left many times by individuals that are otherwise exposed to fishing, and the protection and sustainability created by the MPA is lost. Essentially the MPA will protect the stock indefinitely as long as there is always an unexploited element of the stock protected by the MPA. This simple MPA can be shown to fail when this restriction breaks down (e.g. when derogations are allowed into the MPA to take a percentage of the annual catch of the fishery). More detailed individual-based models for MPAs are currently being developed to assess the role of movement, reserve size, and the connectivity of networks of reserves.

5. Conclusions

Our ‘philosophical’ conclusions are summarised by the schematic below: that rather simple and accessible mathematics and simulations of an individual-based random world can lead to significantly improved understanding of the factors governing fish populations and exploited fisheries. Deterministic models, no matter how complicated, will never achieve this.

![Schematic](image)

On a more practical level, this work shows that:

- heterogeneity and stochasticity can be characterised and quantified at the individual level;
- stochasticity is an important (and usually beneficial) factor driving recruitment;
- population-based estimates of growth rates will always be biased because of the high mortality environment in which fish live;
- management regimes which ignore uncertainty will never support sustainable high yield fisheries, but regimes which either buffer or react to uncertainty can allow sustainable high yields;
- marine protected areas may be an important ingredient in future management, but only if they are implemented on a sufficiently large scale and enforced correctly.

Much of the mathematical work summarised here is at a stage where practical collaboration with fisheries scientists, allowing the models to be challenged and validated using real world data, would be welcomed. I hope
this short account serves to highlight some of the possibilities of simple stochastic models to fisheries scientists and managers.

References
Paper 4, 2005: From ecology to fisheries management: Celtic Sea herring

Leonie Dransfeld¹,*

¹Fisheries Science Services (FSS), Marine Institute, Galway, Ireland.

*Presented as a talk at FSS mini-symposium 2005.
Email: Leonie.Dransfeld@marine.ie

1. Introduction

Celtic Sea Herring can be characterised as a fast growing early maturing stock that is at the most southern limit of its distribution. The stock is primarily recruitment driven and high fishing pressure combined with recruitment failure in recent years has led to critically low biomass levels. This paper examines the spatial and temporal distribution of the various life cycle stages from spawning and nursery grounds to feeding and migratory routes. Environmental controls such as the prevailing ocean circulation in the Celtic Sea and effect of the Celtic Sea front on recruitment and growth are discussed. Efficient management plans require the incorporation of scientific knowledge on the main forces controlling population size. Past management systems are reviewed and new approaches to the management of this stock explored.

2. The Ecology and life cycle traits of Celtic Sea Herring

The Celtic Sea herring stock is assessed in the areas of the Celtic Sea (VIIg,h,j and k) and the entrance to the Irish Sea, (VIIaS). Neighbouring stocks are the Northwest of Ireland and the Irish Sea herring stocks with whom some mixing occurs during the juvenile phase. The life history traits of Celtic Sea herring can be summarised as fast growing and fast maturing. The fish are fully mature by two rings (three year old) and although they have a life span of over 12 years, almost 90% of the population are younger than 4 ringers (ICES-HAWG, 2005). Changes in maturity over time have been documented and are related to a change in growth rates, whereby faster growing fish matured earlier (Molloy, 1979).

Clupea harengus is a determinate one-batch spawner (Blaxter and Hunter 1982) and lays demersal eggs onto spawning beds. In the Celtic Sea, spawning grounds have been well defined through a series of larval surveys, the location of ready to spawn adult fish and anecdotal information of fishermen (Breslin 1998). Information from these sources overlaps and indicates spawning grounds along the south and southwest of the Irish coast, which are consistent over years (see Fig. 1). Individual spawning beds within the spawning grounds have been mapped and consist of either gravel or flat stone (Breslin 1998).

Celtic Sea herring comprises a mixture of autumn and winter spawners with spawning occurring between late September and February. Spawning off the southeast coast occurs in autumn from October to November and in winter with peak spawning in January (Molloy 1989; Breslin 1998). Southwest herring are autumn spawners that spawn between
September and October but an extension of spawning season has been evident in recent years with an increase in winter spawning occurring around the Dingle peninsula. Herring larvae are found between October and January in close proximity to the above described spawning grounds. Larvae are transported by currents either into the Irish Sea or westwards along the south coast depending on prevailing circulation patterns (Molloy and Corten 1975; ICES 1994). During the winter months the hydrographic circulation in the Celtic Sea is primarily wind driven and the current strength and direction depends on wind intensity and direction (see Fig. 2). The dominating winds in the Celtic Sea are southwesterly winds. In case of no/low wind conditions, the currents are density driven and freshwater runoff from the river tributaries cause a coastal current to run in a westward direction. In the summer when stratification sets in between May and October the residual currents are density driven. There is an anticlockwise circulation northwards in the eastern Celtic Sea/Bristol Channel and westwards along the south coast of Ireland. Thus Celtic Sea herring larvae are transported into the Irish Sea if winds are prevailing from a southwesterly direction and they are advected westwards along the southern Irish coast when residual currents dominate. Passive larval transport into the Irish Sea further depends on the formation of the Celtic Sea front. This is a tidal mixing front that is strongest when stratification is setting in between May and October and limits the exchange between the Celtic Sea and the Irish Sea (Fig. 2).

Celtic Sea herring nursery areas are located in the bays and estuaries of the south and southwest coast (Molloy and Corten, 1975) and in the western and eastern Irish Sea. Microstructure analysis of otholiths from juveniles in Irish Sea nursery grounds indicated that a high proportion of individuals caught were winter spawners, originating from the eastern Celtic Sea. Juveniles originally from the Celtic Sea stock were found to have different growth rates depending on whether they resided in nursery areas in the Celtic Sea, the western and eastern Irish Sea. The variability in growth rate patterns occurred mainly in the larval phase and could be attributed to the different temperature regimes of the Celtic and the Irish Seas, suggesting that larval drift into the Irish Sea could be a factor in Celtic Sea recruitment variability (Brophy and Danilowicz 2002). Larval dispersal can further influence maturity at age. In the Celtic Sea faster growing individuals mature in their second year (1 winter ring) while slower growing ones spawn for the first time in their third year (2 winter rings). Pre-recruitment dispersal such as into the Irish Sea and subsequent decrease in growth rates could thus determine whether juveniles are recruited to the adult population in the second or third year (Brophy and Danilowicz 2003).

