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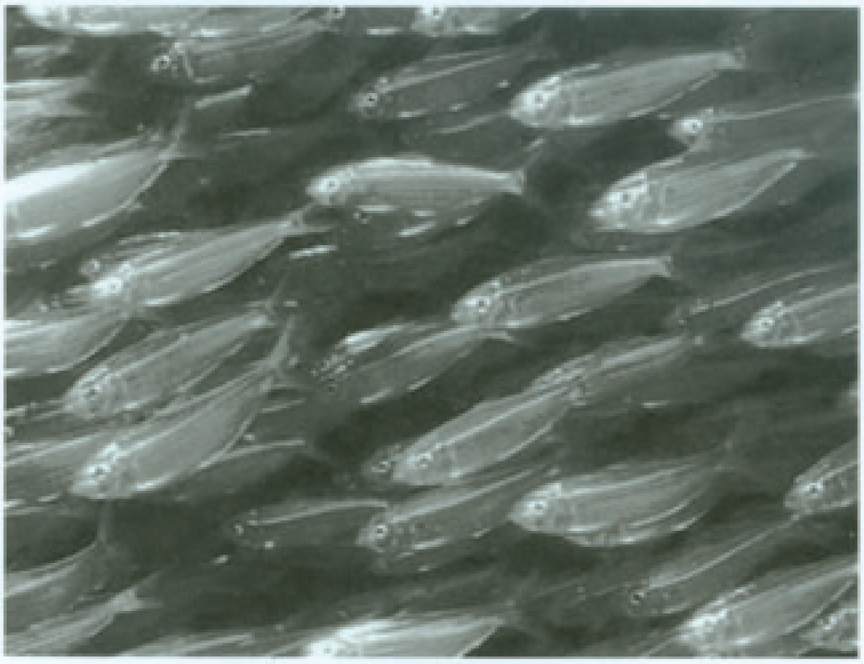
Climate Change and the Economics of the World's Fisheries

EXAMPLES OF SMALL PELAGIC STOCKS



Edited by
Rögnvaldur Hannesson, Manuel Barange, Samuel F. Herrick Jr

NEW HORIZONS IN ENVIRONMENTAL ECONOMICS

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WALLACE E. OATES
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GLOBEC

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Examples of Small Pelagic Stocks

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Climate change and the economics of the world's fisheries: an introduction

Manuel Barange, Rögnvaldur Hannesson and Samuel F. Herrick Jr

When the public, media and politicians talk about climate change, they mostly have in mind changes in weather patterns, impacts on agricultural production and disruptions in supporting energy systems. Yet climate change will affect, and in some cases already is affecting, the dynamics of all components of the earth's system. Climate change is not an isolated phenomenon, but just one dimension of an increasingly evident human imprint on the earth at a global level, which affects climate, land use, resource exploitation and pollution, among other issues. Many of these issues are linked through positive feedbacks that increase their impacts. Therefore, we often refer to 'global change' rather than 'climate change', in acknowledgement of the links between the physical, chemical and biological systems that regulate the earth and the social systems that it supports.

Global change research in the marine ecosystem is a fairly new scientific discipline. Some of its components, like eutrophication and overfishing, are well known, while others, such as climate impacts, are poorly understood. However, there is increasing recognition that global change is affecting the ecological structure and functioning of the marine ecosystem, and the goods and services it provides, in similar ways to the better known terrestrial ecosystems (Barange, 2002).

The Global Ocean Ecosystem Dynamics programme (GLOBEC) was created in 1999 by the Scientific Committee on Oceanic Research (SCOR), the Intergovernmental Oceanographic Commission of UNESCO (IOC) and the International Geosphere-Biosphere Programme (IGBP), to understand how global change will affect the abundance, diversity and productivity of marine populations. This understanding is essential if we are to manage fish and shellfish populations effectively during this period of increased human impact and dependence on these resources.

GLOBEC has implemented a number of national, multinational and regional programmes in pursuit of its objectives (GLOBEC, 1999). One such regional effort is the Small Pelagic Fish and Climate Change Programme (SPACC). SPACC was tasked with the long-range goal of forecasting how changes in ocean climate would alter the productivity of small pelagic fish populations (anchovy, sardine, herring, and so on) in key areas of the world's oceans. Small pelagic fish were selected because they constitute about 30 per cent of the world's fish catch, have a global distribution, and are characterized by dramatic abundance fluctuations in response to ocean climate. Some of these fluctuations are synchronic in nature (Schwartzlose *et al.*, 1999). One of the intentions of SPACC is to assess how climate variability and change will affect the economics of small pelagic fisheries.

Until now, research on the economic implications of climate change on fisheries has been limited and fragmented. In general, countries adjust to changes in the abundance of pelagic fish, regardless of the cause, at highly variable time scales and in an uncoordinated manner. It therefore seemed appropriate for SPACC to convene a workshop to investigate what has been learned from the economic consequences of these variations and adjustments in the recent past, with the objective of taking on board some lessons on how to adapt and respond to future climate changes, and perhaps also to set the research agenda to be followed.

The workshop took place in Portsmouth, hosted by the Centre for the Economics and Management of Aquatic Resources (CEMARE) of the University of Portsmouth, UK. Those involved in organizing the workshop quickly realized that little research had been undertaken on this topic. This could hardly be due to widespread indifference about climate change, but rather because of the great uncertainty regarding the predictability of such effects, in comparison with pressing shorter-term issues such as fishing effort controls or stock recovery plans, for example. People could be understandably reluctant to address unpredictable events whose consequences were also equally unpredictable. Yet the participants soon learned that climate change and climate variability are two sides of the same coin. Climate variability has already had major economic consequences, and the interest in dealing with it and avoiding serious economic consequences is the same as that which requires us to understand climate change impacts.

This volume groups ten case studies that range from historical fluctuations of Atlanto-Scandian herring and their impacts, to management adaptations to possible regime shifts; from the differential consequences of pelagic fisheries collapses in Southeast Asia to the globalized nature of fishmeal markets. The case studies are complementary and yet self-standing, highlighting the need for a more coordinated assessment of impacts, and calling for more focused research. While this volume may not provide detailed solutions to global problems of growing concern, it aims to enthuse practitioners to embark on research in an area intimately linked to the sustainability of our marine resources at a time when pressures on them appear to be greater than ever.

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1. Global warming, small pelagic fisheries and risk

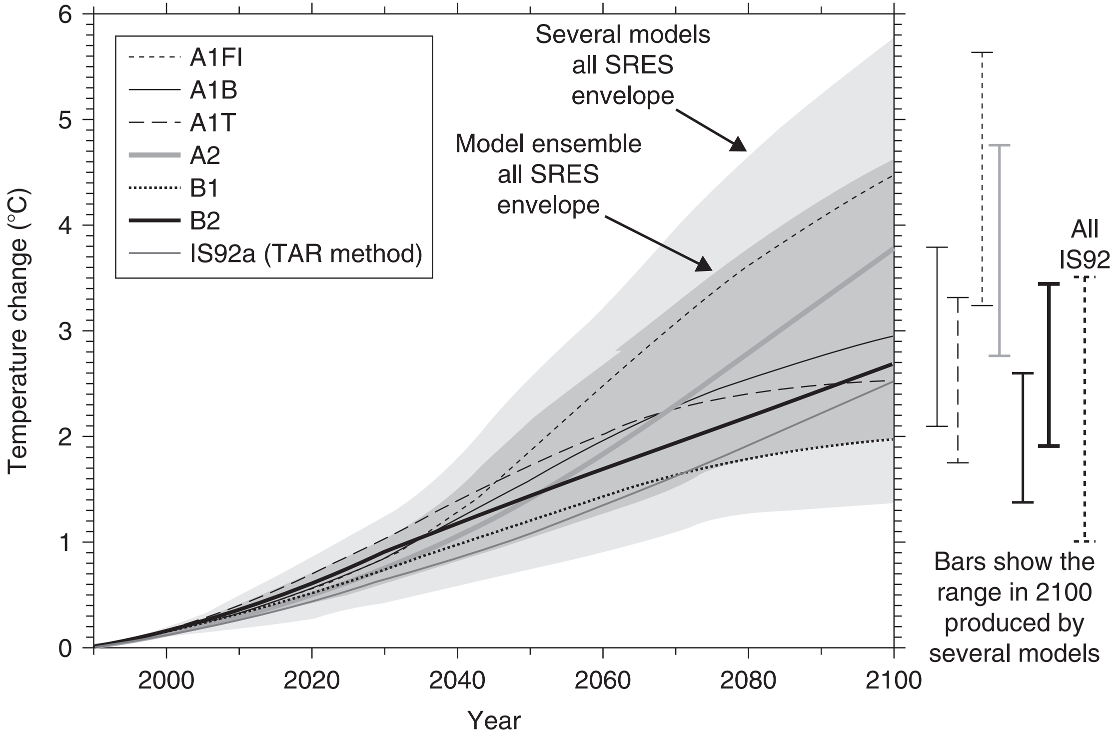
Ragnar Arnason

INTRODUCTION

Citing the huge increase in the global emission of so-called greenhouse gases[1] and their accumulation in the atmosphere, meteorologists have for some time predicted substantial global warming, as well as other climate changes. These predictions have more recently been supported by the calculations of several large-scale meteorological models.[2] Such models broadly agree that there will be warming of the earth's atmosphere during the current century and beyond. However, they differ in their prediction of future temperature increase. Thus, by the year 2100, the various models predict global temperature rises ranging from 2–4.5°C relative to the base year of 1990. Taking into account the confidence intervals presented by the model builders, the likely range of temperature increase is between 1.5 and 6°C for the world as a whole. These predictions are illustrated in Figure 1.1, which was obtained from the IPCC (Intergovernmental Panel on Climate Change) home page (IPCC, 2003).

Observed rises in global temperatures have so far been in reasonable accord with these predictions. It should be noted, however, that the number of observations that can be compared with these predictions is as yet small. Moreover, the years for which rapid, sustained rises in temperature are predicted (see Figure 1.1) are yet to arrive. Therefore, these global warming predictions are at this time largely unsupported by experience.

According to the global warming models, temperature changes around the world will be uneven. Some regions will hardly warm up at all – they may even get colder – while others will warm up substantially more than the global average. Thus, for instance, the models predict that the temperature rise in Arctic and sub-Arctic regions will substantially exceed the global increase. This applies especially in the high Arctic, where the ice cover is expected to diminish substantially, with the effect that the surface absorption of solar radiation will greatly increase. Farther south, partly because of the effects of melting ice and possible changes in ocean currents, the situation is

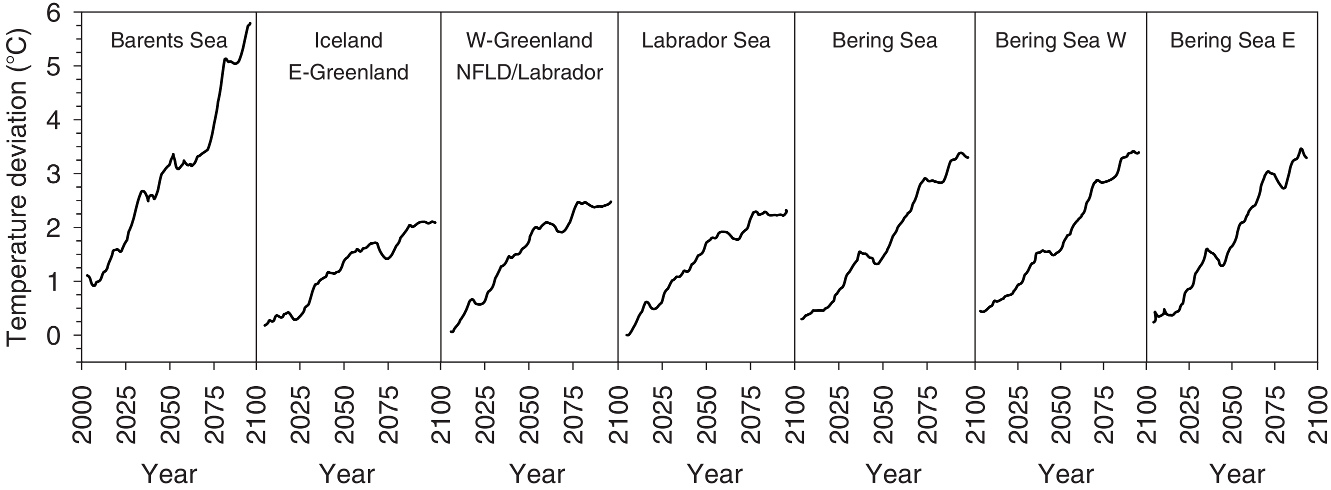


Source: IPCC, 2003.

Figure 1.1 Predicted temperature increases much less clear. In certain sub-Arctic ocean areas, where many of the world's most valuable fish stocks reside, it may be that ocean temperature will rise little or not at all. Figure 1.2 shows the predicted temperature rises for the various Arctic and sub-Arctic ocean regions. Thus, it appears that predictions of global warming remain highly uncertain. This applies on a global scale, where the subjective confidence interval (see Figure 1.1) is about the same as the mean prediction (a temperature increase of approximately 3°C by 2100). The uncertainty is substantially greater at smaller geographical scales, that is, regions.

When it comes to the impact of global warming on fish stocks and fish catchability, the uncertainty is further increased. There are several reasons for this. First, as discussed above, there is great uncertainty regarding the extent, speed and regional incidence of global warming. Second, fisheries depend very much on local conditions: upwelling, mixing of water masses, water salinity, water oxygenation, currents, ice formation and melting, and so on. Temperature is only one of the factors influencing fish stocks. On the other hand it is well known that changed temperatures have an effect on all these other hydrographical factors. For instance, it is thought likely that global warming will alter the intensity and possibly also the configuration of ocean currents (IPCC, 2001; ACIA, 2004) and consequently, also the most favourable geographical regions for fishing. This effect may be either large or insignificant. Some hydrological models suggest that global warming will have a major impact on the world's ocean current systems.[3] If that were the case, then there would be a correspondingly major impact on fishing conditions around the world. Unfortunately, however, it seems that existing hydrological and oceanographic knowledge is simply insufficient to predict the impacts with a reasonable degree of confidence, even on large geographical scales. Third, any changes in habitat conditions attributable to global warming will alter the conditions for the various species in the marine ecosystem in different ways. This will almost certainly give rise to a complicated and possibly drawn-out process of species adjustments and readjustments. The outcome of that process for individual species is very hard to predict. It may, for instance, be that species that experience favourable environmental changes are reduced in stock size because of a lessened supply of prey or a greater abundance of predators whose stock sizes are also affected by global warming.

It therefore follows that there is great uncertainty about the impact of global warming on commercial fish stocks, including small pelagics and, therefore, the fisheries based on them. There is simply insufficient hydrological, biological and ecosystem knowledge to map predictions of global warming, uncertain as they are, into predictions for fish stocks and fisheries with a reasonable degree of confidence (ACIA, 2004).



Source: ACIA, 2003.

Figure 1.2 Average predicted temperature rises in northern waters

We are thus faced with uncertain predictions about global warming and even greater uncertainty as to how any particular realization of global warming may influence fish stocks, their growth rates, sustainability and catchability. The question is, what is the appropriate course of action under these circumstances? Is it reasonable to respond to the uncertainty by cutting back on harvest rates? Or should we respond to the possible demise of valuable fish stocks in the future by harvesting more now? Or, since we know very little, is the best policy perhaps to do nothing?