Juveniles migrate from nursery areas to spawning areas at first time of spawning. Age distribution of the stock suggests that recruitment in the Celtic Sea occurs first in the eastern part of the Celtic Sea and follows a westward movement (ICES 1994). Tagging experiments and analysis of otolith microstructure have shown that juveniles migrate from the Irish Sea to the Celtic Sea (Molloy et al. 1993; Brophy and Danilowicz 2002; Brophy and Danilowicz 2003). The juveniles are believed to originate from the Celtic Sea stock and reside in Irish Sea nursery areas. In autumn and winter they migrate back into the Celtic Sea for first time spawning (Molloy et al. 1993).
Adult migration occurs from spawning grounds to feeding grounds and vice versa in spring and autumn (Burd and Bracken 1965). Shoals congregate and move into the shallow coastal waters for spawning after which the shoals disperse into deeper offshore waters for feeding in the central Celtic Sea (Molloy 1980).

Figure 1. Schematic presentation of the life cycle of Celtic Sea and VIIj Herring.

3. Potential environmental influence

In the Celtic Sea, herring is at its most southerly distribution in the northeast Atlantic and is therefore expected to be vulnerable to environmental fluctuations. Average sea surface temperatures in the Celtic Sea have been progressively increasing in the last decades (Reynolds et al, 2002). Warm water temperatures cause a fast growing and fast maturing stock but high temperatures are likely to have a negative effect on recruitment as has been shown in other species at the southern limits of their distribution (Brander, 1998). Physical factors controlling the dispersal of larvae from the Celtic Sea into the Irish Sea such as wind driven circulation and the formation of the Celtic Sea front influence transport to nursery grounds and can therefore affect larval survival, growth rates and subsequent maturation age (Brophy and Danilowicz 2002; Brophy and Danilowicz 2003).
Fig. 2  Schematic presentation of prevailing oceanographic conditions. Fronts are: 1) the Celtic Sea front in the eastern Celtic Sea, a tidal-mixing front which limits exchange between tidally mixed water from the Irish Sea and stratified water from the Celtic Sea; and 2) the Irish shelf front to the west of the Celtic Sea, a thermohaline front separating coastal shelf water from Atlantic water. Residual currents are the Irish coastal current, a clockwise density current and the Atlantic shelf edge current. Circulation is mainly wind driven with prevailing south-easterly winds from October to May and density driven from May to October when the Celtic Sea is stratified.

4. The fishery

Historically, the fishery for Celtic Sea Herring was small and catches remained below 15 000 t in the early part of the last century until the 1950s (Burd and Bracken 1965). Sharp increases in the catches in the 1950-60s period coupled with low recruitment caused the fisheries to collapse and a closure was implemented in 1977 (Molloy 1980). The fishery was reopened in 1982 and ICES Divisions VIIa S and VIIg were joined with Division VIIj to form a new management and assessment area. A number of good year classes recruiting to the stock in the 1980s helped to rebuild the stock. In addition, management measures in the form of rotational spawning ground closures were implemented (Molloy, 1989). Since the mid nineties there has been a strong decline in biomass. Poor recruitment in the mid/late 1990s followed by stronger year classes in '99 and '00 have resulted in a high proportion of young fish in the population (ICES, 2004). Recruitment of the 2001/2002 year class was estimated as the weakest on record, resulting in an almost complete absence of what would now be three year old fish. Estimates of spawning stock biomass suggest that it is currently below Bpa and possibly even below Blim while fishing mortality is high (ICES, 2005).
The Celtic Sea Herring fishery is almost exclusively exploited by the Irish fleet, who have targeted inshore spawning aggregations in the first and fourth quarter. In recent years an increasing proportion of the catch has been taken in a summer fishery, which takes place offshore on hard fish. No internationally agreed management plan exists, but the stock is being managed by the South and West Pelagic Advisory Committee who have set up management objectives to:

- To build the stock to a level whereby it can sustain annual catches of around 20,000 t.
- In the event of the stock falling below the level at which these catches can be sustained the Committee will take appropriate rebuilding measures.
- To introduce measures to prevent landings of small and juvenile herring including closed areas, and or appropriate time closures.
- To ensure that all landings of herring should contain at least 50% of individual fish above 23 cm.
- To maintain and if necessary expand, the spawning box closures in time and area.
- To ensure that adequate scientific resources are available to assess the state of the stock.
- To participate in the collection of data and to play an active part in the stock assessment procedure.

The rotational spawning box closures are still in operation to this date. Although their effectiveness to protect spawning stock biomass is difficult to evaluate, there are some strong indications that they enhance spawning success of first time spawners. Strong recruitment was seen in 2001/2002 and this coincided with the closure of the eastern spawning box C (Mine Head). Once reopened, this cohort dominated the catches in this area.

5. Current management drawbacks and possible alternatives

The management of Celtic Sea herring has been confounded by its life traits and ecology. One of its characteristics is its highly variable recruitment, which appears to be very sensitive to environmental fluctuations. As the stock depends on few year classes, success/failure in recruitment results in large biomass fluctuations over short periods of time making the stock very unstable. A management strategy, which relies on long-term TACs puts the stock at risk of collapse, when low recruiting year classes fail to provide the biomass needed for the exploitation. In addition, current management is slow to react to the fast changing state of the stock. The TAC implementation is essentially lagging almost two years behind the scientific advice. With its productive but also highly variable nature, Celtic Sea herring is close to its southern neighbours, the sardines and anchovies, and management strategies should reflect its fast changing nature. Early detection of recruitment success or failure should be attempted before their signs become visible in the fishery. Recruitment can be measured through abundance indices of 0 and 1 ringers in survey series such as the ground fish or acoustic surveys. However signals in the Celtic Sea have shown to be noisy, possibly due to the unknown proportion of juveniles residing in the Irish Sea before recruiting to the Celtic Sea stock. In the North Sea, a
recruitment index for 0-ringer herring is derived from the abundance of large herring larvae captured by a Methot-Issac Kidd (MIK) net. This index appears to perform well and correlates with the abundance of 1 ringers in the subsequent year (ICES, 2005). Results of recruitment indices could be used to formulate early warning signs about the state of the stock, which could be coupled with harvest controls rules to allow a rapid management response to changes in the population abundance. This in turn would reduce the risk of stock depletions while ensuring fishing at the maximum sustainable yield.

References
Paper 5, 2005: Technical conservation measures in the Irish *Nephrops* fishery

Emmet H. Jackson$^1$,* & Dominic Rihan$^1$

$^1$Bord Iascaigh Mara (BIM), Dublin, Ireland.

*Presented as a talk and posters by Emmet Jackson at FSS mini-symposium 2004 and 2005. Email: jackson@bim.ie

1. Introduction

On prawn grounds, the efficiency of standard trawl gear is such that a mixed catch of target and bycatch species is expected and considered normal. Prawn grounds are typically located in areas characterised by sea-beds of mud or sandy mud, in relatively shallow water (<100m). Such low energy areas are often highly productive and provide ideal spawning and nursery conditions for a variety of species, including commercial whitefish. When targeting prawns, in these areas, a large number of juvenile fish are caught as bycatch. This, unwanted element of the catch is of twofold importance as it impedes the sorting of prawns and reduces the biomass, or volume, of juvenile fish over the grounds.

Technical Conservation Measures (TCM’s) designed to improve the selectivity of prawn trawl gear, increase the value of the retained catch and decrease the volume of discarded juvenile fish and unwanted bycatch, have become the focus of international technical research programmes.

Technical innovations tested during the last decade have included rigid and flexible sorting grids fitted into the body of the trawls and the introduction of net panels and devices within the mouth of trawls. Given the cost, time and human resources required to develop and test experimental designs and bring them from prototype to a working model with industry acceptance, the nature of technical programmes to evaluate innovative techniques and modifications is recognised as being a lengthy and arduous process, fraught with technical difficulties and practical challenges.

Impetus and support for technical conservation measures focussing on the improvement of trawl selectivity has been provided by the European Union through legislation directed at the recovery of economically endangered whitefish stocks on many traditional fishing grounds. One of the areas of particular concern in the restoration and recovery of threatened whitefish stocks has been the release and survival of juvenile commercial fish species (e.g. cod, whiting and haddock) from gear types targeting whitefish and prawns. In support of this initiative, the EU has legislated that an inclined separator panel should be fitted to prawn trawls operating in the Irish Sea to allow the separation of juvenile fish from commercially valuable prawns. In addition, the EU has funded a concerted programme of research between member states to specifically address the issue of selectivity in prawn trawls through a 23-partner project entitled NECESSITY.
2. Materials and Methods

To date, three trials have been completed in the Irish *Nephrops* fishery under the NECESSITY program, with two trials investigating different inclined separator panel designs and one trial testing a coverless trawl design. The trials conducted that tested the inclined separator panel and coverless trawl will be described here.

2.1 Separator Panel

By utilising differences in behaviour at the mouth of the trawl, the horizontal separator panel trawl was developed in the early 1980’s by scientists from Fisheries Research Services (FRS) in Scotland. The original separator trawls were designed with two separate codend and extension pieces, divided by a horizontal panel of netting and tailored to join at the upper and lower body of the trawl. *Nephrops* and finfish species such as cod and flatfish were observed to tire and fall back under the separating panel, while haddock and whiting tended to rise, passing over the panel and into the upper portion of the trawl. Early trials indicated the panel worked, but a poor reaction by the industry due to perceived difficulties in installation, repair and maintenance meant take up of the device was poor. Taking into account these differences in fish behavioural characteristics, a modification of the original horizontal separator panel concept was developed in Ireland in 2000 primarily as a means of separating spawning cod from *Nephrops* in the Irish Sea *Nephrops* fishery, allowing *Nephrops* fishermen to continue to fish in an area that was otherwise closed under the EU Cod Recovery Programme, see Fig. 1.

Previous trials by BIM in the Irish Sea suggested that this design of inclined separator panel was effective at sorting not only cod but also other whitefish such as whiting and haddock from the *Nephrops* component of the catch. The separation, while reducing sorting time for the fishermen, also had the more positive outcome of keeping commercial round fish separate from the abrasive projections of *Nephrops* and other crustaceans, dogfish and shells, which can damage retained fish and decrease their market value. Along with a reduction in the volume of bycatch and in sorting time there was also evidence that the bulk of the *Nephrops* catch remained comparable to that of a standard *Nephrops* trawl. On this basis this device was selected for testing in the Smalls *Nephrops* fishery as part of the NECESSITY project.

![Figure 1. Inclined separator panel as seen in flume tank. A: Incline panel, B: Escape opening](image)

Figure 1. Inclined separator panel as seen in flume tank. A: Incline panel, B: Escape opening
2.2 Coverless Prawn Trawl

The ‘coverless’ trawl avoids the capture of unwanted bycatch prior to it entering the trawl rather than trying to enhance escapement from within the net as with devices such as separator panels and square mesh windows. Unlike the standard trawl the ‘coverless’ trawl lacks the cover sheet, which is used to prevent fish, herded into the net from escaping. As *Nephrops* tend to fall back in the net along the bottom sheet as they enter the trawl mouth, the only reason for having an upper panel or cover sheet in the prawn trawl is to maintain the capture of fish such as haddock and whiting that tend to rise as they fall back into the trawl. The net was developed and tested by the Seafish Industry Authority (SFIA) in the UK in conjunction with Stuart Nets Ltd in Scotland.

![Figure 2. Birdseye view of nets in flume tank courtesy of SFIA. A: Standard prawn trawl, B: Coverless prawn trawl.](image)

Based on the evidence from trials conducted by SFIA and given the similarities between UK and Irish *Nephrops* fisheries in terms of gear and grounds, BIM decided to test this gear and compare the results with the findings of SFIA and BIM trials using the inclined separator panel.

During the trials all commercial fish from all the codends were measured to assess the differences in the performance of those nets fitted with the TCMs and those without. See Table 1 for summaries of the trials.
Table 1. Boat details and trial histories

<table>
<thead>
<tr>
<th>TCM Tested</th>
<th>Inclined Separator Panel</th>
<th>Coverless Trawl</th>
</tr>
</thead>
<tbody>
<tr>
<td>Vessel</td>
<td>Rose of Sharon</td>
<td>Margaret Mary</td>
</tr>
<tr>
<td>Vessel Number</td>
<td>DA17</td>
<td>DA56</td>
</tr>
<tr>
<td>Owner</td>
<td>Michael Kirwan</td>
<td>Niall Connolly</td>
</tr>
<tr>
<td>Length Overall (m)</td>
<td>23.90</td>
<td>23.00</td>
</tr>
<tr>
<td>GT</td>
<td>180.20</td>
<td>135</td>
</tr>
<tr>
<td>Engine Power (kW)</td>
<td>413</td>
<td>441</td>
</tr>
<tr>
<td>Fishing Area</td>
<td>'The Smalls' VIIg</td>
<td>'The Smalls' VIIg</td>
</tr>
<tr>
<td>Dates of Trials</td>
<td>12-17/06/04</td>
<td>08-18/06/05</td>
</tr>
<tr>
<td>Hauls Observed</td>
<td>20</td>
<td>18</td>
</tr>
<tr>
<td>Average haul length (hrs)</td>
<td>1</td>
<td>5</td>
</tr>
</tbody>
</table>

3. Results

The initial results from the two trials were positive showing a reduction in the amount of whitefish caught in the modified *Nephrops* nets.