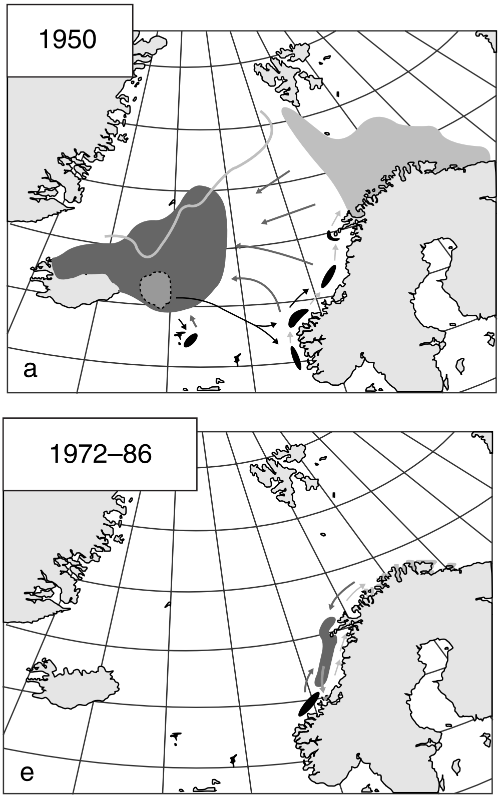
Here I address these questions. More generally I investigate the practical implications of global warming for the optimal fisheries policy. It turns out that there are no simple rules in this respect. What is optimal depends on several factors, including (i) the actual extent of global warming and its impact on fish biomass and growth, (ii) the nature of the uncertainty and in particular how risk is related to fish biomass, and (iii) the characteristics of the fishery.

While this chapter is primarily concerned with the general theoretical problem, to the extent that it deals with real fisheries it will focus on main stocks of small pelagic species in the North Atlantic, namely Atlanto-Scandian herring, blue whiting and capelin. In the Arctic Climate Assessment study (ACIA, 2003), it was found that global warming of the magnitude predicted would probably be beneficial to both Atlanto-Scandian herring and blue whiting, but detrimental to capelin. The main reason for this is that global warming is thought to expand the habitable range of Atlanto-Scandian herring and blue whiting, but to contract that of capelin, a species preferring cooler water. In the past, the geographical range of Atlanto-Scandian herring has fluctuated widely, apparently largely in response to varying environmental conditions. This is illustrated in Figure 1.3.

The chapter is organized broadly as follows. In the next section I address the nature of the problem. In the following section I seek the optimal response to increased uncertainty as well as the various responses that may be appropriate. In the third section I investigate the appropriate administrative response when the fisheries are not optimally managed. Finally, I consider the practical implications of the analysis.

BASIC MODELLING CONSIDERATIONS

Global warming will affect fish stock growth and distribution both directly by altering fish habitat, and indirectly via ecosystem adjustments. As discussed above, these changes are, given our current state of knowledge, not predictable. Because of its various impacts on human society, global warming is also likely to affect the input and output prices of the fishing



Source: Vilhjálmsson, 1997.

Figure 1.3 The range of Atlanto-Scandian herring 1950–1999

industry, and thus the economics of fishing. These changes are even less predictable than those on fish stocks. We are therefore in a situation where we confidently believe changes will take place, but are uncertain about their magnitude and even, in some cases, the direction of the change. The problem is to characterize the socially most beneficial fisheries policy under these circumstances.

Any reasonable bioeconomic fisheries model must include a biomass growth function and a fisheries profit function. For analytical purposes, these functions are typically expressed simply (see Clark, 1976; Hannesson, 1993) as:

$$G(x), \text{ where } G_{xx} < 0 \text{ and } G(0) = G(x_1) = G(x_2) = 0 \text{ for some } x_2 > x_1 \geq 0 \tag{1.1}$$

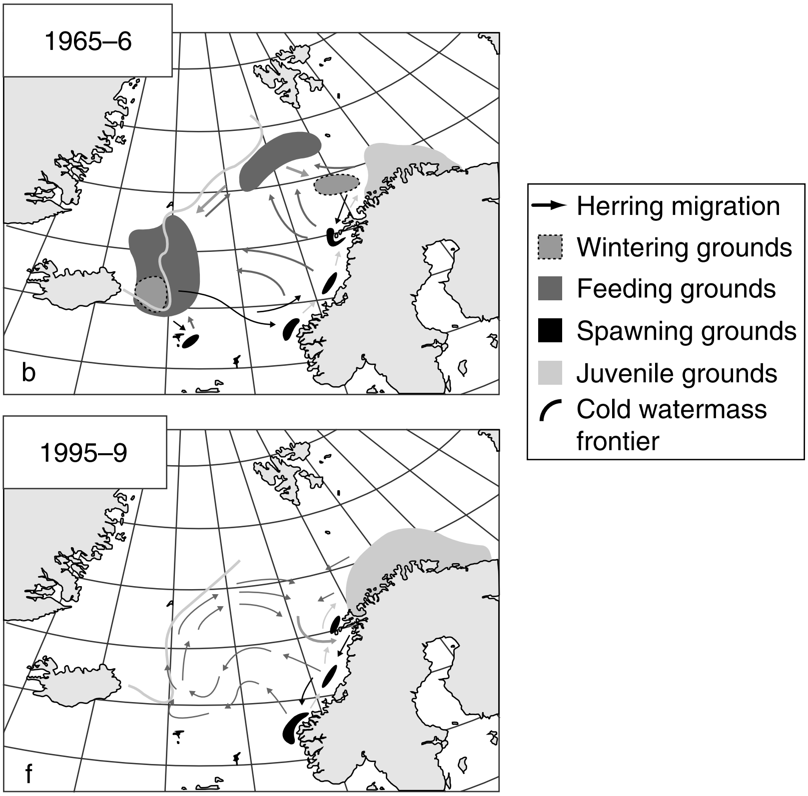


Figure 1.3 (continued)

$$\Pi(q, x), \text{ where } \Pi(0, x) \leq 0,\ \Pi_x \geq 0 \text{ and } \Pi(.,.) \text{ is concave in both arguments,} \qquad (1.2)$$

where x represents biomass and q the volume of harvest. The important thing to notice about these functions in the current context is that they are autonomous, that is they do not shift over time, and they are non-stochastic, that is they are known with certainty. Note also that the profit function does not have to be restricted to measure commercial profits. It can just as easily be seen as measuring the social benefits flowing from the fishery, including consumer surplus and, very importantly, the non-use benefits people may derive from the existing stock of fish. In what follows, however, we will proceed on the assumption that it measures commercial profits.

To capture the change and uncertainty implied by global warming, the biomass growth and profit functions may be written as

$$G(x, t, u), \tag{1.3}$$

$$\Pi(q, x, t, v), \tag{1.4}$$

where the variable t refers to time, and u and v are random variables with some (possibly time- and stock-dependent) probability distribution. The idea is that the dependence of these functions on time expresses the (foreseeable) changes that will take place, and their dependence on random variables the uncertainty about the changes. The functions in (1.3) and (1.4) express this dependence in a fairly general way. In practice, restrictions on the form of these functions may be imposed.

Under these conditions, the socially optimal (risk-neutral) fisheries policy can be derived by maximizing the expected present value of the flow of benefits from each fishery. Denoting the expectations operator by the symbol '*E*' and the rate of discount by r, the essence of this problem may be formally expressed as:

$$\begin{gathered}\underset{\{q\}}{\mathrm{Max}}\, E\left(\int_0^{\infty} \Pi(q, x, t, v)\cdot e^{-r\cdot t}dt\right) = \underset{\{q\}}{\mathrm{Max}} \int_0^{\infty}\left(\int_a^b \Pi(q, x, t, v)\cdot dv\right)\cdot e^{-r\cdot t}dt, \\ s.t.\ \ \dot{x} = G(x, t, u) - q, \\ q, x \geq 0, \\ x(0) \text{ given}\end{gathered} \tag{I}$$

A solution to this general problem is very difficult to derive. Therefore, in analytical work, simplified versions are usually employed. This is the route I follow below. In empirical work, an approximate solution may be derived by stochastic simulations (see Pascoe, 2000) or the employment of Bayesian decision theory (for example, Prato, 2000).

Most ocean fisheries are not operated optimally. In fact most of them exhibit the characteristics of common property fisheries, with resultant excessive fishing effort, overexploited fish stocks, and little or no profit or even net social benefit. For such fisheries, the solution to problem (I) is of little practical relevance. What is more important in those fisheries is to avoid irreversible or poorly reversible damage, such as species extinction or near extinction. This aspect of the problem is addressed further below.

THE NON-STOCHASTIC CASE

In this section I disregard the uncertainty associated with the impact of global warming on fisheries and proceed as if global warming alters the relevant function in a non-stochastic manner. If this is the case, the general biomass growth and economic benefit functions of the previous section are reduced to the non-autonomous form $G(x, t)$ and $\Pi(q, x, t)$.

The Optimal Fishery

The fisheries optimization problem now becomes

$$\underset{\{q\}}{\mathrm{Max}} \int_0^\infty \Pi(q, x, t) \cdot e^{-r \cdot t} dt, \tag{II}$$

$$s.t. \;\; \dot{x} = G(x, t) - q,$$

$$q, x \geq 0,$$

$$x(0) \text{ given.}$$

The interior (that is $q, x > 0$) solution to this problem is summarized by the two differential equations:

$$G_x(x, t) + \frac{\Pi_x(q, x, t)}{\Pi_q(q, x, t)} + \frac{\dot{\Pi}_q(q, x, t)}{\Pi_q(q, x, t)} = r, \tag{1.5}$$

$$\dot{x} = G(x, t) - q, \tag{1.6}$$

where $\dot{\Pi}_q \equiv d\Pi_q/dt$, so $\dot{\Pi}_q/\Pi_q$ is the rate of change in marginal benefits of harvest over time. Note that this rate of change is mathematically comparable to the rate of discount, r. For instance, it has the same dimension (t^{-1}). The only material difference is that it is variable over time. Therefore, this term can be regarded as a modification to the rate of discount at each point of time. Hence, if marginal profits of harvest are increasing, then it works as a lower rate of discount (and presumably more conservation) at that time, and vice versa.

Solving these differential equations (for the given initial condition and the appropriate transversality conditions) yields the path of biomass and harvest. In general terms these solutions may be written as:

$$x^*(t) = X(x(0); r, t), \tag{1.7}$$

$$q^*(t) = Q(x(0); r, t). \tag{1.8}$$

However, the form of these solutions and even their qualitative nature are extremely difficult to determine in general.

There is nevertheless one important practical message of general validity that can be gleaned from the above conditions. According to equations (1.5) and (1.6), the optimal fisheries policy under exogenous shifts such as global warming can be myopic. This means that it is sufficient to adapt to changes as they take place; it is not necessary to foresee the changes. This is clearly of substantial practical importance. It should be kept in mind, however, that this does not apply if one of the inequality conditions $q, x \geq 0$ becomes binding along the optimal path. If that is the case, the optimal policy has to be forward-looking. It goes without saying that the same applies with respect to other similar constraints (that is critical thresholds and irreversibility) not explicitly modelled above.

To make further headway, consider optimal equilibria. The equilibrium conditions may be written as:

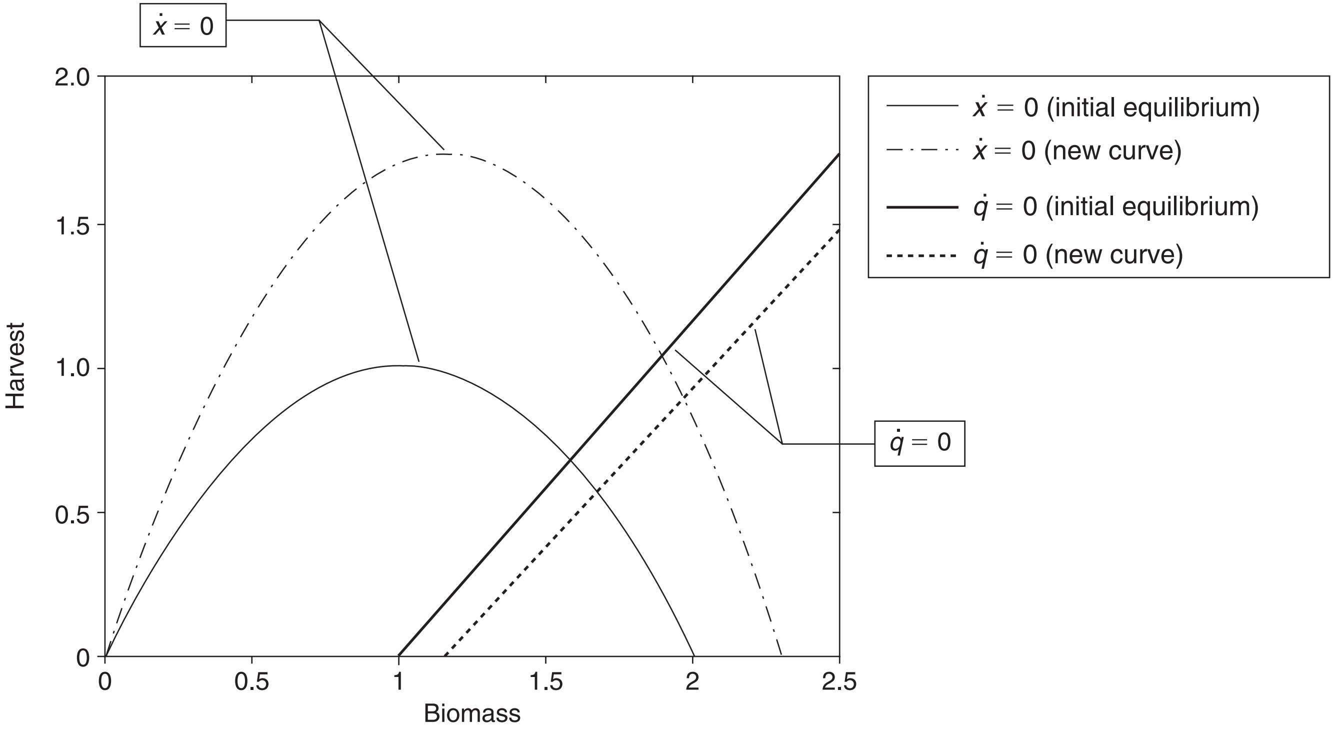
$$G_x(x, t) + A(q, x, t) = r, \tag{1.9}$$

$$G(x, t) = q, \tag{1.10}$$

where $A(q, x, t)$ is the marginal stock effect defined by $A(q, x, t) \equiv \Pi_x(q, x, t)/\Pi_q(q, x, t) > 0$. In what follows, unless otherwise stated, I assume that the optimal equilibrium is characterized by $G_x < 0$. This implies that the marginal stock effect is numerically greater than the discount factor, which is in accord with observed facts in most fisheries.[4]

Now assume that global warming has a positive effect on biomass growth and marginal biomass growth, that is G_t, $G_{xt} > 0$, and a non-negative effect on the marginal stock effect, that is $A_t \geq 0$. Then it is possible to show (Appendix 1.1) that global warming has a positive effect on the optimal equilibrium stock size, but that the impact on the optimal harvest level is uncertain. These results are readily understandable. If the fish stock becomes biologically more productive, it makes sense to take advantage of this by maintaining a higher stock size. Whether this will lead to higher optimal equilibrium harvest or not, however, depends not only on the shift in the biomass growth function, but also the shift in the marginal stock effect, $A(q, x, t)$.

This can be illustrated with the help of a phase diagram in harvest, biomass space. On the above assumptions regarding the derivatives G_t, G_{xt} and A_t, the impact of global warming is to shift the biomass and harvest equilibrium curves ($\dot{x} = 0$ and $\dot{q} = 0$), as illustrated in Figure 1.4. Here, the initial equilibrium functions are the solid curves. The new equilibrium curves after the climate shift are the dotted curves. Clearly, there is an unequivocal



Note: New curves are dotted.

Figure 1.4 Positive climate shift: shift in optimal equilibrium curves increase in equilibrium biomass following a climate shift. However, as is also clear from the diagram, whether or not there is an increase in the equilibrium harvest depends on the relative shift in the two equilibrium curves.