A net incorporating an inclined separator panel shows that a much reduced catch of whitefish is possible (Fig. 3). However, while Fig. 3 seems to suggest that a large amount of cod escaped from the net, this is not true. The percentages are large but misleading as there were very little cod caught during the trials. The large number of whiting and haddock released however are representative of much larger numbers. As the incline panel leads to an opening in the net, there was a small corresponding loss of prawns.

![Figure 3. Average percentage separation for the inclined separator panel for cod, haddock, whiting and prawns.](image)

Figure 4 shows the percentage retention and selectivity parameters for whiting as calculated from one haul. The horizontal dark line shows that on entering the net the whiting have a 93% chance of escaping through the separator panel opening irrespective of length, the panel is not length...
selective, it simply splits the catch. Those remaining fish entering the 80mm codend over 22cm (L50 + SR) will not escape through the meshes of the codend. The L50% may be so low due to the fact the majority of fish caught on the trial were juveniles. Length frequency curves demonstrate the same results. Fish measured from the codends of the standard and test nets and those from the cover bag exhibit the same length ranges. The only difference is found in the frequency of the fish. On average 90% of the whiting will escape as a result of the incline panel been installed.

Figure 4. Whiting selectivity and percentage retention for one haul.

Preliminary analysis of the coverless trawl data is proving to be very positive with a noticeable difference in the amount of whitefish between the two sets of gear, see Table 2. This simple design modification has shown to consistently reduce catches of juvenile whiting and hake. Along with a reduction of over 50% for whiting and hake the coverless trawl displays a rise in the amount of commercially retained prawns by an average of 13% over all the hauls. There was no difference in the amount of cod caught between the nets. This can be explained by the behaviour of the cod in the net: they tend not to rise as they fall back into the trawl. Catch rates for haddock were very low resulting in no observable difference in volume between nets.

<table>
<thead>
<tr>
<th></th>
<th>Standard Prawn Trawl</th>
<th>Coverless Prawn Trawl</th>
<th>% Reduction</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cod</td>
<td>412</td>
<td>378</td>
<td>8</td>
</tr>
<tr>
<td>Hake</td>
<td>750</td>
<td>435</td>
<td>42</td>
</tr>
<tr>
<td>Whiting</td>
<td>6679</td>
<td>2855</td>
<td>57</td>
</tr>
</tbody>
</table>

Table 2. Total number of fish for each gear with % reduction.
4. Discussion

The results from the trials are positive and indicate that clean prawn catches, with a much-reduced catch of whitefish are possible with a separator panel fitted into the gear. Although the panel presents promising results it must also be noted that this design can easily be tampered with thus making its ability to separate whitefish from the prawn catch redundant. The panel can also be blocked by debris above the panel resulting in all whitefish been caught in the codend, or it may be blocked below the panel resulting in total loss of prawns and fish entering the net. This fact has proved problematic for the fishermen who do not want to lose any commercial catch. Feedback from fishermen who have used the device suggests that they use the device when they encounter large amounts of small or juvenile whitefish to reduce the bulk of bycatch.

The ‘coverless’ prawn trawl demonstrates a significant difference in the amount of whitefish caught when compared with a standard prawn trawl. As a gear modification the coverless trawl is uncomplicated and cannot be tampered with as easily as the separator panel. This design works well in fisheries where prawns are the dominant catch as with the Smalls and Irish Sea fisheries. In other fisheries where the catch is mixed and the fish element has a larger economic importance to the fishermen then the reduction in catches of whiting or hake may not be as acceptable. The increase in the prawn catch is a positive result and compares well with the inclined separator panel, which demonstrated a corresponding small loss of prawns with whitefish. Further testing of this design will continue in 2006 and it is hoped that similar results will be obtained.

5. Conclusions

While most the fishermen involved in these trail agree for the need to reduce bycatch in the fisheries some expressed concerns at the loss of the whitefish component in their catch and thus earnings. Others would rather have cleaner catches of Nephrops. These opposing views highlight the difficulties in designing a TCM that will meet the needs of all.

As a conservation measure, the separator trawl and the coverless trawl have the potential to contribute to the recovery of the whitefish stock in the Irish Sea and Smalls, assuming that the majority of fish released from the gear survive. In order to attempt to verify this and observe fish behaviour in relation to the gears, BIM are planning to fit an underwater camera system to monitor the performance of the separator in future trials.
Paper 6, 2005: The role of gear technology and selectivity in the advisory process

Dominic Rihan¹,*

¹Bord Iascaigh Mara (BIM), Dublin, Ireland.

*Presented as a talk at FSS mini-symposium 2005.
Email: Rihan@bim.ie

1. Introduction

ICES has recently changed how it presents advice on fisheries and the marine ecosystem to its customers. In line with its move to an "Ecosystem Approach to Marine Resource Management", it is now presenting integrated advice from each of the main advisory committees: the Advisory Committee on Fishery Management (ACFM); the Advisory Committee on Ecosystems (ACE); and the Advisory Committee the Marine Environment (ACME).

Integrated advice, including fisheries advice, is now presented on a regional ecosystem basis with respect to the following areas:

- The Barents Sea (ICES Subarea I and parts of Subarea II);
- Waters around Iceland (Division Va and parts of Subareas XII & XIV);
- Waters around the Faroe Islands (Division Vb);
- The North Sea (Subarea IV), the Skagerrak (Division IIIa) and the Eastern Channel (Division VIId);
- West of Scotland (Subarea VI);
- The Irish Sea (Division VIIa), West of Ireland (Division VIIb&c), the Celtic Sea and SW of Ireland (Divisions VIIf,g,h,j,k), the Western Channel (Division VIIe) and northern parts of the Bay of Biscay (Divisions VIIa,b,d,e);
- The Iberian Region (Division VIIc and Subareas IX and X);
- The Baltic Sea (Subdivisions 22-32); and
- Deep-water south of 62°N (water depths >200m).

In addition to these regional ecosystems, widely migrating stocks including salmon are dealt with separately:

- Widely migrating stocks (blue whing, Norwegian spring spawning herring, mackerel, horse mackerel and hake);
- Elasmobranchs; and
- North Atlantic Salmon

For fisheries advice, all stocks belonging to a given area are discussed in context with that area, together with an overview of the ecosystem, the state of the stocks and fisheries in that area. Consideration is given to mixed fisheries and where necessary “critical species”, which appear to be
overexploited or at risk of overexploitation, are highlighted and overall advice about the mixed fishery is based on the needs of these critical stocks. For those fisheries for which the mixed fisheries issues are known to be minor the advice is given on a stock basis.

While this move to area-based management seems sensible, it has become apparent that there is a tendency within certain management regimes, particularly the EU, to reduce emphasis on gear based management strategies in favour of effort control and closed areas. It is important to note, however, that the ecosystem approach does recognise the need to consider issues such as non-target catch and impacts on habitats, which quite clearly are gear dependent. There is a change within management towards a more fisheries based system of advice rather than the existing single species stock-based models. As part of this change there is a need to consider the reaction of fishermen to factors such as regulatory controls, stock and market fluctuations. A likely prerequisite of this is the integration of fishermen’s knowledge into the management process. In the future, advice will be based on catch options by species mix rather on a an individual species basis, and, as a consequence, it is important to have information on the individual fleets exploiting these mixed fisheries, how they interact with each other and their relative dynamics. A possible format for future management advice, from a gear technology perspective therefore, might include a number of important elements, such a provision of information on regulations and their effects, changes in technology use within the fishing industry and potential benthic/ecological impacts caused by changes in fishing gears or fishing patterns.

2. ICES/FAO WGFTFB

The ICES Working Group on Fishing Technology and Fish Behaviour (WGFTFB) was created in 1983. In 2002, the Food and Agriculture Organisation (FAO) joined with ICES to co-sponsor the WGFTFB, giving the working group a global mandate. The directive of the WGFTFB is to initiate and review investigations of scientists and technologists concerned with all aspects of the design, planning and testing of fishing gears used in abundance estimation, selective fishing gears used in by-catch and discard reduction; and benign environmentally fishing gears and methods used to reduce impact on bottom habitats and other non-target ecosystem components, including behavioural, statistical and capture topics.

The Working Group’s activities focuses on all measurements and observations pertaining to both scientific and commercial fishing gears, design and statistical methods and operations including benthic impacts, vessels and behaviour of fish in relation to fishing operations. The Working Group provides advice on application of these techniques to aquatic ecologists, assessment biologists, fishery managers and industry. Given this remit it would seem WGFTFB should have a strong role to play in the provision of data describing key issues in the new management approach.

2.1 What Advice can WGFTFB give?
Areas where FTFB should supply advice include the following:
Parameterised Selectivity Data
- Technical measures – selectivity of fishing gear with respect to target and non-target species
- Evaluate the effect of regulations in force, including the actual practices in the fisheries.

Fisheries based advice
- Identification of fisheries/metiers
- Monitoring and describing fisheries practices including technological change and changes in the use of technology
  - Technical aspects
  - Human aspects – adaptations to legislation, resource and market conditions
- Spatial fleet behaviour

New management instruments
- Adaptation to closed areas – redistribution of effort
- Advice on catchability (changes and sources of variability) in relation to effort management
- Technological creep
- Capacity definitions in relation to capacity control programmes

Ecosystem approach
- Impacts on non-target species (including birds and mammals)
  - Measure impact and advise on means to reduce impact
- Gear impacts on habitats
  - Measure impact and advise on means to reduce impact

Facilitating Dialogue
- Facilitating dialogue between industry and scientists
- Providing advice to the Regional Advisory Councils and relevant ICES working Groups and Study Groups

2.2 Recent Developments
Dialogue between the ICES Fisheries Technology Committee, which oversees the work of WGFTFB, with the ICES Advisory Committee on Fishery Management (ACFM) on how WGFTFB could better contribute into the advisory process began in 2003. The issue was discussed further at the WGFTFB meeting in 2004 and following this the chairman of the working group was invited to attend the Annual Meeting of Stock Assessment WG chairs (AMAWGC) in February 2005. The objective of AMAWGC is to develop the approaches and methodologies needed for WG’s to implement the new approach in order for the ACFM to have the relevant input. At this meeting it was agreed that the Assessment WG reports in future should include information on the following:
- The fisheries and their impact;
- Effect of fishing on the ecosystem
- Mixed fisheries and fisheries interactions
- Regulations in force and their effects
Factors affecting fishing operations

It was further agreed that WGFTFB be requested to provide appropriate information on the above including information required for fisheries-based forecasts, technological changes and changes in fishing practices, implementation of regulations and other fleet adaptations, ecosystem effects of fishing and potential mitigation measures. This advice should be structured on the eco-regions as defined by ICES and the assessment WG’s and should be provided in collaboration with other Working and Groups e.g. WGRED (Working Group for Regional Ecosystem Description) or WGECO (Working Group on Ecosystem Effects of Fishing Activities).

2.3 How to provide this information?

The provision of this information in the future will require a considerable amount of work in the first instance to establish structures for the provision of the information. To address this in the first instance, WGFTFB adopted a rolling Term of Reference at the 2005 Working Group meeting and allocate part of meeting in the future specifically to discuss and collate information for the provision of advice to the Assessment Working Groups. This advice will be provided in the form of working reports, which will be an extension of the current National Report system adopted by the FTFB some years back. It is also anticipated that members of the FTFB will participate in assessment Working Group meetings in the future. It is also felt that WGFTFB could act as an interface between industry and scientists, recognising that gear technologists often have better skills in communicating with the industry. At an EU level this would require WGFTFB to be involved and contribute to the RACs.

3 Conclusions

It is concluded that gear technologists have an important advisory role to play in the future assessment process. WGFTFB is seen as an ideal conduit for this information. The areas identified that input could be provided can be summarised as:

- Fleet/Metier definitions
- Monitoring technological creep
- Assisting with the development of methodologies for monitoring fishing gear impacts on habitats and review/monitor gear related remedial measures
- Facilitating dialogue between industry and scientists
- Providing data on gear selectivity for fishery-based forecasting
- Formulating of advice on technical regulations relating to gear and evaluate their effectiveness
- Advising to ICES WGs and RACs on gear development
Paper 7, 2005: The management of shellfisheries in Ireland

Oliver Tully$^1$*

$^1$Inshore Fisheries Section, BIM, New Docks Road, Galway.

*Presented as a talk at FSS mini-symposium 2005. Email: tully@bim.ie

1. Introduction

In 2005 the Department of Communications, Marine and Natural Resources established the Management Framework for Shellfisheries in Ireland which delegates the development of advice relating to the management of shellfisheries to various national and local advisory committees (Anon 2005). Although An Bord Iascaigh Mhara (BIM) is taking a lead role in implementing the policy outlined in the Framework the approach is integrated, inclusive and co-operative with an expectation that expertise from all agencies and industry will combine to formulate the best possible advice on the management of shellfisheries.