The effect of negative impacts of climate change, that is, those that reduce biomass growth, can be inferred from Figure 1.4 by simply taking the dotted curves as the initial curves. Obviously the impact on optimal equilibrium biomass is negative. As for positive climate change, the impact on harvest is uncertain. Therefore, contrary to what some would find intuitively obvious, it is by no means clear that climate change that is detrimental to the growth processes of fish stocks will necessarily lead to lower optimal harvest.

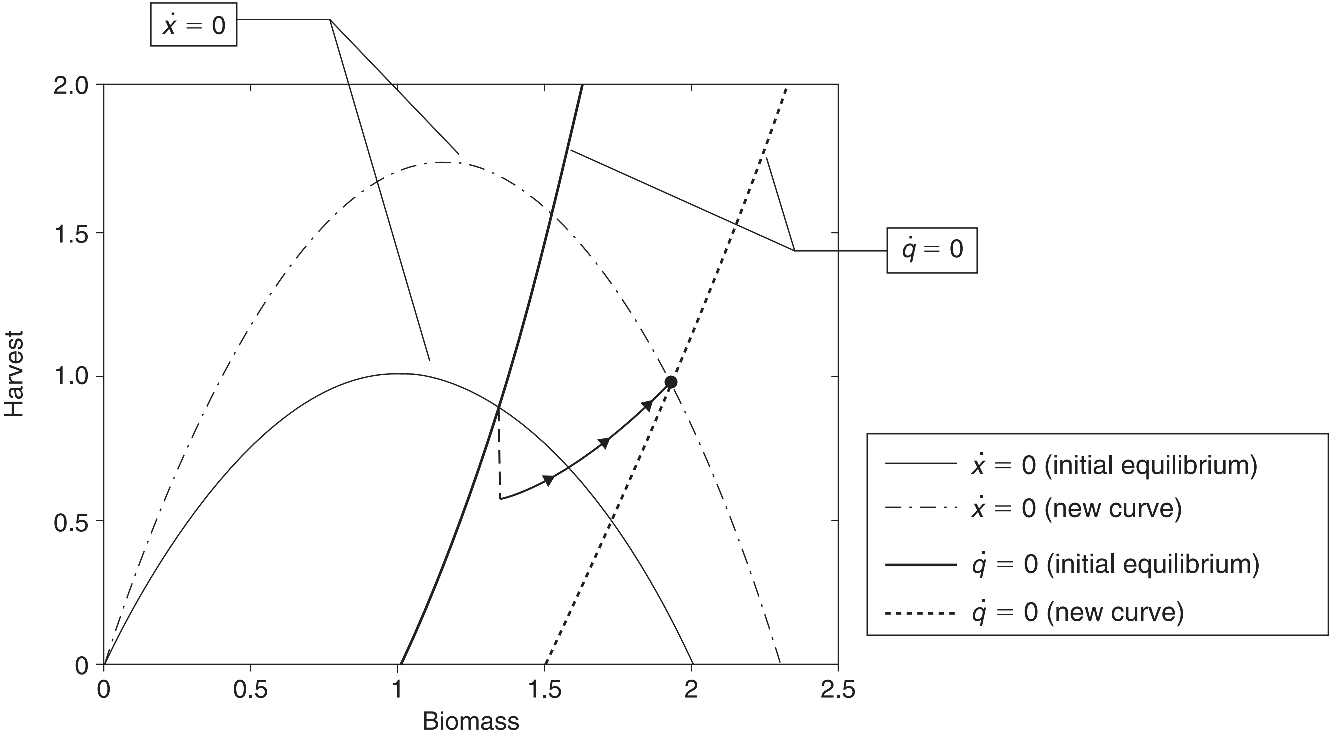
There is one particular case in which the impact of climate change on optimal equilibrium harvest is determinate. This is the case where biomass (provided it is positive) has no effect on the profit function. This may be referred to as extreme schooling (Clark, 1976). Under extreme schooling, the marginal stock effect is obviously zero and equilibrium relationship (1.9) reduces to

$$G_x(x, t) = r. \tag{1.9a}$$

It is now easy to verify (Appendix 1.1) that if global warming has a positive effect on biomass growth, that is G_t, $G_{xt} > 0$, both equilibrium biomass and harvest will unequivocally increase, and vice versa.

The optimal adjustment path of biomass and harvest from an initial equilibrium to a new one is, of course, no less important than the equilibria themselves. Unfortunately, this path is quite difficult to characterize in general. However, in a single-species case, it has generally been found (for example, Plourde, 1970; Clark, 1976) that these paths are monotonically increasing in both biomass and harvest whenever the actual biomass level is below that of optimal equilibrium and monotonically declining in both variables in the opposite case.[5]

Now, as already seen, in the case of an advantageous climate change, the initial equilibrium biomass level is sub-optimal. Therefore, the optimal policy is to select a relatively conservative harvesting rate so both biomass and harvest monotonically increase along the optimal adjustment path towards the new optimal equilibrium. However, as explained in Appendix 1.2, before embarking on this monotonic path, there will be an initial shift in harvest that may be either positive or negative. The more likely case, where the harvest drops initially before embarking on its monotonically rising path towards the new equilibrium level, is illustrated in Figure 1.5. In this case, the initial impact of an advantageous climate change on harvest is different from its long-term impact. Indeed, as illustrated in Figure 1.5,



Note: New equilibrium curves are dotted.

Figure 1.5 Possible optimal adjustment paths it may take considerable time before the harvest rate attains its pre-climate change level, if it does so at all. Needless to say, in the case of an adverse climate shift, similar effects with an opposite sign may apply. That is, harvest may be initially increased before settling down on its long-term declining path towards a new equilibrium.

This non-monotonicity effect on harvest would be particularly dramatic in the case of a linear technology (that is a profit function linear in harvest rate). In such a case, global warming having beneficial effects on biomass growth implies an initial downward shift in harvest to its minimum level (possibly zero) until the new equilibrium biomass is reached, at which point the optimal harvest jumps upward to its new and higher equilibrium level. Conversely, global warming having adverse effects on biomass growth implies an initial upward jump in harvest to the maximum level possible until the new equilibrium biomass is reached, at which point the optimal harvest shifts down to its optimal equilibrium level. Importantly, in the linear case, when there is a discrete shift in biomass growth and/or profit functions, inequality constraints will be binding. Therefore, it can no longer be optimal merely to adapt to changes as they occur. Forecasts become necessary, and optimal adjustments will begin before the shift actually takes place. This is illustrated in Figure 1.6, where a discrete

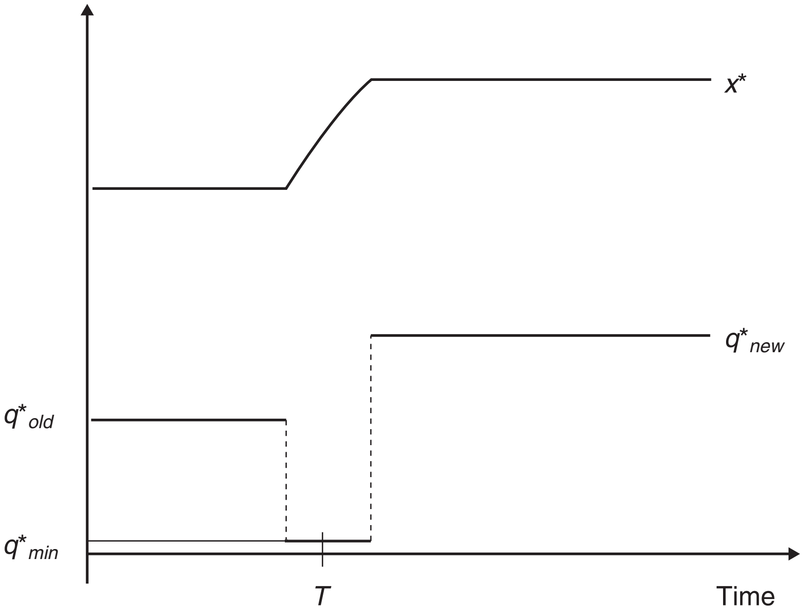


Figure 1.6 Optimal adjustments in the linear case

positive shift in the biomass growth process is assumed to take place at time T.

Of course, it is also possible in the nonlinear case to hit inequality constraints on the harvest level. These constraints do not have to be technical, as in problems (I) and (II). There may for instance be (rigid) political constraints on how much harvests can be reduced. In such a case, it will also be necessary to abandon the myopic policy and to start adjusting to the new conditions before they actually occur.

Global warming will presumably occur gradually. It is possible (although not particularly likely) that the impact on biomass growth and profit functions will be similarly gradual. In that case, it is possible that the optimal fishery can stay at or very close to equilibrium while the period of global warming and associated changes work themselves out. If that is the case, optimal paths will not involve (significant) discrete jumps as illustrated in Figures 1.5 and 1.6. Rather, they will reflect reasonably smooth monotonic movement from the initial position to the new equilibrium.

The Competitive Fishery

The expression 'competitive fishery' refers here to a fishery that is subject to the common property problem and as a result suffers from competition between its members for shares in the possible catch. In spite of various types of management measures, most ocean fisheries are still competitive in this sense.

The defining characteristic of a competitive fishery is that each fishing firm attempts to maximize its profits without regard for the shadow value of the fish stock. Within our modelling framework this may be expressed as:

$$\Pi_{q(i)}(q(i), x, t) = 0, \quad \text{for all } i = 1, 2, \ldots N, \tag{1.11}$$

where N is the total number of firms in the industry, and t, as before, reflects exogenous shifts in the profit function attributable to, for example, climate change. It follows that each firm's harvest is a function of the stock of biomass and the parameters of the situation, that is $q(i) = Q(x, t; i)$. In what follows I assume, for convenience of exposition, identical firms, that is $Q(x, t; i) = Q(x, t, j) \equiv Q(x, t)$.

The number of firms in the fishery may be taken to evolve according to the differential equation

$$\dot{N} = \Psi(\Pi(Q(x, t), x, t)), \tag{1.12}$$

where the function Ψ has the properties $\Psi_{\Pi} > 0$ and $\Psi(0) = 0$. It follows that firms move in and out of the fishery depending on whether profits are positive or negative. Equilibrium in the competitive fishery thus requires

$$\Pi(Q(x, t), x, t) = 0. \tag{1.13}$$

Finally, biomass evolves in the usual way as the difference between biomass growth and total harvest:

$$\dot{x} = G(x, t) - N \cdot Q(x, t). \tag{1.14}$$

According to equation (1.13), there cannot be any profits in this fishery and, consequently, little or no social benefits,[6] except perhaps in disequilibrium. Therefore, the only reasonable management objective is to safeguard the stock for sensible use in the future. This certainly implies the avoidance of serious stock reduction, not to mention irreversible or poorly reversible stock collapse. As shown later, global warming may cause problems of this nature.

The possible evolution of the competitive fishery over time is illustrated in the phase diagram shown in Figure 1.7. Here, the two equilibrium curves ($\dot{x} = 0$ and $\dot{N} = 0$) are drawn. Note that, to the right of the $\dot{N} = 0$ curve, the fishery enjoys profit, but to the left of the curve it suffers a loss. Similarly, underneath the $\dot{x} = 0$ curve, biomass is increasing, but above it the biomass is declining. Where these two equilibrium curves intersect, the fishery finds itself in an overall bioeconomic equilibrium. This equilibrium can be referred to as (x^e, N^e). An example of an adjustment path to this equilibrium is drawn in Figure 1.7. In this path, the fishery begins close to the virgin stock equilibrium and then evolves towards the long-term equilibrium in a cyclical manner. Note, however, that although cyclical adjustment is a common feature of models of this type (Wilen, 1976; Hannesson, 1993), the existence of cycles depends on the parameters of the problem. For some other parameter configuration, the fishery might approach equilibrium in a non-cyclical way.

In the competitive fishery, there cannot be significant social benefits in the long run. Therefore, within that management regime, the primary concern is with the probability of exhausting the biomass. From that perspective two observations need be made. First, if biomass ever falls below a certain critical level (the minimum viable biomass level, x_{min}), it is unavoidable that the stock will be exhausted. Second, for the equilibrium (x^e, N^e) to be stable, it is necessary that it occurs to the right of the maximum of the biomass equilibrium curve (see Appendix 1.3).

It should be intuitively clear (and is rigorously provable) that even when the equilibrium is stable, it may well be the case that the approach path to

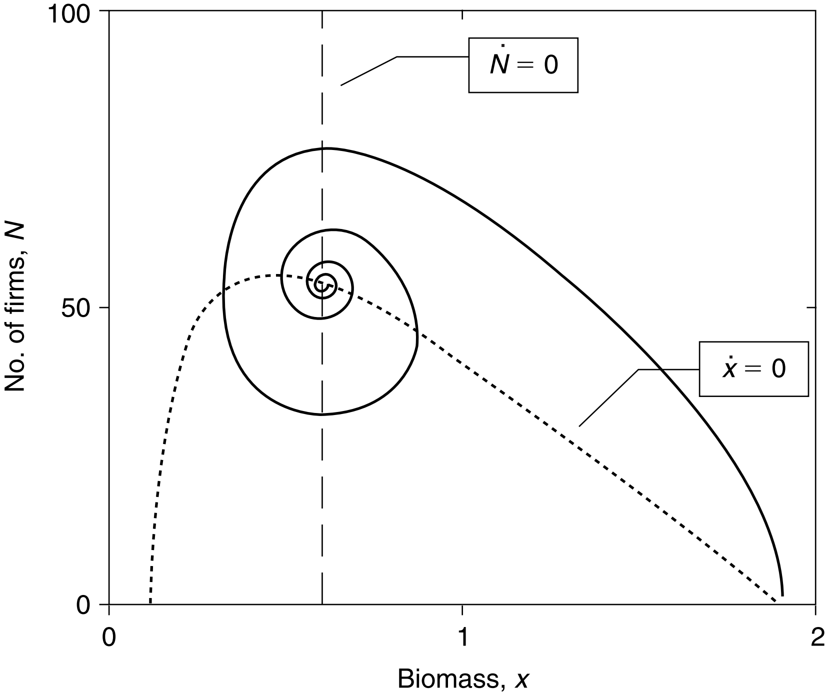


Figure 1.7 The competitive fishery: phase diagram

this equilibrium takes biomass below the minimum viable biomass level x_{min}. Clearly, the higher x_{min} and the lower x^e (while still stable), the greater the likelihood that this will occur. Similarly, if the equilibrium (x^e, N^e) is unstable, the biomass will almost certainly be exhausted in due course.

From this we deduce that if global warming does one or more of the following: (i) increases the minimum viable biomass, (ii) shifts the maximum of the biomass equilibrium curve ($\dot{x} = 0$) to the right, or (iii) shifts the economic equilibrium curve ($\dot{N} = 0$) to the left, the chances of irreversibly harming the resource will be increased. If global warming reduces the biomass growth function, it is likely to induce the first two shifts and thus increase the probability of stock collapse. If global warming increases the fisheries profit function, for example, because the price of fish rises, the probability of stock collapse also increases.

It is important to realize that small pelagics are particularly vulnerable in this respect because they usually exhibit a high degree of schooling. As a result, the profit function is fairly insensitive to stock size, except perhaps at very low stock levels (Bjørndal, 1987). This means that the economic equilibrium curve ($\dot{N}=0$) tends to be located at a low biomass level. Indeed, this biomass level may well be close to or below the critical levels discussed above (x_{min} and the maximum of the $\dot{x}=0$ curve), with the result that the fishery is highly susceptible to stock collapse even before the impacts of global warming take effect.

What are the policy implications of this? First, owing to the inherent instability of most small pelagic fisheries, those that still survive are generally already subject to controls (TAC, entry limitations and so on) that prevent them from collapsing. Within the framework of this model, this can be represented as a restriction on the number of vessels in the fishery.[7] This is illustrated in Figure 1.8, in which two biomass equilibrium curves are drawn. The upper one represents the situation before a climate shift, the lower one the biomass equilibrium curve following an adverse climate change. For convenience of exposition, the economic equilibrium curve ($\dot{N}=0$) is assumed not to shift. The restriction on the number of vessels is drawn as the straight line N_{max}.