2. Shellfisheries

The Framework focuses on a specific number of species and stocks only. It is not a system for management of inshore fisheries but specifically shellfisheries wherever the Irish fishing fleet, both inside and outside of the national 12 nm limit, exploits these species. The 15 species managed under the Framework are crab (*Cancer pagurus, Maja brachydactyla, Necora puber, Carcinus meanas*), lobster (*Homarus gammarus, Palinurus elephas*), Shrimp (*Palaemon serratus*), scallop (*Pecten maximus, Aequipecten opercularis*), whelk (*Buccinum undatum*), cockles (*Cerastoderma edule*), Clams (*Ensis siliqua, Spisula spp., Tapes decussates*), periwinkle (*Littorina littorea*).

3. Overview of the Framework

There are three central Committee structures that are supported peripherally by expert working groups (Fig. 1).

1. Local Advisory Committees (LAC): These are industry led local groups who develop, where appropriate, fishery management plans for local stocks.

2. Species Advisory Groups (SAG): There are 4 SAGs one each for crab, lobster, shrimp and molluscs. These groups reflect the co-operative and inclusive approach and include representatives from state agencies and industry. The main task of the groups is to develop fisheries management plans for each species. They will integrate where necessary preferred local or regional approaches to management submitted to them by the LACs. The interaction between
local and national groups allows for flexibility in the geographic scale of management that is necessary given the range in stock structures and species biology. Scientific working groups, who collate relevant biological and fishery data, support their work.

3. Inshore Fisheries Review Group (IFRG): This group includes representatives from Seafood Policy, Sea Fisheries Control and Sea Fisheries Administration of the DCMNR, representatives from the state agencies and from SAGs. The group reviews management plans and recommendations submitted to it by the SAGs and where appropriate recommends implementation to the Marine Minister.

The Framework allows for the integration of the work of local industry led committees into national policy. Necessarily, this has to be guided, informed and conditional in order that consistent, logical and effective legislation will result. The species based approach allows for a variety of geographic scales to be sensibly incorporated into the planning process for fisheries. This integration and visible inclusion or reflection of locally led initiatives in national management planning for fisheries is a key deliverable.

The composition of the various Committees is, above all, expert based. All members have specific responsibilities and expertise whether in the area of science, administration, policy, legislation, fishing or environmental impacts.

---

**Fig. 1.** Overview of the structure of the Framework for the management of shellfisheries. Successful implementation of the framework relies on support services from science, policing, fisheries administration, policy and the views and knowledge of fishermen.
4. Delivery of management

Although the management system is advisory, and therefore without statutory authority, it is inclusive of all the statutory bodies responsible for marine fisheries and the marine environment in Ireland. Its methods of operation, furthermore, clarify the responsibilities of all concerned using particular terms of reference and guidelines. The output from the Framework therefore, although advisory, will have been produced by state and industry representatives who have a mandate and responsibility to deliver sustainable development of fisheries. Furthermore, policy guidelines for shellfisheries management developed in the Framework provide assurance that recommendations from advisory groups, consistent with international directives relating to sustainable development. The statutory authority (DCMNR) receiving management proposals and plans from this advisory Framework can therefore be assured that appropriate checks and balances with regard to national and international policies have already been integrated and that the implications of those proposals in terms of the biological, social and economic issues have been adequately considered.

5. A single species approach to management

The Framework adopts a single species approach to management for a number of reasons. Although there are significant interactions between some shellfisheries most of them are targeted single species fisheries. Examples include whelk, crab, lobster and shrimp fisheries. Although crab and lobster are captured on the same grounds by-catch rates of lobster in crab fisheries and crab in lobster fisheries are approximately 25% of targeted catch rates so even here there is an operational capacity to target single species by strategic positioning of gear over small spatial scales. By-catch species and undersized fish of the target species are mostly discarded live and the impact of the fishing gear on the seabed is not significant in trap fisheries. The main potential effect on the ecosystem of these fisheries is the removal of a high proportion of the biomass of the target species from the environment. In some cases and where the species occupies a central or key niche in the community this could alter ecosystem structure and functioning. It is, however, the objective of management and of the shellfisheries framework, to maintain stocks at levels which are sufficiently high to optimise the viability and profitability of fisheries using a suite or technical measures and input or output controls.

Although apparently contrary to current trends towards an ecosystem approach to management the single species approach adopted by the Framework reflects the operation of shellfisheries in Ireland as single species targeted fisheries. The Framework does uphold many of the objectives of a broader definition of ecosystem management in joining up the economic, social, spatial, temporal, scientific, institutional and environmental issues and institutions concerned. Its relatively low level of specification on environmental issues reflects the nature of the fisheries that it will manage but it does integrate statutory bodies with responsibility for environment into its component committees.

A species approach, rather than a regional or fixed geographic approach to management, is consistent with the need to define boundaries based on biology rather than existing administrative structures. In fisheries
management the geographic boundary that is relevant is that defined by the
distribution of the stock and of the distribution of fishing activity on that stock.
FAO guidelines for fisheries management ask "Have the management
measures developed taken into account the whole stock unit over its entire
area of stock distribution?" Boundaries other than those of the stock and its
fishery are artificial and in fact existing (land based) geographic or
institutional boundaries may act to impede integrated management. The first
step in this case is to integrate or 'join up' existing institutions and break
down barriers to communication so that the distribution of the fishery is
encompassed. There are no appropriate fixed geographic scales on which
boundaries may be drawn up for fisheries management. It simply depends
on the stock structure and this is different in each species. The species
based approach adopted by this Framework allows a species specific set of
boundaries to be identified around which the Species Advisory Groups are
constituted.

6. Interaction of Shellfisheries and other users of the marine
environment

The Committees of the Framework are not all encompassing and
inclusive of marine stakeholders i.e. they are not Coastal Zone Management
(CZM) groups but focus solely on particular fisheries only. This objective is
set within the terms of reference for the operation of the SAGs. The
deliverables for the SAGs are clear; formulation of single species
management plans for fisheries that are consistent with national policy
guidelines and EU or global directives and conventions relating to
sustainability and environmental protection. These groups provide for the
first time fora for the development of proactive management planning and
policies for shellfisheries. This sector will therefore have a clear mandate
and point of view in any future CZM forum.

7. Co-operative management and the role of science

There is a limited amount of biological data available relevant to the
management of shellfish stocks in Ireland. National fisheries monitoring
programmes have, generally, not been in place for these species and are
only now developing. This is not unusual in Europe where crustacean
fisheries assessment is poorly developed and often non-existent (Tully et al
2005a). Advice on stocks is now being developed using commercial fisheries
data and using length based yield and egg per recruit methods (Tully et al
2005b).

One of the reasons for developing co-operative management between
the state and industry (and there are many) is to enable the management
system to come to terms with the scientific uncertainty. In circumstances
where scientific answers are vague, and may even be 'commentary' rather
than analytical in content, top down management decisions are generally not
possible. In a co-operative system, decisions, which may be poorly informed,
are at least taken by consensus but more importantly there is explicit
acknowledgement of uncertainty and a realisation that monitoring of the
effects of management decisions and adapting where necessary is a
fundamental part of the management process. Although goals may be clear,
the road to achieving the objectives set out in a management plan may be punctuated by meanderings and even U-turns! Adaptive and responsive management is critical in these circumstances.

In a co-operative management system biological data and the assessment process is scrutinised closely by stakeholders and the industry may be the main provider of data. There is an increased need for transparency and understanding of the assessment process in such a system. This is a positive feature that can lead to consensus in decision-making. As reported by Hilborn (2003) increasingly complex models lack transparency, are remote from actual data and may even then give highly uncertain output. The future for fisheries is not founded on increased complexity of fisheries models but on simplifying the approach, moving towards agreed management procedures, monitoring of the stock response to particular decisions and adapting regulations as new data arises (Hilborn 2003). The co-operative model is one where this approach has the best chance of success.

References
Hilborn (2003). The state of the art in stock assessment: where we are and where we are going. Scientia Marina 67 (suppl. 1), 15-20.
Discussion from FSS Symposium 2005

The discussion session of the meeting took the form of open questions being presented to the floor by the chairman. The chairman then left the discussion to run its course, noting down the various relevant group responses. It was hoped that fishery managers would be present at this meeting and that their points of view would be incorporated. Their absence from the meeting may be reflected by the one-sided nature of the discussion. Please note that responses and comments are given in the form of a general consensus. The responses and comments presented should not be taken as indicative of the personal opinion of any of the individual participants.

Does the current European fisheries management system actually work?

To answer this question (and question 2) the group agreed that three main components should be considered: science, management and social (fishing industry and surrounding community). Initially, the group found it difficult to give a simple answer, mainly because of the observation that most European fisheries do not seem to have any properly defined objectives with regard to their management. If there are no objectives it is hard to measure the success of any particular strategy! The group agreed that a good measure of the failure of the current system is the massive levels of misreporting that is thought to now occur – this suggests that the industry has no confidence in the system (else why would they feel the need to misreport?).

The group considered some of the reasons why the system may be failing. There was general agreement that the current system was too ‘rigid’, mainly because of the inflexibility of the CFP and relative stability pact – this is something that may have to be worked around though. There were several points made about the fact that the current system seems to involve only biological scientists (looking at the status of stocks) and managers (acting on the scientific advice) and not enough consideration is made of economic or social factors – this could explain the apparent lack of confidence in the system by the fishing industry?

An example was made of North Sea Herring – this stock is apparently ‘well managed’ according to scientists (the stock is in a ‘good’ biological state) but ‘badly managed’ when considering the economic returns from this stock. There were suggestions that the problems in the system were mainly due to the decision making process by managers (and politicians) and a lack of industry input. Conversely, the point was made that many of the current problems are due to a lack of ability to tackle over-capacity and over-investment or enforce quotas and effort restrictions.

How do we initiate change in the current fisheries management process?

The first points made in response to this question queried whether it should be ‘science’ that initiates the changes in the system (given that the group considers a lot of the problems come from the management not the science component). However, several people argued that as scientists we
should not continue ‘feeding into’ a system when we know it doesn’t work. It was agreed that, as scientists the least we should do is to encourage a debate (both in the public domain and between science, management and industry) – this debate should question the current system and point out problems as well as suggesting possible solutions. It was accepted that the current system will not change overnight and any new approaches should take this into consideration. It was suggested that any new or different approaches to management could be applied initially to ‘candidate stocks’ and the success of an approach measured against defined objectives for each particular stock.

It was agreed that any new approach should have more industry involvement. Working groups (usually made up only of scientists) could include industry representatives and should definitely include more experts who understand the whole fishing process (e.g. fishing gear technicians) as well as economists – this could lead to more balanced advice being given to managers. There were suggestions that changes may be brought about through RACs (Regional Advisory Councils) made up of industry and stakeholders but there were also concerns raised that the same ‘problems’ would still exist.

A possible change discussed were ITQs (individual transferable quotas) with much stronger enforcement as a means to reduce over-capacity and over-investment. It was felt that ITQs are at least a step in the right direction towards giving fishermen ‘control of their own destiny’ (which was suggested as one of the reasons for the lack of confidence in the system by the industry).

A further new approach suggested by the group was separate identifiable ‘fishery managers’ with responsibility for ensuring that a particular fishery is ‘well-managed’ (from perspective of both scientists and industry). It was felt that one of the major problems in the current system is the lack of ‘known people’ who can facilitate communication between individuals involved with each component of the system (science, management, industry). A ‘fishery manager’ would solve this problem and give a defined responsibility to a particular person rather than ‘trusting to the system to work’.

**Is there any merit left in the ‘absolutist’ approach to fisheries science?**

This question relates to several earlier presentations at the meeting that were critical of the current scientific ‘absolutist’ approach to stock assessment – ‘absolutist’ meaning trying to know everything about the stock to make an assessment rather than just looking at an indicator such as age structure (for example). The first point made was the fact that the current scientific approach will underpin most approaches for the foreseeable future – the scientific system will not change overnight. It was agreed that the absolutist approach requires large amounts of accurate data and this may never be available for all stocks. A number of points were then made relating to the fact that the current system of funding seems to be aimed at reducing the ‘errors’ in data (so that the absolutist approach can work) by simply employing more scientists. This approach does not take into account the fact that many ‘unknown unknowns’ that can affect stocks (e.g. weather fluctuations) will never be known even with lots more funding and more
scientists. Perhaps one of the main problems is 'too many scientists and not enough fish!'? It was suggested that the current fisheries science system is far too reactive (due to the absolutist approach) and it should be more investigative and consider other approaches (whether these are ever actually used or not). It was agreed that there would certainly be some merit in attempting to use a ‘relative’ or indicator approach for less valuable stocks or for stocks where data is poor, while accepting that the absolutist approach is probably necessary for highly valuable or socially important stocks (assuming there is reliable data). The final comment made relates to points made in response to earlier questions, namely that it is hard to measure the merits of any particular approach when there are no properly defined management objectives (taking into account science, economics and social factors).