Note that, as shown in Figure 1.8, this fishery would be dynamically unstable and, therefore, doomed to extinction in time, even before an adverse environmental change. With the upper bound on the allowable number of vessels, the fishery becomes stable before the adverse climate shift, with a locally stable equilibrium point at the intersection of the vertical ($\dot{N}=0$) line and the horizontal N_{max} line. Thus, as shown by Figure 1.8, provided the constraint on vessel numbers is imposed before biomass has fallen too much, this equilibrium will be attained.

Now consider the situation where global warming leads to a reduction in the biological productivity of the stock so that the biomass equilibrium curve is shifted to the lower one depicted in Figure 1.8. In that case, the previous N_{max} restriction will not be sufficient to maintain the fishery, which will collapse, as illustrated in Figure 1.8, possibly very quickly, unless the restrictions on vessel numbers are adjusted downwards sufficiently promptly.

THE STOCHASTIC CASE

As stated earlier, the impact of global warming on biomass growth in the future is highly uncertain. One way to model this is to represent biomass growth by a stochastic differential equation instead of the usual non-stochastic one. Following the convention in this field (Merton, 1971; Kamien and Schwartz, 1981; Pindyck, 1984) let us write this equation as:

$$dx=(G(x,t)-q)\cdot dt+\sigma(x,t)\cdot dz, \tag{1.15}$$

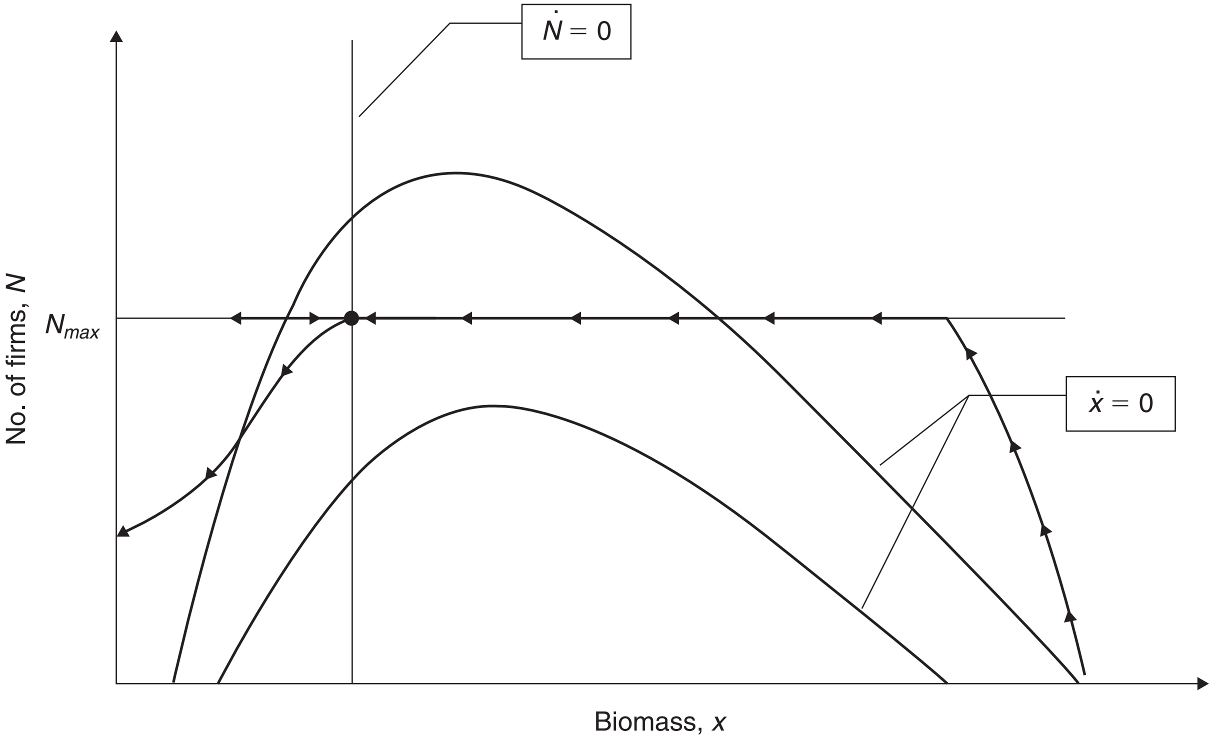


Figure 1.8 Adverse global warming leading to a stock collapse

where, as before, x refers to biomass, q to harvest and t to time. $G(x, t)-q$ is the deterministic part of biomass growth, as in the previous section. The uncertainty is represented by the additional stochastic term, $\sigma(x, t)\cdot dz$, where dz is the increment of the stochastic process, z, and $\sigma(x, t)$ is a non-stochastic term[8] often referred to as volatility in the finance literature (see Hull, 1997). z is actually the well-known Wiener process (or Brownian motion) defined by $dz=\varepsilon(t)\cdot\sqrt{dt}$, where $\varepsilon(t)$ is standard normal white noise, $\varepsilon(t)\sim N(0, 1)$. Thus, the stochastic increment, dz, is distributed as $dz \sim N(0, \mathrm{dt})$. It immediately follows that the distribution of the stochastic increment as a whole, $\sigma(x, t)\cdot dz$, is:

$$\sigma(x, t)\cdot \mathrm{dz}\sim N(0, \sigma(x, t)^2\cdot dt). \tag{1.16}$$

So, the expected value of the stochastic increment is zero, but its variance is non-zero and increases linearly with the length of the time interval under consideration. To avoid violating biological laws (zero biomass growth and negative biomass), the restriction that $\sigma(0, t)=0$ is imposed.

Expression (1.15) seems in many respects to provide a reasonable representation of the uncertainty regarding the impact of global warming on fish stock growth. The expected value of instantaneous biomass growth is the deterministic part, $G(x, t)-q$. However, there will be stochastic deviations whose variance increases the longer the time horizon (that is dt). This seems to reflect commonly held views regarding the uncertainty of global warming. Moreover, by the appropriate specification of the volatility function, $\sigma(x, t)$, as a function of time, it is possible to capture the feeling of increased uncertainty as the process of global warming advances. Note however, that expression (1.15) assumes symmetric uncertainty (normal distribution) and takes it for granted that biomass and harvest can be observed without error.

The impact of the specification in expression (1.15) may be visualized as generating a confidence interval around the deterministic biomass growth function whose width depends on the volatility $\sigma(x, t)$. This is illustrated in Figure 1.9.

The problem is to maximize the expected present value of benefits from the fishery. This may be written as

$$\underset{\{q\}}{\mathrm{Max}}\, E\left(\int_{t_0}^{\infty}\Pi(q, x, t)\cdot e^{-r\cdot t}dt\right), \tag{III}$$

$$s.t.\ dx=(G(x, t)-q)\cdot dt+\sigma(x, t)\cdot dz,$$

$$q, x\geq 0,$$

$$x(0)\text{ given.}$$

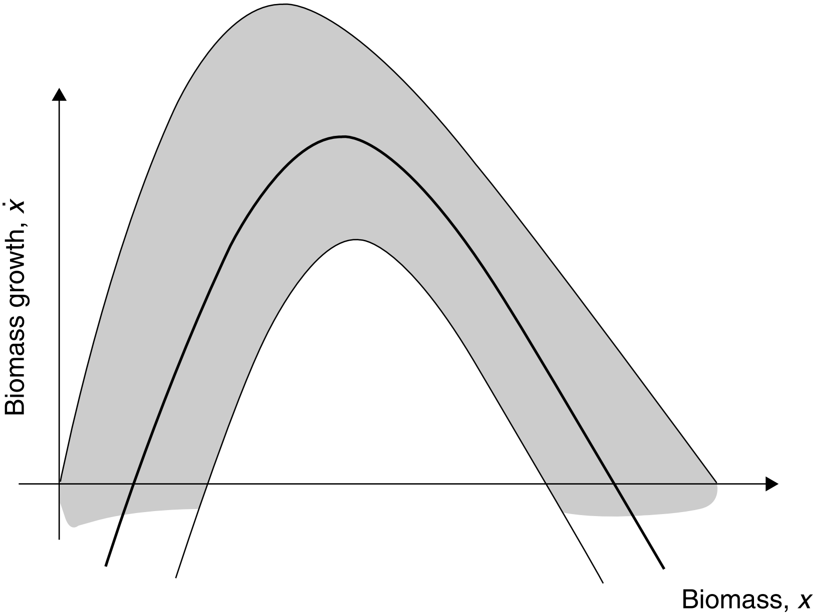


Figure 1.9 Biomass growth: confidence interval

This type of problem is usually dealt with by methods of dynamic programming (Pindyck, 1984). Following that approach, let $V(x(t_0), t_0)$ represent the maximum value of the objective functional at some initial time, t_0. More precisely

$$V(x(t_0), t_0) = \underset{\{q\}}{\mathrm{Max}}\, E\left(\int_{t_0}^{\infty} \Pi(q, x, t) \cdot e^{-r \cdot t} dt\right),$$

subject to the given constraints.

According to the principle of optimality (Bellman, 1957), this may be rewritten approximately as:

$$V(x(t_0), t_0) \simeq \underset{\{q\}}{\mathrm{Max}}\, E\left(\Pi(q, x, t)dt + V(x(t_0) + dx, t_0 + dt)\right) \quad (1.17)$$

with the equality sign applying when $dt \rightarrow 0$. Expanding the last term on the right hand side of expression (1.17), and applying Itô's stochastic calculus (Merton, 1971; Kamien and Schwartz, 1981), yields after some manipulations the following basic dynamic programming condition for solving (III).

$$-V_t(x,t) = \underset{\{q\}}{\mathrm{Max}}\left(\Pi(q,x,t) + V_x(x,t)\cdot(G(x,t)-q) + \frac{1}{2}\cdot\sigma(x,t)^2\cdot V_{xx}(x,t)\right). \tag{1.18}$$

It is worth noting that, in the special case where the profit function is autonomous (not explicitly dependent on time), the value function may be written as

$$\int_0^\infty \Pi(q,x)\cdot e^{-r\cdot t}dt = V(x)\cdot e^{-rt}$$

and the function on the left hand side of equation (1.18) reduces to $-V_t = r\cdot V(x)$. To facilitate the analysis, I adopt this simplifying assumption in what follows.[9] Note also that the expression in parentheses on the right hand side of equation (1.18) may be regarded as a generalization of the Hamiltonian function employed in optimal control theory. The first two terms are the usual Hamiltonian function (V_x equals the co-state variable or shadow value of biomass familiar in optimal control theory). The third term represents the generalization attributable to the stochasticity of the biomass growth function. This term disappears if the volatility equals zero, $\sigma = 0$, or the value function is linear in biomass. Finally, note that the complete expression on the right hand side of equation (1.18) is Pontryagin's maximum principle for the stochastic case (Pontryagin *et al.*, 1962).

Carrying out the maximization required in equation (1.18) yields the condition (for an interior solution)

$$\Pi_q(q,x) = V_x(x), \tag{1.19}$$

implying a (feedback) solution for harvest, which we may write as:

$$q^*(t) = Q(x,t).$$

Substituting this solution into equation (1.18) and differentiating with respect to x, yields:

$$\begin{aligned} r\cdot V_x &= \Pi_q\cdot Q_x + \Pi_x + V_{xx}\cdot(G(x,t)-q) + V_x\cdot(G_x - Q_x) \\ &\quad + \sigma_x\cdot\sigma\cdot V_{xx} + \frac{1}{2}\sigma^2\cdot V_{xxx} \\ &= \Pi_x + V_{xx}\cdot(G(x,t)-q) + V_x\cdot G_x + \sigma_x\cdot\sigma\cdot V_{xx} + \frac{1}{2}\sigma^2\cdot V_{xxx}. \end{aligned}$$

Now, it can be shown that $V_{xx} \cdot (G(x, t) - q) + 1/2\sigma^2 \cdot V_{xxx} = EdV_x/dt$ (Pindyck, 1984). To see this take a second order Taylor expansion of V_x

$$\begin{aligned} dV_x &= V_{xx} \cdot dx + \frac{1}{2} V_{xxx} \cdot dx^2 \\ &= V_{xx} \cdot ((G(x, t) - q) \cdot dt + \sigma(x, t) \cdot dz) \\ &\quad + \frac{1}{2} V_{xxx} \cdot ((G(x, t) - q) \cdot dt + \sigma(x, t) \cdot dz)^2. \end{aligned}$$

Taking expectations, and applying Itô's lemma, yields

$$EdV_x = V_{xx} \cdot (G(x, t) - q) \cdot dt + \frac{1}{2} \sigma^2 \cdot V_{xxx} \cdot dt.$$

Moreover, by equation (1.19), $\Pi_q(q, x) = V_x(x)$. Substituting this and rearranging, the following characterization of the optimal solution is derived:

$$\begin{aligned} &G_x(x, t) + \frac{\Pi_x(Q(x), x)}{\Pi_q(Q(x), x)} + \frac{Ed\Pi_q(Q(x), x)/dt}{\Pi_q(Q(x), x)} \\ &\quad + \sigma_x(x, t) \cdot \sigma(x, t) \cdot \frac{d\Pi_q(Q(x), x)/dx}{\Pi_q(Q(x), x)} = r. \end{aligned} \tag{1.20}$$

Comparing equation (1.20) with the corresponding non-stochastic expression (1.5) reveals that the stochastic biomass growth, as specified above, leads to two modifications of the optimality condition. First, instead of the certain $\dot{\Pi}_q/\Pi_q$ term in the non-stochastic case, we now have the expectation of this rate of change. The reason, of course, is that according to stochastic specifications, it is not known for sure what this will be. If it is known, this expected term reduces to the certain rate in expression (1.5). Second, the term $\sigma_x \cdot \sigma \cdot (d\Pi_q/dx)/\Pi_q$ is added to the optimality condition. This term is obviously zero in the non-stochastic case ($\sigma = 0$). More interestingly, it is also zero if the volatility parameter, σ, does not depend on biomass ($\sigma_x = 0$). This means that if volatility, which may be regarded as a measure of the risk associated with biomass growth, does not alter with the size of the biomass, it can be ignored in the optimal fishing rule. This makes good intuitive sense. If the risk is independent of biomass, why should biomass be adjusted to avoid it? Finally, this term is also zero if marginal benefits of harvest are independent of biomass.[10] This last observation is particularly relevant in the case of small pelagics which, as already pointed out, often exhibit extreme schooling behaviour (that is $\Pi_x \equiv 0$) so that $\Pi_{qx} = \mathrm{V}_{xx} = 0$.

It is convenient to write the last term on the left hand side of equation (1.20) as the multiple $-s^2 \cdot E(\sigma, x) \cdot \Psi(x)$, where $s = \sigma/x$ is a dimensionless quantity similar to the coefficient of variation, $E(\sigma, x)$ is the elasticity of volatility with respect to biomass, and $\Psi(x) \equiv x \cdot (-V_{xx}/V_x)$ is the coefficient of relative risk aversion. Rewriting equation (1.20) accordingly:

$$G_x + \frac{\Pi_x}{\Pi_q} + \frac{Ed\Pi_q/dt}{\Pi_q} = r + s^2 \cdot E(\sigma, x) \cdot \Psi(x). \tag{1.21}$$

Now, obviously, $s^2 > 0$. Moreover, if there is risk aversion, $\Psi(x) > 0$. Therefore, if the elasticity of volatility with respect to biomass is positive ($E(\sigma, x) > 0$), the stochastic modification works as a higher rate of discount, and vice versa. The stochastic modification, therefore, suggests a lower optimal biomass if $E(\sigma, x) > 0$, and a higher one if $E(\sigma, x) < 0$. This seems to make good sense. With risk aversion, risk is costly. Therefore, if risk increases with biomass, the optimal stochastic policy is to revise biomass downwards. If, on the other hand, risk is reduced with higher biomass, the optimal response is to increase biomass.

This, however, is not the complete story. As pointed out by Pindyck (1984), stochasticity also affects the expected value term on the left hand side of equation (1.21). Employing Itô's lemma once again, the expected value of the stochastic differential of $d\Pi_q$ is:

$$E(d\Pi_q) = \Pi_{qx} \cdot (G(x) - Q(x))dt + \frac{1}{2} \cdot \sigma^2 \cdot \Pi_{qxx} \cdot dt. \tag{1.22}$$

Obviously, the sign of this term cannot be determined without further assumptions. Therefore, the impact on the optimal biomass level may work against the impact of the stochastic modification discussed above. For instance, if there is no expected change in biomass ($G(x) - Q(x) = 0$) and Π_{qxx} is negative, the impact of this expected term is to increase the rate of extraction irrespective of the sign of $E(\sigma, x)$.

The essence of the above results can be made a little more transparent if one is willing to make the simplifying assumption that the total uncertainty of the situation can be captured by a risk function. In this case, the non-autonomous profit maximization problem can be written in the following certainty equivalent form as

$$\underset{\{q\}}{\text{Max}}\, E\left(\int_{t_0}^{\infty} (\Pi(q, x, t) - R(x, t)) \cdot e^{-r \cdot t} dt\right), \tag{IV}$$

$$s.t.\ \dot{x} = G(x, t) - q,$$

$$q, x \geq 0,$$

$$x(0) \text{ given},$$

where $R(x, t)$ is the risk function, which for mathematical convenience is assumed to be twice continuously differentiable and convex in biomass, x. The basic idea is that the function $R(x, t)$ is designed in such a way that problem (IV) is identical to problem (III). In other words, problem (IV) is assumed to be the certainty equivalent of problem (III).[11] This means that the function $R(x, t)$ somehow captures the uncertainty associated with biomass growth and global warming, as well as the attitude toward this uncertainty. Of course, such a function may not exist at all, in which case the representation in (IV) is not equivalent to that in (III). It may nevertheless be a reasonable approximation.

Solving problem (IV) yields the following two conditions corresponding to expressions (1.5) and (1.6):

$$G_x(x, t) + \frac{\Pi_x(q, x, t)}{\Pi_q(q, x, t)} + \frac{\dot{\Pi}_q(q, x, t)}{\Pi_q(q, x, t)} = r + \frac{R_x(x, t)}{\Pi_q(q, x, t)}, \qquad (1.23)$$

$$\dot{x} = G(x, t) - q. \qquad (1.24)$$

According to equation (1.23), the impact of risk can be viewed as a modification of the rate of discount – a risk premium. From equation (1.23), if the risk function is falling in biomass, $R_x < 0$, that is risk decreases with increasing biomass, the risk correction is towards a lower rate of discount and, thus, *ceteris paribus* towards higher biomass and a lower rate of harvest. Obviously, for $R_x > 0$, this risk correction works in the other direction.

Now, if global warming leads to higher risk at lower biomass, that is, $R_{xt} < 0$, this is equivalent to a higher risk premium at lower biomass and consequently higher optimal biomass, *ceteris paribus*, and presumably lower harvest rate. Other impacts of global warming on the marginal risk function will have other implications for the optimal biomass and harvest.

MAIN CONCLUSIONS

Global warming, if it materializes on the scale predicted, will substantially impact ocean conditions and, consequently, fish stock habitat. As a result, commercial fish stocks, their size, density and geographical distribution, will be affected. These changes may well have significant economic consequences. It follows that it is of some importance to respond to these changes in an optimal manner, even before they happen. In this context, it is important to be aware that there is great uncertainty concerning the magnitude, timing and regional incidence of global warming. Natural science only provides us with a prediction of a substantial global warming of some

3–3.5°C by the end of the 21st century, with a confidence interval of a similar magnitude. Moreover, owing to the complex physical and ecological relationships involved, the impact of any given global warming in this range on the various fish habitats, not to mention individual fish stocks and their growth, is for the most part unknown. Further, there is a good chance that global warming will affect fish catchability, fishing costs and fish prices in similarly unforeseen ways.

We are therefore in a situation where there is a high probability that the fish stock renewal processes, harvesting and marketing economics to which our fishing industries have grown accustomed during the 20th century will be significantly altered during the 21st. In general, however, we do not know the direction of the change, its magnitude or its timing. Even considering specific regions, the uncertainties regarding how global warming will affect the ocean habitat and the subsequent ecosystem responses preclude reliable predictions. The question is what to do in this situation.

The analysis of this chapter suggests that there are not many rules of general validity in this situation. The optimal policy appears to depend on the particulars of each fishery. Even assuming full knowledge of the impact of global warming on biomass growth and how the biomass uncertainty (stock growth volatility) depends on the size of the fish stocks, it is not possible to be sure about the impact on the optimal harvest, neither initially nor along the new adjustment path, or even in the new equilibrium. For instance, even when it is known for sure that biomass growth will be adversely affected, the optimal response will not necessarily be a more conservationist harvesting policy. In fact, as demonstrated above, the initial impact will often be to increase the harvest. The long-term harvesting policy is generally indeterminate. Obviously, general policy rules are even less available when uncertainties about global warming and its impact on the biomass growth and other crucial variables of fisheries are added.

Therefore, it appears that to be able to deduce the optimal fisheries policy under uncertain global warming, one generally needs to study the economics of the specific fishery and the nature of the uncertainty (probability distributions). With empirical information of this type, it may be possible in some cases to identify the optimal policy or at least approximate it by numerical means. In short, it appears that the main general rule regarding the optimal fisheries policy response to global warming is that there is no such rule. The appropriate response depends on each particular situation. Not only are general policy recommendations, such as more conservation, not supported by the analysis of this chapter, but the very existence of such general policy rules is refuted.

There is nevertheless one broad rule of practical use that can be derived from the analysis. It seems that in some, perhaps many cases, it is sufficient to adjust to changes as they occur. In other words, the optimal policy does not need forecasts of global warming. It is sufficient to observe and then to adjust. Only in the situation where the optimal response to changes turns out to be non-feasible because of critical thresholds, or physical or political constraints, will it be necessary to adjust to changes before they occur, implying the need for forecasts. Critical thresholds are for instance ecosystem regime shifts. Physical constraints on harvests are for instance inability to have negative harvests or exceed a certain capacity bound. Political constraints are, for example, the insurmountable need to harvest at least a certain minimum level. *A priori*, it appears that the political constraints are more likely to apply than the physical ones.

There is one specific case, possibly relevant for small pelagic fisheries, in which the optimal fisheries policy under global warming is much more clear-cut. This is the case of extreme schooling species, that is where the fisheries profit (benefit) function does not depend on biomass (except possibly at very low biomass). In such cases an adverse change in biomass growth will unequivocally lead to reduced harvesting policy, and vice versa.

The situation for competitive (that is common property) fisheries is very different from that of optimal fisheries. In competitive fisheries, profits (that is net social benefits) are generally close to zero. Therefore, what counts is to maintain the biomass, that is to avoid the risk of serious and possibly irreversible stock reduction. It follows immediately that, in such a fishery, the appropriate response to an increased risk of stock collapse is to take steps to reduce the risk. This normally implies a more conservative fisheries policy. Therefore, if global warming increases the risk of a stock collapse, which may well be the case, more conservation may be warranted.

APPENDIX 1.1: EQUILIBRIUM SHIFTS: THE OPTIMAL FISHERY

The General Non-stochastic Model

The optimal equilibrium equations, that is equations (1.9) and (1.10), are:

$$G_x(x, t) + A(q, x, t) = r,$$

$$G(x, t) = q$$

$$\text{where} \quad A(q, x, t) \equiv \frac{\Pi_x(q, x, t)}{\Pi_q(q, x, t)}.$$

The first partial derivatives of the function A are:

$$A_x = \frac{\Pi_{xx} \cdot \Pi_q - \Pi_{qx} \cdot \Pi_x}{\Pi_q^2} < 0, \quad \text{if } \Pi_{qx} \geq 0. \text{ [}\Pi\text{ increasing and jointly concave in } q \text{ and } x\text{]}$$

$$A_q = \frac{\Pi_{xq} \cdot \Pi_q - \Pi_{qq} \cdot \Pi_x}{\Pi_q^2} > 0, \quad \text{if } \Pi_{qx} \geq 0. \text{ [}\Pi\text{ increasing and jointly concave in } q \text{ and } x\text{]}$$

Conducting a comparative statics exercise on the equilibrium equations yield:

$$x_t \equiv \frac{\partial x}{\partial t} = \frac{G_{xt} + A_t + G_t \cdot A_q}{-(G_{xx} + A_x + A_q \cdot G_x)},$$

$$q_t \equiv \frac{\partial q}{\partial t} = \frac{-(G_{xx} + A_x) \cdot G_t + G_x \cdot (G_{xt} + A_t)}{-(G_{xx} + A_x + A_q \cdot G_x)}.$$

Under the assumptions made in the text, namely $G_x < 0$ and that G_t, $G_{xt} > 0$ and $A_t \geq 0$, it is now easy to verify that $x_t > 0$ and the sign of q_t indeterminate.

The Extreme Schooling Case

In this case the marginal stock effect, $A(q, x, t)$ is identically zero. Hence, the optimal equilibrium equations, that is equations (1.9a) and (1.10), are:

$$\mathrm{G}_x(x, t) = r,$$
$$\mathrm{G}(x, t) = q.$$

The comparative statics results are:

$$x_t \equiv \frac{\partial x}{\partial t} = \frac{G_{xt}}{-G_{xx}},$$

$$q_t \equiv \frac{\partial q}{\partial t} = \frac{-G_{xx} \cdot G_t + G_x \cdot G_{xt}}{-G_{xx}}.$$

For favourable global warming impacts, that is G_t, $G_{xt} > 0$, as above, both of these partial derivatives are positive (note that $G_x = r \geq 0$).

APPENDIX 1.2: COMPARATIVE DYNAMICS

According to Pontryagin's maximum principle, along the optimal path the following necessary condition must apply at all times:

$$\Pi_q(q, x, t) = \lambda,$$

where λ represents the shadow value of biomass. Initially, that is following a positive shift in the climate parameter t, but before biomass changes, the following must hold:

$$dq = \frac{(\partial\lambda/\partial t - \partial\Pi_q/\partial t)\cdot dt}{\Pi_{qq}}.$$

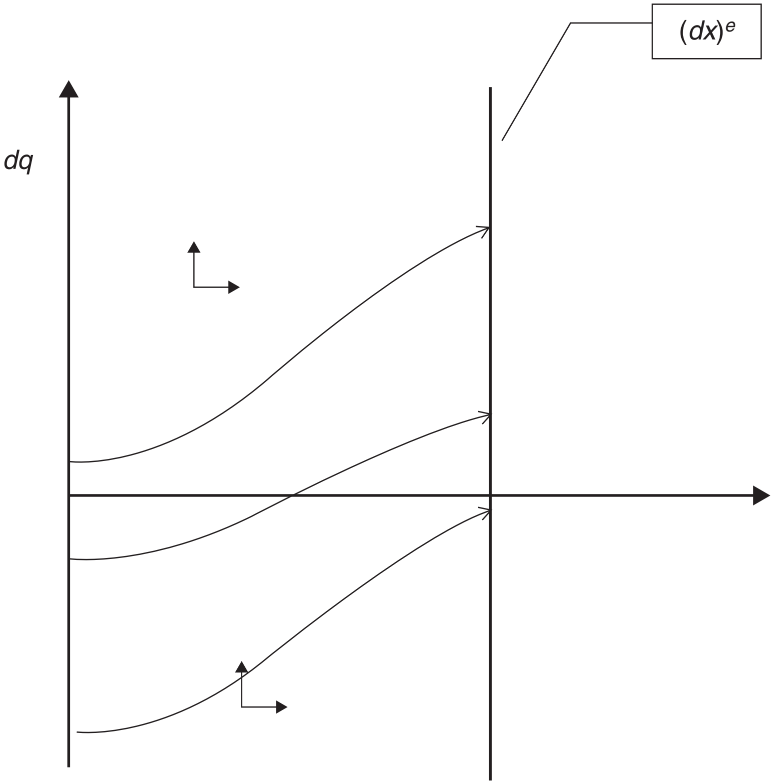


*Figure A1.1 Example of paths in (*dx, dq*) – space*

Now, a favourable climate change must increase the shadow value of the biomass (at least initially). Therefore, $\partial\lambda/\partial t > 0$. However, $\partial\Pi_q/\partial t$ can be of any sign, so $dq > 0$ if, and only if, $\partial\Pi_q/\partial t$ is not too large, that is $\partial\lambda/\partial t - \partial\Pi_q/\partial t > 0$. Thus, the initial change in harvest can be either positive or negative.

Graphically, we may represent the possibilities in (dx, dq) space, where dx and dq are deviations from the initial point. We have already established that the new biomass level is higher than the initial, so the equilibrium $dx > 0$. Moreover, owing to the nature of the optimal dynamics from below biomass equilibrium, both biomass and harvest must be growing. Therefore, we can derive the general shape of dynamic approach paths to the new equilibrium as in Figure A1.1.

While the initial shift in harvest (that is dq) can be either positive or negative, the figure makes it clear that it can only be positive if the new equilibrium harvest is substantially higher than the initial one. This verifies what we have already found, that for the initial shift in harvest to be positive, $\partial\Pi_q/\partial t$ must be sufficiently large.

APPENDIX 1.3: THE COMPETITIVE FISHERY

Instability at Maximum of the $\dot{x} = 0$ Curve

$$\dot{x} = G(x, t) - N \cdot Q(x, t)$$

$$\dot{N} = \Psi(\Pi(Q(x), x))$$

The Jacobian matrix corresponding to this system is

$$J = \begin{pmatrix} G_x - N \cdot Q_x, & -Q \\ \Psi_x, & 0 \end{pmatrix}$$

Obviously for this matrix to be negatively semidefinite (a necessary condition for stability), $G_x - N \cdot Q_x \leq 0$. However, at the maximum of the $\dot{x} = 0$ curve, $G_x - N \cdot Q_x = 0$, and it is positive to the left of this point.

NOTES

1. The main greenhouse gases are carbon dioxide, CO_2, Nitrous oxide N_2O, methane, CH_4, and the sulphur (di)oxides, SO_2 and SO_4.
2. For an overview of many of these models, see IPCC (2003).

3. Possibly weakening the Gulf Stream substantially; see ACIA (2004).
4. The most important exception would be fisheries based on strongly schooling species, where the biomass has little effect on the harvesting profit function.
5. This does not apply to the case of optimal extinction, that is when the optimal long-term biomass level is zero.
6. Note that, in a perfect market system, profits are synonymous with social benefits.
7. Note that the total catch is defined as $N \cdot Q(x, t)$. It follows that there is always an N that satisfies the constraint $TAC = N \cdot Q(x, t)$. Moreover, this also applies when the harvest of individual boats, for example $Q(x, t)$, is constrained.
8. Actually, as $\sigma(x, t)$ depends on x it is, strictly speaking, stochastic. The sentence in the text is intended to mean that for any given biomass level, x, the $\sigma(x, t)$ function is non-stochastic.
9. Note that, in the current context, this is tantamount to assuming that global warming does not affect the profit function directly, only biomass growth.
10. Recalling that $\Pi_q(Q(x), x) = V_x(x)$, this is equivalent to saying that the maximum value function is linear in biomass. Note also, as Pindyck (1984) pointed out, that the ratio $-V_{xx}/V_x$ may be regarded as an index of absolute risk aversion.
11. The classic concept of certainty equivalents was defined by Markowitz (1952) as a certain payment yielding the same utility as that of an uncertain prospect.