**Can we broaden the scope of single species assessment and advice?**

This question relates to the proposed changes in the ICES system to take into account the effect of fisheries on ecosystems (and vice-versa) as well as fisheries interactions (mixed fisheries). The first point raised questioned the need for a system-wide change in approach – perhaps ecosystem effects need only be considered on a ‘case by case’ basis? An example was made of the possible conflicts between seaweed cultivation and shellfisheries – any approach to manage either of these needs to take into account the interactions. Several people raised the idea of having independent environmental impact evaluations completed on each fishery. This would serve a dual purpose as scientists/managers would understand ecosystem impacts of fisheries better, while fishermen would be able to sell their product as ‘eco-friendly’ (e.g. the ‘dolphin-friendly tuna’ tag that already exists) and receive higher prices. Furthermore it was suggested that precautionary advice based on considering ecosystem interactions is simply the ‘correct’ (in a moral sense) approach to take. A final cautionary point was made – many ‘ecosystem models’ have been proposed as a way of modelling all the interactions in a fishery-ecosystem. However, such a modelling approach is likely to require a huge amounts of extra data compared to the current absolutist approach for single species fishery models. Such a modelling approach would lead to ‘extreme-absolutism’ and all the likely inherent problems (already discussed by the group in relation to the absolutist approach in earlier talks and responses to question 3).

**Can the management of inshore and offshore fisheries compliment each other?**

The first response to this question was a request to redefine the term ‘inshore’. Many stocks that are considered ‘inshore’ stocks (usually shellfish) are actually based a long way from land. A more correct term would be ‘spatially specific’ as these species usually do not move large distances. We use the terms SS (spatially specific – shellfish) and NSS (non-spatially specific). Related to points made to earlier questions, it was suggested that NSS stocks suffer from not having any properly defined management objectives. Although not many SS stocks currently have properly defined management this is changing and management will be based on a regional
basis (not species basis) with clear objectives decided with stakeholder involvement. It was suggested that trying to manage highly migratory NSS stocks across boundaries (e.g. international, regional) is always likely to fail – compromises because of competing interests and negotiations, and problems with enforcement are all likely to hinder management of these stocks. As suggested in a response to question 2, a fishery manager with overall responsibility for a fishery (working ‘between’ the industry, upper management, and scientists) would go some way to solving this problem. It was suggested that the regional management structure being put into place for SS stocks would provide the ideal ‘test’ for this ‘fishery manager’ approach. A final point was made that when considering any management approach, the main stakeholders should not simply be thought of as the ‘fishing industry’. Instead there should be some awareness of the ‘shared ownership’ of the sea (and fish stocks) by ‘all of us’ and other stakeholders such as environmentalists should also be considered.
## Appendix – List of participants

The FSS mini symposium 2004 was attended by:

<table>
<thead>
<tr>
<th>Participant</th>
<th>Institution</th>
</tr>
</thead>
<tbody>
<tr>
<td>Paul Baxter</td>
<td>University of Leeds, UK</td>
</tr>
<tr>
<td>Lisa Borges*</td>
<td>University College Cork / FSS, Marine Institute, Ireland</td>
</tr>
<tr>
<td>Edward Codling</td>
<td>University College Cork / FSS, Marine Institute, Ireland</td>
</tr>
<tr>
<td>Edward Fahy</td>
<td>FSS, Marine Institute, Ireland</td>
</tr>
<tr>
<td>Emmet Jackson</td>
<td>BIM, Dublin, Ireland</td>
</tr>
<tr>
<td>Graham Johnston</td>
<td>FSS, Marine Institute, Ireland</td>
</tr>
<tr>
<td>Ciaran Kelly</td>
<td>FSS, Marine Institute, Ireland</td>
</tr>
<tr>
<td>Marco Kienzle</td>
<td>FRS, Aberdeen, UK</td>
</tr>
<tr>
<td>Daniel McDonald</td>
<td>BIM, Dublin, Ireland</td>
</tr>
<tr>
<td>Phil McGinnity</td>
<td>FSS, Marine Institute, Ireland</td>
</tr>
<tr>
<td>Denise O'Brien</td>
<td>FSS, Marine Institute, Ireland</td>
</tr>
<tr>
<td>Jon Pitchford</td>
<td>University of York, UK</td>
</tr>
<tr>
<td>Gavin Power</td>
<td>GMIT, Galway, Ireland</td>
</tr>
<tr>
<td>Steve Simpson</td>
<td>University of Edinburgh, UK</td>
</tr>
<tr>
<td>Olliver Tully</td>
<td>BIM, Galway, Ireland</td>
</tr>
</tbody>
</table>

*Now at RIVO, Netherlands.

The FSS mini symposium 2005 was attended by:

<table>
<thead>
<tr>
<th>Participant</th>
<th>Institution</th>
</tr>
</thead>
<tbody>
<tr>
<td>Silvana Acevedo</td>
<td>NUIG, Galway, Ireland</td>
</tr>
<tr>
<td>John Boyd</td>
<td>FSS, Marine Institute, Ireland</td>
</tr>
<tr>
<td>Edward Codling</td>
<td>University College Cork / FSS, Marine Institute, Ireland</td>
</tr>
<tr>
<td>Ian Doonan</td>
<td>FSS, Marine Institute, Ireland</td>
</tr>
<tr>
<td>Leonie Dransfeld</td>
<td>FSS, Marine Institute, Ireland</td>
</tr>
<tr>
<td>Antonio Hervas</td>
<td>BIM, Galway, Ireland</td>
</tr>
<tr>
<td>Emmet Jackson</td>
<td>BIM, Dublin, Ireland</td>
</tr>
<tr>
<td>Graham Johnston</td>
<td>FSS, Marine Institute, Ireland</td>
</tr>
<tr>
<td>Ciaran Kelly</td>
<td>FSS, Marine Institute, Ireland</td>
</tr>
<tr>
<td>Eoghan Kelly</td>
<td>BIM, Galway, Ireland</td>
</tr>
<tr>
<td>Glenn Nolan</td>
<td>OSS, Marine Institute, Ireland</td>
</tr>
<tr>
<td>Jon Pitchford</td>
<td>University of York, UK</td>
</tr>
<tr>
<td>Dominic Rihan</td>
<td>BIM, Dublin, Ireland</td>
</tr>
<tr>
<td>Emer Rogan</td>
<td>University College Cork, Ireland</td>
</tr>
<tr>
<td>Olliver Tully</td>
<td>BIM, Galway, Ireland</td>
</tr>
</tbody>
</table>