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2. The collapse of the Norwegian herring fisheries in the 1960s and 1970s: crisis, adaptation and recovery

Torbjørn Lorentzen and Rögnvaldur Hannesson

INTRODUCTION

Over the past 10–15 years, research on climate change and its consequences has moved to the top of the research agenda worldwide (IPCC, 2001 and ACIA, 2004). Some of this research has dealt with the economic consequences of climate change, such as how crop growth will be affected, or what will happen to the need to heat or cool buildings. However, global warming will not just affect the atmosphere and plants and animals on land, it will also affect ocean temperature and currents, and thereby plant and animal life in the sea. This, in turn, will affect growth and yield of fish stocks, both in capture fisheries and in fish farms. Whether the economic impacts will be negative or positive depends on the type of changes that take place in the ecosystem. Generally speaking, a change in climate is expected to affect growth rates, recruitment, geographic location and distribution of wild fish stocks, and growth rates in places suitable for fish farms (Stenevik and Sundby, 2004).

The effects of climate change on fish stocks will have direct or indirect consequences for the economic outcome in fisheries and aquaculture. Changes in ocean temperature have been observed on time scales of various lengths, from relatively frequent events such as *El Niño* to less frequent water temperature regime shifts. The latter are associated with extreme fluctuations in the abundance of fish stocks such as herring, even though they may not be the sole cause of the collapses in such stocks (Stenevik and Sundby, 2004).

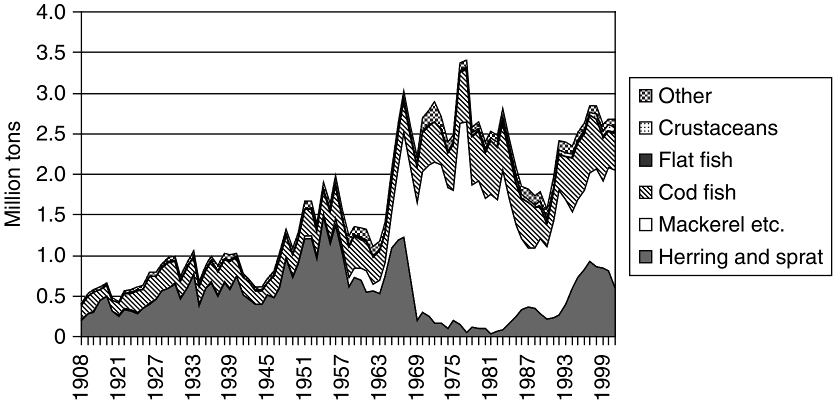
In times past, people have had to cope with major changes in fish stocks associated with climate variability in the ocean. Some of these changes, in particular the herring collapses in the 1960s and 1970s, have been of a magnitude that is probably comparable to what might result from global warming and its impacts in the oceans. As a prelude to a study of how global warming might affect Norway's fisheries, it seems reasonable to investigate the impacts of the collapses of the Atlanto-Scandian and the North Sea herring stocks in, respectively, the early 1960s and 1970s. This case study will therefore serve as a useful backdrop to an analysis of expected socioeconomic effects from significant climate change.

In this chapter we describe the most important socioeconomic impacts brought about by the disappearance of the Atlanto-Scandian herring. We also comment on some of the effects of the overexploitation of the North Sea herring, and review the changes that took place in the wake of the collapse of these stocks, in particular the changes in catches, production, exports and fleet and industry structure. We also try to shed some light on how local communities were affected, and how economic agents adapted to the sudden and dramatic change in income that took place.

Below, we briefly discuss the development of the Norwegian fish landings in the 20th century, showing the volatility of the herring fishery and how it came to be replaced by other species. We then describe the fishing industry along the Norwegian coast in the period between the two world wars, and in particular the herring fishery, limiting our description to the west coast, mainly because the herring fishery was concentrated on that part of the coast. Following that, we focus on the changes that took place in the herring fishery after the Second World War. We then deal with the crisis in the herring fishery, which started in the late 1950s, the effects and possible explanations of the crisis, and, last but not least, how the fishing industry adapted to the collapse in the herring fishery. Finally, we describe briefly how two small fishing communities, Bømlo and Fedje, evolved after the collapse in the herring fishery.

THE NORWEGIAN FISHERIES IN THE 20TH CENTURY: THE IMPACT OF HERRING

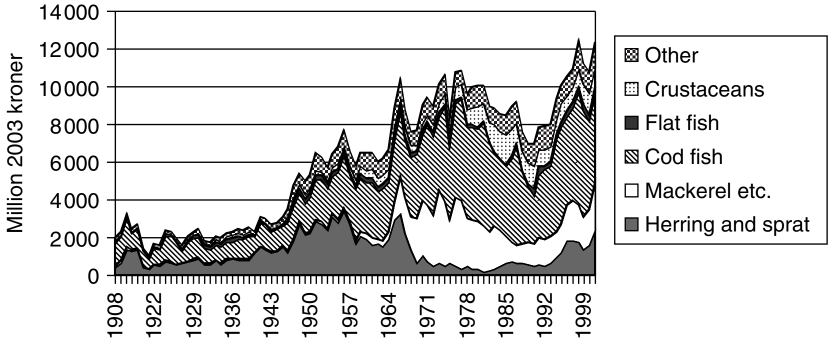
The total landings of fish and crustaceans in Norway increased about fivefold in the 20th century, from about half a million tonnes to 2.5 million tonnes (Figure 2.1). In terms of a constant value of currency (2003 kroner) the increase was even greater, some sixfold (Figure 2.2). The development was not smooth: there were major fluctuations, more so in terms of quantity than in value. Fluctuations in the catches of herring were a major, if not the major, source of fluctuations in the total catch. Herring has always been a capricious fish, a source of legends and of shifting economic fortunes. Catches increased enormously from the end of the Second World War up



Note: For a further explanation of the categories, see Appendix 2.1.

Source: Statistics Norway: Fishery Statistics.

Figure 2.1 Quantity of fish and crustaceans landed in Norway 1908–2001



Source: Statistics Norway: Fishery Statistics.

Figure 2.2 Value of fish and crustaceans landed in Norway 1908–2001

to the mid-1950s. Then there was a steep decline, followed by a rapid increase again in the 1960s. In the late 1960s the Atlanto-Scandian herring stock collapsed, and the North Sea stock met a similar fate a few years later. Thereafter, the herring fishery was a shadow of its former self until the late 1980s, when the Atlanto-Scandian stock recovered, and its trend has been upwards since then.

The rise in herring catches in the 1960s was largely due to significant advances in the purse-seine fishing technology. Productivity increased greatly with the development of the hydraulic power block to haul in the purse-seine net. This led to larger, more efficient vessels replacing the small-boat purse-seine fleet where the net was hauled in by hand. Another important development was the introduction of electronic fish-finding equipment that enabled fishers to see the herring shoals underwater instead of trying to locate them by searching for ripples on the sea surface. These innovations increased the fishing power of the fleet enormously over a short period of time, and are generally alleged to have led to the demise of the herring stocks. The shoaling behaviour makes it relatively easy to catch herring even when the stock has been severely depleted, so a diminishing profitability does not provide a strong enough feedback mechanism to stop overfishing until it is, perhaps, too late.

Not only have there been large fluctuations in the total catch, there have also been major changes in the composition of the catch. After the herring collapse in the late 1960s, there was an enormous increase in the catches of mackerel and capelin, so large in fact that the total catch volume held up well, and even increased, despite the collapse of the herring. As capelin and mackerel were used mainly for comparatively low-value meal and oil, the overall value of the catches would have fallen had it not been for an increase in the landings of the more highly-valued cod. In this case, therefore, the decline in one fishery spurred the development of another. North Sea mackerel met a fate similar to the herring, but new mackerel stocks came on the scene, and the capelin stock held up well until the late 1980s.

THE DIFFERENT TYPES OF HERRING

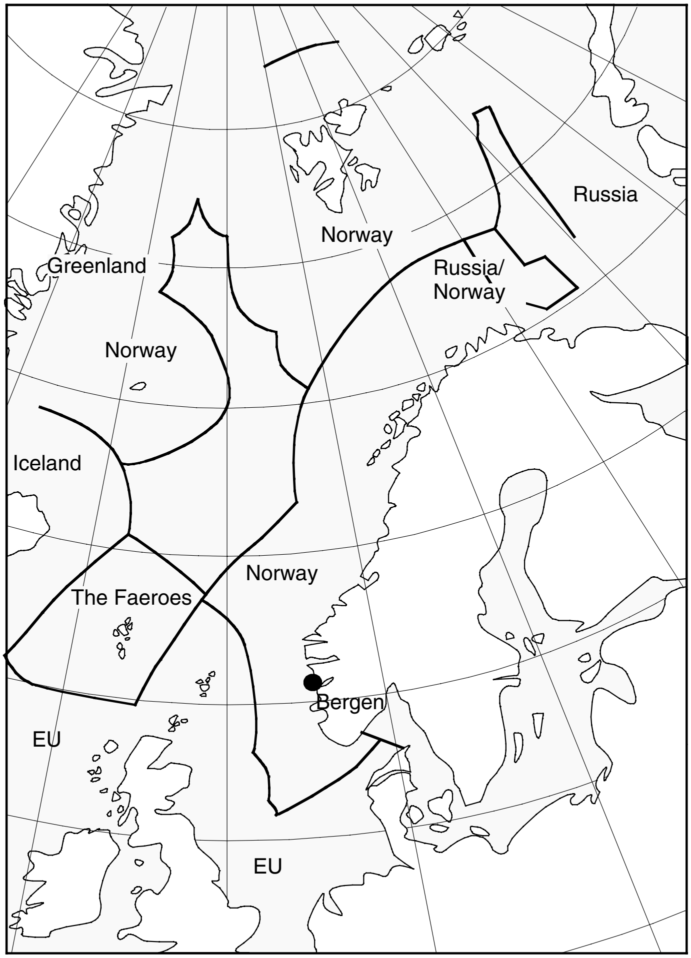
The herring in the Northeast Atlantic is classified into two separate groups or populations, respectively the Norwegian spring-spawning herring (NSSH), also called Atlanto-Scandian herring, and the North Sea herring, both *Clupea harengus L.* The criteria for classification are dependent on migration patterns and spawning areas. The fishery based on NSSH or Atlanto-Scandian herring is economically the biggest, and the NSSH fishery is further divided into subgroups depending on the size of the fish and fat content, and the location and time of the year when it is caught. To be more precise the Atlanto-Scandian herring fishery is further divided into 'Winter herring' fishery, 'Fat herring' fishery, 'Big herring' fishery, 'Spring herring' fishery, 'Small herring' fishery and 'Iceland herring' fishery. It could be confusing with so many concepts for a fishery. However, because some of these concepts will be used in the chapter, it is appropriate to give clarification. The winter herring fishery is divided respectively into the big herring fishery and the spring herring fishery, and the notions reflect the seasonality. The big herring fishery took place from November/December to February, and the spring herring fishery took place in February/March. The small herring fishery took place along most of the Norwegian coast (from Rogaland to the Finnmark counties in the north) throughout most of the year. The fat herring fishery took place from north of Stadt in the northern part of the West coast of Norway to Finnmark, and the fishery started late in the summer or at the beginning of the autumn. The main areas of operation for the fat and small herring fisheries were outside Troms and Nordland counties. The Iceland herring fishery took place off Iceland in July and ended at the beginning of September. Norwegian fishers initiated the latter fishery at the beginning of the 1900s. Fishers from Norway deployed land-seine, drift net, and later purse-seines to catch the herring.

THE HERRING FISHERY IN THE INTER-WAR PERIOD

Between World War I and World War II, the labour market in Norway was dominated by primary industries, even though the secondary and service industries were expanding. In Norwegian coastal areas, people employed in fisheries usually pursued other occupations as well, for example, farming, carpentry, and in some areas, factory work. Nevertheless, many small communities along the coast were more or less completely dependent on fishing. Figure 2.3 shows Norway and the sea areas it fishes as its own, as well as the location of Bergen, the main fishing port in Hordaland county.

The herring fisheries dominated the fishing industry in Hordaland county, and were the most important fisheries in all of Norway in terms of employment and the quantity and value of landings. Many municipalities along the coast of Hordaland were based on herring fisheries, primarily the ‘winter herring’ fishery. Table 2.1 shows the proportion of fishers in the working population, in typical fishery-dependent municipalities.

In eight municipalities, more than 60 per cent of the workforce was employed in the fishing industry, and it would be expected that changes or fluctuations in the fish stocks and landings would have significant effects on such single-industry communities. These figures only cover people directly employed in the fishing industry. Many people were employed in subcontracting activities, for example, as crew on pilot boats involved in the herring fishery, as fish buyers, as workers in the fish-processing industry, and in box- and barrel-making factories, shipyards, boatyards, engine factories and engineering workshops.



Note: The lines indicate the exclusive economic zones of Norway, Russia, Iceland, The Faeroes and the EU. The border between Russia and Norway has not yet been agreed, but there is a joint Russian/Norwegian Fisheries zone in the disputed area.

Figure 2.3 The location of the winter herring fisheries

Askøy is an example of a municipality that has few fishers (see Table 2.1). On the other hand, the indirect economic activities derived from the herring fishery were an essential part of Askøy's economy during the first part of the 20th century.

Table 2.1 Fishery-dependent municipalities in Hordaland county in the inter-war period

Fishers as a proportion of the total number of people working	Municipality
> 60%	Bømlo, Bremnes, Austevoll, Sund, Fjell, Herdla, Hjelme, Fedje
20–60%	Austrheim, Fitjar, Moster, Fjelberg, Valestrand, Sveio
5–20%	Skånevik, Kvinnherad, Tysnes, Strandvik, Os, Askøy, Manger, Hordabø, Lindås

Source: Johansen (1989).

The Winter-Herring Fishery

The fishing communities in Hordaland depended primarily on the herring fishery in winter. In the period 1920–1940 the winter herring fishery amounted to about 50 per cent of the total sales value of fish in the county. About 5–6000 fishers took part in the fishery. If we also include sprat and other herring fisheries, the herring amounted to about 70 per cent of the total value at first sale.

The west coast has had two rich herring periods: the first from 1808 to 1870, the second from 1890 to about 1960. If we include fishing activity north of Stadt, the latter period can be prolonged to about 1967. After the first rich herring period in 1870, the rural districts in the inner parts of the coastal area dropped more or less out of the fishery, so when the herring fishery reopened two decades later they did not resume their fishing activity.

In the second rich period, the herring fishery *Storsildfisket* (the big herring fishery) began along the coast of Møre/Romsdal county in November–December each year. Some decades later the same fishery began immediately after the turn of the year. Later in the season the fishery moved farther south along the coast of Sunnhordaland and Ryfylke as the centre of the *Vårsildfiske* (the spring herring fishery) in February and March. The winter herring fishery is the sum of the big herring and the spring herring fisheries. The quantity caught by the fishers of Hordaland amounted to 22–23 per cent of the total landings of herring during the period 1920–1940. Typically, when the fishery was good with high volumes, the price was low, and when the volume was low, the price was high. Hence, it may be said that the market mechanism partially, but far from totally, internalized the income risk by reducing the variance in the aggregated income.

Geographical Differences in the Use of Fishing Gear

In the late 1930s, about 6000 fishers from Hordaland county participated in the winter herring fishery. The most common fishing method in Hordaland then was land-seining, which involves small vessels setting hand-manoeuvred nets along the coast. Land-seining was particularly risky in that the fishers had to wait for the herring, and could only catch them when they became available close to shore.

Most of the land-seine fishery was organized by a seine group. This form of organization reduced the economic risk to fishers, because investment in seines was divided among a group of them. The income generated from a number of seines increased the probability of each fisher receiving a share of the catch and a more equitable distribution of the proceeds. The average net income from the herring fishery in the interwar period was 200–300 kroner per fisher during the three-month harvesting period. In the late 1930s, the average total income from fishing was about 700 kroner per full-time fisher in Hordaland County. As a comparison, the average income for small farmers and farmers was about 1000 and 1400 kroner respectively, and average taxable annual income overall was 1400–1700 kroner per taxpayer in Hordaland in the late 1930s (Table 2.2). During World War II, the price of fish in general increased, as did the income received by fishers (Figure 2.4).

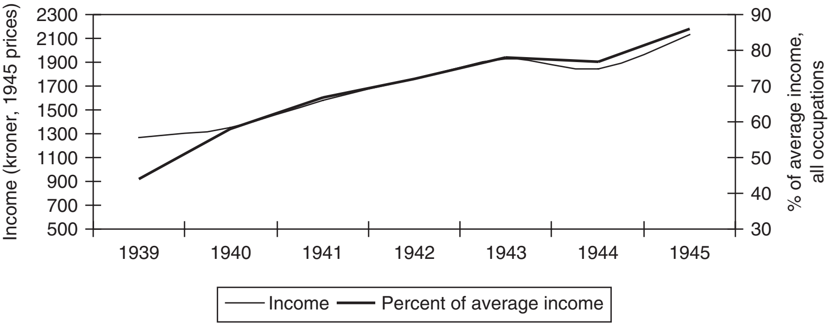
Gradually the land-seine was replaced by modern, more efficient, capital-intensive purse-seine technology, which provided the mobility to operate farther offshore for herring. This evolution stimulated investment in the fishery along the west coast, particularly in the towns of Stavanger, Haugesund, Bergen, Florø, Måløy and Ålesund, and also in Møre/Romsdal and Rogaland counties.

Table 2.2 Estimated income per taxpayer in Hordaland county 1937–1939

Year	Fishers (full-time)	Small farmers	Farmers	Average in the county
1937	636	965	1373	1481
1938	732	1005	1427	1580
1939	735	1079	1487	1668

Note: 1 krone in 1935 is equal to about 30 kroner in 2005.

Source: Johansen (1989) and Norway's Official Statistics (NOS): Tax statistics for budget year 1937/1938–1939/1940.



Source: Based on data from Johansen (1989).

Figure 2.4 Average income of fishers in Hordaland

The North Sea Herring Fishery

The North Sea herring fishery has a long history. Fishers from Holland were engaged in an extensive herring fishery as early as the 1600s. The herring fishery in the North Sea began to decline in the early 1900s, and from 1920 to 1933 there was hardly any herring fishing there. In 1934, Norwegian fishers, operating with drift nets, took part in the fishery, and by the late 1930s there were about 500 fishers involved. The bulk of the catch was landed in Hordaland county. During the 1960s the North Sea herring fishery changed completely – mainly because of the introduction of the modern purse-seine technology, and the efficiency gains that followed.

Data on aggregate landings from the different herring fisheries show how they have fluctuated over the years (Figures 2.1, 2.2 and 2.5). During the inter-war period the small herring fishery accounted for 4 per cent of the total landed value in Hordaland county, sprat another 13 per cent, and the winter herring fishery for about 49 per cent. It is clear that the herring and sprat fisheries were important for the fishing sector and for the communities along the coast of Hordaland county.

The Sprat Fishery

The land-seine fishing fleet that fished for herring – especially small or juvenile herring – also fished for sprat. Landings of sprat increased after 1900, among other reasons because of technical improvements in the canning industry. The export of canned, smoked sprat in oil virtually exploded in the period 1903–1915 (Johansen 1989). Landings and prices were highly volatile, even higher than in the herring fishery. This is largely attributable to the short lifespan of sprat, and also its sensitivity to changes in ocean currents, water temperature and the abundance of forage fish.

In 1934, 194 fish-processing operations canned sprat in Norway, and 31 of these were in Hordaland county. The development of the sprat fishery was similar to that of the herring fishery – that is, the purse-seine technology gradually took over. On average, some 2500–3500 fishers participated in this fishery in the late 1930s.

Fishing and Other Occupations

Income from fishing was usually not high enough to cover the cost of living for an entire year (Table 2.2), so fishers typically engaged in other occupations besides fishing. In the inter-war period it was normal for fishers to supplement their income through farming, carpentry, masonry and other crafts. However, the tendency was towards increased specialization within the labour force, and fishing became more of a full-time occupation as vessels and fleets adopted post-war technology to expand the size and scope of their operations.

After World War II, the importance of land-seine and set-net fishing gears diminished, and the use of purse-seines increased (Table 2.3). This represented a shift towards more capital-intensive gear, and consequently the labour input per ton of catch decreased, though it became much more specialized.

Table 2.3 Norwegian landings of winter herring according to fishing gear, 1945–1956

Year	Purse-seine	Land-seine	Drift net	Other categories of fishing net
1945–1947	42%	4%	25%	29%
1948–1950	47%	9%	24%	20%
1951–1953	61%	1%	29%	9%
1954–1956	68%	1%	27%	5%

Note: Figures may not add up to 100% due to rounding.

Source: Johansen (1989).

A TIME OF CHANGE

The years between 1940 and 1960 were a period of change for the fisheries off the west coast of Norway. First there were the problems resulting from the war and occupation, and then the reconstruction and investment in a new fishing fleet – for instance investment in new fishing technology and bigger boats. Finally, there was the sudden collapse of the winter herring fishery in the late 1950s and of the Atlanto-Scandian herring stock about ten years later.

The years following World War II and into the early 1960s can be regarded as a crossroads for the Norwegian fishing industry. Until then, Norwegian fisheries were typical coastal ones, most taking place near the archipelago. However, the crisis in the herring fishery in the late 1950s forced more and more boats out of the archipelago and into the open sea in search of alternative fishing grounds. Significant improvements in fishing technology then made it possible for Norway to commence distant-water fishing.

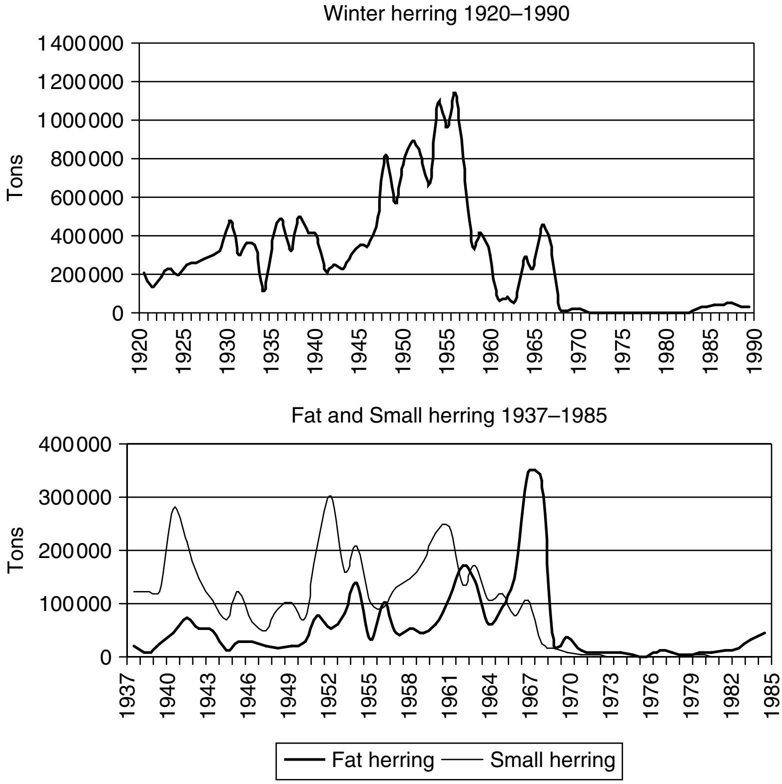
The Winter Herring Fishery after World War II

In the immediate post-war years (1946–1947), 17 000–18 000 fishers participated annually in the winter herring fishery. In 1950 total participation was 24 600 fishers, increasing further to about 28 000 by 1957. Some 4000–4500 fishers from Hordaland county were involved. If we also include crew on freight boats, the winter herring fishery probably employed 1000 additional men in Hordaland. It employed more fishers in Møre/Romsdal county, Rogaland county and Sogn/Fjordane county than in Hordaland county. Figure 2.5 shows how the volumes of the catches developed over time. While the annual landings before the war, in 1938, were 490 000 tons, the volume rose in 1948 to 840 000 tons, and to about 1 170 000 tons by 1956.

The Fishery Moves North

During the period 1945–1956 the winter herring fishery moved north along the coast, away from Rogaland and Hordaland counties and towards Møre/Romsdal and Sogn/Fjordane counties.[1] Fishers from Rogaland and Hordaland traditionally harvested spring herring, but as migration patterns of the herring changed, Hordaland fishers also began catching the big herring later in the year.

Herring catches by fishers from Rogaland and Hordaland counties averaged 150 000 tons a year during that period, but average total landings



Winter herring 1920–1990

Fat and Small herring 1937–1985

Source: Statistics Norway.

Figure 2.5 Landings of Atlanto-Scandian herring

of herring in Hordaland were about 440000 tons. Hordaland received landings from other areas because the production capacity in surrounding counties was too small to absorb all the catches there.

THE CRISIS IN THE HERRING FISHERY: CONSEQUENCES, ADAPTATION, CAUSES

Before the war and immediately thereafter there was great optimism, and investment in Norwegian herring fisheries increased substantially. Increased landings created greater optimism and the expectation of greater prosperity. However, as fishing became more capital-intensive, the risk of being inflexible and economically vulnerable to negative shocks, such as lower fish prices and less catch, increased.

The decline of the winter herring fishery started in the 1957 season, after ten years of steady growth. That year the purse-seine fishery failed, and the following year the drift net fishery failed. Landings fell from >1 100 000 tons to <100 000 tons by 1961. There was then a minor recovery from 1964 to 1967, but the fishery then virtually disappeared until the mid-1980s (Figure 2.5).

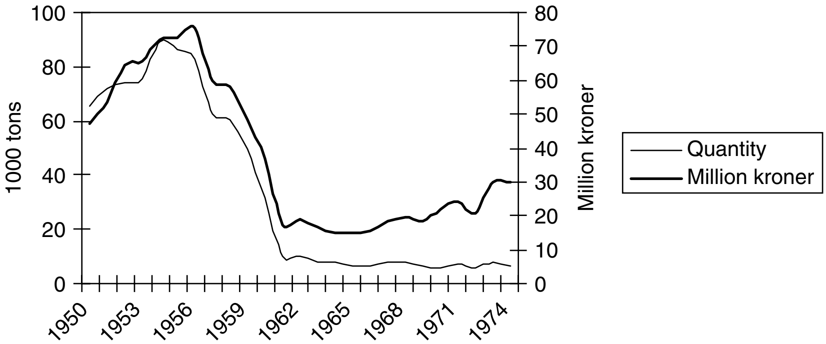
On average, a purse-seiner had to catch between 9000 and 10 000 hectolitres of herring to break even financially in the late 1950s. The break-even catch gave the crew an income of 7000–8000 kroner per fisher for a fishery of 2–2½ months (one 1955-krone is equal to 11.5 kroner in 2005 – prices). The average salary for an industrial worker at that time was about 1000 kroner per month. Under the economic conditions prevalent then, getting a crew for the purse-seiners was not a problem. After the collapse in the herring fishery in 1957, fisher income reduced dramatically, and many had to make do with just 300 kroner per month (Johansen, 1989).

The combination of decreasing landings of herring, bad weather and a boom in investment that increased participation and harvesting capacity in the fishery, made the herring fishery unprofitable. This crisis laid the foundation for the changes in Norway's management of its fisheries that were to follow.

Multiplier Effects

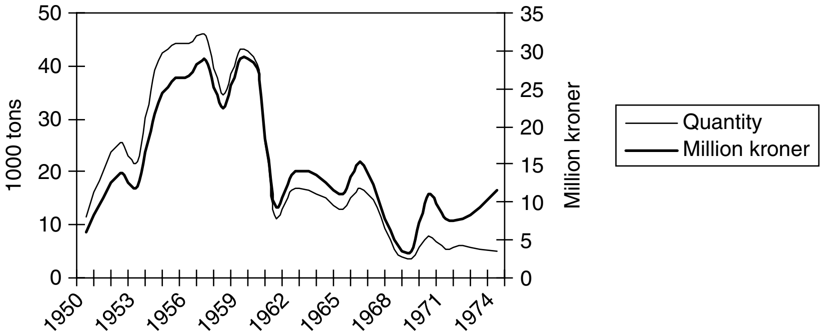
It would be expected that a dramatic and sudden change in income at the top of the value chain would be quickly transmitted to the lower levels. The sudden reduction in herring landings affected respectively the oil and fishmeal industry, the industry that made salted herring products, exporters of fresh and frozen herring, the canning industry, the barrel industry and boatyards. No less than 75 per cent of the raw material for the oil and fishmeal industry came from the winter herring fishery. Further, the tax income and activity in many fish-dependent municipalities were negatively impacted by the crisis.

To illustrate how the collapse in the winter herring was transmitted to the rest of the value-added chain, we should look at the values and quantities for Norwegian exports of salted herring (Figure 2.6), frozen herring (Figure 2.7) and spiced and salted herring (Figure 2.8) during the period 1950–1974. Values and quantities for all Norwegian fish and fish product exports over the same time period are presented in Figure 2.9. Because frozen and salted herring products can be kept in store for some time, minor time lags between the decrease in landings and the decrease in exports (Figures 2.5–2.8) would be expected. In the years before the collapse of the



Source: Statistics Norway: Fishery Statistics.

Figure 2.6 Export of salted herring 1950–1974

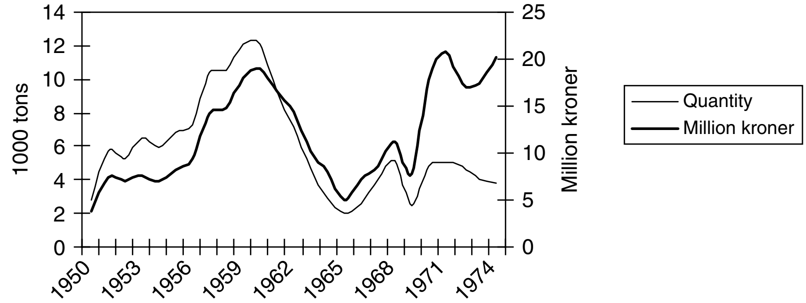


Source: Statistics Norway: Fishery Statistics.

Figure 2.7 Export of frozen herring 1950–1974

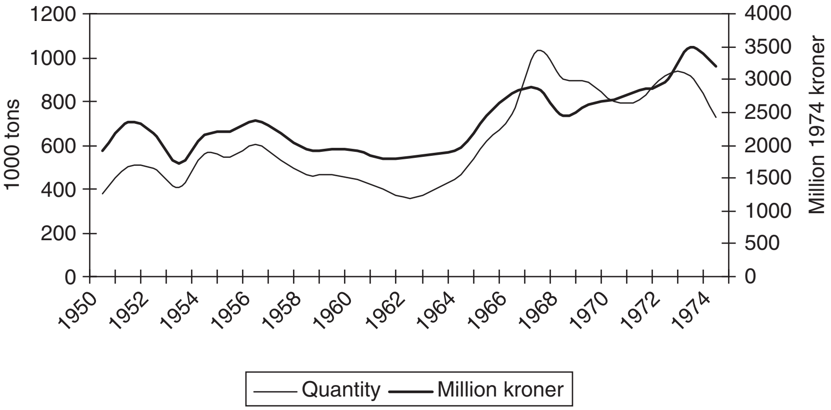
winter herring fishery, the export value and quantity increased. The turning point was about 1958–1960, depending on the export product, and 2–4 years after the decline started, volumes and values of exports declined significantly. The decline was slightly offset by a positive price effect, probably because of the reduction in supply. The slight increase in exports in the early 1970s is due to the landings of herring from the North Sea.

The collapse of the North Sea herring fishery started about 1966–1967, and the effect on exports is indicated in the export figures for frozen and for spiced and salted herring (Figures 2.6–2.8). The slight increase in exports about 1966–1967 can be explained by the temporary increase in



Source: Statistics Norway: Fishery Statistics.

Figure 2.8 Export of spiced and salted herring 1950–1974



Source: Statistics Norway: Fishery Statistics.

Figure 2.9 Total export of fish and fish products 1950–1974

the landings then of winter herring. The collapse of the winter herring fishery is also revealed by the falling exports of all fish and fish products (Figure 2.9), which dropped in value and volume for the period 1956/57–1962; the effects of the collapse of the North Sea herring are shown by the dip in aggregated exports during the period 1967–1971.

What Caused the Crisis?

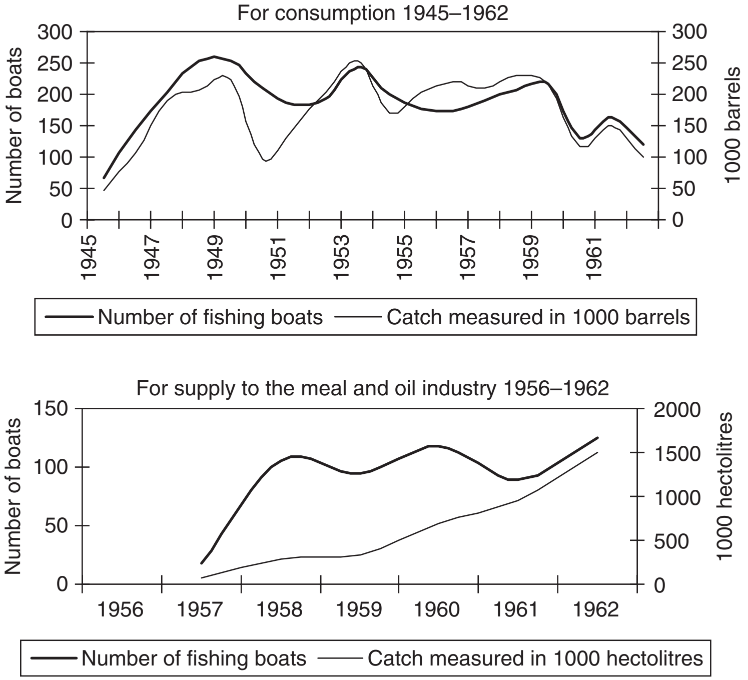
In the 1952/53 season, fishers observed fewer herring than in previous years. Furthermore, an article in the newspaper *Fiskaren* in 1954 indicated that the increase in total landings was less than expected given the increase in harvesting efficiency and capacity of the fishing fleet. These improvements tend to squeeze profitability in the fishery, and increase the probability of overexploiting fishery resources.

In the period 1950–1959 no strong age classes of herring were found. About 60 per cent of the catch in 1959 was fish of the 1950 year-class. It has also been argued that the nutrition and spawning conditions were impacted unfavourably then. Today, oceanographers attribute the changes to changes in ocean temperature and currents (Johansen, 1989). Not only was recruitment during the period 1950–1959 weak, but one- and two-year-old herring were also being heavily exploited by the Norwegian fleet then. (These issues are discussed in the press, for example in *Fiskaren* 12, 16 May 1954 and *Fiskaren* 11 November 1969.)

During the 1960s, herring researchers at the Institute of Marine Research (IMR) in Norway came to believe that, to achieve sustainability of the resource, it was necessary to protect juvenile herring and simultaneously to reduce the exploitation of so-called fat herring between three and four years old. Some researchers concluded then that the collapse in the herring fishery in the late 1950s was mainly attributable to overexploitation of juvenile herring. The overexploitation hypothesis has come to be widely accepted, but oceanographers have since pointed out that the demise of the herring stocks coincided with declining temperatures in the Northeast Atlantic (Stenevik and Sundby, 2004). The causal relationships may therefore be more complicated. Herring stocks fluctuated and periodically disappeared long before the technological advances in fishing that put them under unprecedented pressure in the 1960s (Fasting, 1960).

How Did the Fishing Industry Adapt to the Crisis?

The winter herring fishery was pivotal for most Norwegian west coast fishers, both during the inter-war period and following the war until its collapse in 1957. Half their income was based on that particular fishery, and their allocation of fishing effort between fisheries depended mainly on the size of the boat. The fishing boats that took part in the winter herring fishery can be roughly divided into two groups: boats 80–90 feet or smaller boats. In June and July the large boats engaged in the herring fishery off Iceland and in the North Sea, and when they were not fishing, they often engaged in shipping. The small boats took part in the fat herring fishery, the small herring fishery, and in the sprat fishery in the fjords and the archipelago. North of Stadt, a combination of fat herring and small herring (juvenile) operations was typical, whereas south of Stadt, the typical combination was small herring (juveniles) and sprat.



Source: Johansen (1989).

Figure 2.10 Norwegian catches of herring close to Iceland, and vessel participation in the fisheries

When an important fishery fails, fishers will seek alternatives, so when the winter herring fishery failed in the late 1950s, fishing on juvenile herring intensified. Some characterized this fishery, that occurred in the fjords in North Norway, as 'vacuum cleaning'. The fish were delivered to the oil and fishmeal industry (Johansen, 1989). South of Stadt, landings of juvenile herring also increased, but the fish were destined for the canning industry.

Hordaland was the most important spratfishing county in Norway over the period 1940–1960. The sprat fishery also became more efficient and more capital-intensive with the transition from land-seine to purse-seine. Herring caught in summer (July–September) off Iceland belonged to the Atlanto-Scandian winter herring (fat and big herring) resource, the so-called 'spring spawning' herring (NSSH) caught along the coast of Norway in winter. Norwegian harvests, and vessel participation in the herring fishery off Iceland in the period 1945–1962, are shown in Figure 2.10.

The location of the herring fishery off Iceland has changed over the years. Until about 1960 it was located along the coast north of Iceland. Later, it took place east of Iceland (Figure 2.3), partly in the open Atlantic north of the Faeroe Islands up to Jan Mayen. The season also increased in duration, originally from July to September but extending to July–November/December during the 1960s.

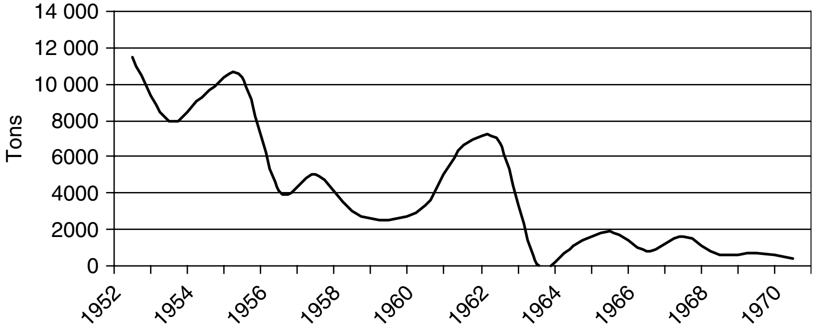
During the heyday of the herring fisheries off Norway, the supply of herring to the oil and fishmeal factories was plentiful. After the collapse in the late 1950s and 1960s, this shortfall was compensated for by shifting herring caught off Iceland from human consumption to the production of oil and meal (Figure 2.10).

The North Sea Herring Fishery

The North Sea herring fishery has a long tradition, in particular for the fishing industries of Holland, Sweden, Denmark and Germany. Norwegian fishers used drift nets to catch herring in the North Sea in the period 1900–1935, and the Norwegian share of the catch was marginal compared with that of other nations. However, because of the rich winter herring fishery and full capacity utilization, there was no incentive to increase effort in the North Sea. After the collapse of the winter herring fishery in the late 1950s, and with the introduction of modern purse-seine technology, it became technologically possible and profitable for Norwegian fishers to exploit the herring resources in the North Sea. Hence, this fishery expanded dramatically during the 1960s.

Herring are Gone – So are Tuna

During the 1950s landings in the tuna (*Thunnus thynnus*) fishery along the coast of Norway started to decrease (Figure 2.11). When the tuna fishery was at its highest it employed 4000–5000 men. Figure 2.11 shows how this fishery evolved over time. As tuna feed on herring, they tend to follow the herring, and when the herring disappeared in the late 1950s and early 1960s, so did the tuna. Fisheries biologists have explained the disappearance of the tuna partly by their being overexploited by both the Norwegian fleet and Iberian fishers, and partly by the collapse of the Atlanto-Scandian herring. As a consequence of the decline in the seasonal fisheries off Norway, the Institute of Marine Research in Norway sent a research vessel in 1959 to distant-water fishing grounds to investigate whether it would be feasible for Norwegian fishing vessels to exploit tuna resources in distant areas. Some attempts were made, but the conclusion was that it was not profitable.



Source: Statistics Norway: Fishery Statistics.

Figure 2.11 Landings of tuna in Norway 1952–1970

Other Fisheries Along the Coast of Hordaland

The saithe fishery was important for the fishers of Hordaland county. Saithe forage both on herring and herring roe, so when the herring disappeared in the late 1950s it also brought a decline to the saithe fishery.

The fishers of Hordaland county also participated in other fisheries (Table 2.4), though the number participating in all fisheries, except that for basking sharks, fell sharply during the 1950s and 1960s.

Local Differences in Hordaland County

Within Hordaland county, the purse-seine fleet was concentrated in the municipalities located in the middle of the county: Austevoll, Sund and Fjell. Most of the purse-seiners were located there, and their prosperity was founded on the winter herring fishery, the tuna fishery and the herring fishery off Iceland.

The fishery was more fragmented in the municipalities in the northern part of Hordaland county. These municipalities – Herdla, Hjelme, Fedje and Austrheim – had fewer large fishing boats, but more medium-sized and small boats. Net gear (land and drift net) was typically used to catch herring. Fishers there also caught other species such as saithe, pollack (whiting), cod, haddock, bream, small whales and ling.

Typical of the municipalities in the south was greater diversity in the size of the fishing boats and the types of fishing gear used. The most important fisheries were the mackerel and lobster fisheries. Fishing tended to be based on local resources, close to the coast.

Table 2.4 Participation in different fisheries by fishers from Hordaland county 1947/48 and 1959/60

Fishery	1947/48	1959/60	Change
Saithe	811	260	−551
Mackerel	1064	660	−404
Salmon	588	395	−193
Lobster	2035	983	−1052
Crabs	829	328	−501
Fjord cod	196	184	−12
Mackerel shark	192	60	−132
Basking shark	56	151	+95
Eel	172	135	−37
Small whale	115	71	−44
Bream, ling, whiting, flatfish, haddock, etc.	2049	1216	−833
Total reduction in number of fishers			−3664

Source: Johansen (1989), p. 12.

In general, we can conclude that the coastal fishery was based on relatively local resources, mainly sprat, mackerel, lobster, crab and the young year-classes of saithe. On the other hand the herring fishery was pursued more intensively by the purse-seine fleet, which was continuously improving technologically, and expanding its area of operation.

In a historical perspective, the Norwegian economy was evolving towards greater specialization and industrialization, that is, away from a subsistence economy, at the same time as fishery resources started to decline in the late 1950s. Combined occupations of fishing and farming gradually became less and less important.

A Coast in Crisis – Major Problems in the Fishing Industry about 1960

For the Norwegian fishing industry, the period from 1930 into the 1950s was generally positive, and income earned by fishers increased relative to the situation in many other occupational groups. Needless to say, the optimism turned to pessimism when the winter herring fishery collapsed in 1957, and the situation got worse in the years that followed. Concomitantly, other fishery resources were showing signs of stress, among them the lobster, saithe and tuna fisheries. Earlier it was briefly described how the fisheries and the fishers were affected by the crisis in the late 1950s. Local

Table 2.5 Overview of the shortfalls in tax revenues in different municipalities in Hordaland county, 1957–1958

Municipality	Number of inhabitants	Number employed in the winter herring fishery	Percentage shortfall in tax
Fedje	934	400	−21.4
Austrheim	2313	250	−15.9
Hjelme	1046	400	−48.8
Herdla	5274	800	−41.1
Fjell	5316	800	−30.9
Sund	3130	500	−40.6
Austevoll	3400	800	−57.6
Fitjar	3157	250	−22.5
Bremnes	4732	400	−14.7
Moster	1755	200	−13.0
Bømlo	1466	400	−34.5
All counties	32 523	5200	−31.4%

Source: Johansen (1989) and Hordaland county courthouse archives.

public finances are another indicator of how the collapse in the herring fishery affected people in the fishing communities.

Table 2.5 shows the economic condition in a sample of affected municipalities, and how much the tax income was reduced when the herring fishery collapsed. The dramatic reduction in tax from both individual taxpayers and from companies, affected the supply of public services and public investment. The fiscal situation in many municipalities could be characterized as critical. Many people likened the situation to a natural catastrophe (Johansen, 1989).

Government Response to the Crisis

In response to the collapse of the herring fishery, the government granted loans to ease economic hardship in the herring fleets and provided economic assistance to those fishers hardest hit by the collapse. Many exited the fisheries, especially those who came from the inner parts of the fjords and who combined fishing and farming. The government also initiated a major highway development programme to employ people in road construction. A number of those displaced from the fisheries found employment in international shipping. In Austevoll, for example, the number of merchant sailors increased from 157 in 1956 to 307 in 1959 (Johansen, 1989).

Table 2.6 Number of fishers in Hordaland county in 1948 and 1960

Year	Total number of fishers	Number of full-time fishers	Number of fishers combining their fishing activity with small-scale farming
1948	7959	1197	4496
1960	4650	1344	1532
Change over 12 years	−3309	+147	−2964

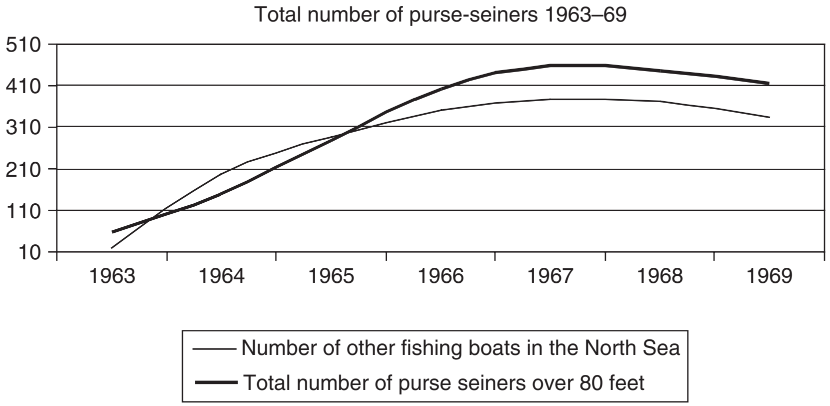
Source: Norwegian Central Bureau of Statistics (1948, 1960).

Table 2.6 compares the number of people who were full-time fishers with those who combined fishing and farming. Besides farming, people combined occupations like building and construction, marine transport and work in the manufacturing industry. Because income from the herring fishery accounted for such a large share of the combined total income, its collapse pushed the peasant-fisher 'system' out of balance. The decline of the farmer-fisher combination was also promoted by the general progression of Norwegian society towards increased specialization of the labour force, industrialization, increased education, urbanization and increased career choice.

In addition to short-term responses to the crisis caused by the herring collapse, industry, commerce and local government tried to implement a long-term policy, which focused on establishing more all-round or versatile economic activity, and on increasing the efficiency of and developing the fishing industry in general. The relatively rapid technological development then implied that the fishing industry did not have to be entirely based on seasonal fisheries. The main goal was to develop policies that would increase efficiency and provide year-round activity and greater profitability in the fisheries. Clearly, the collapse of the herring fisheries inspired people to contemplate, and search for, new opportunities.

Technological Changes in the Fishing Industry

Up to the late 1950s the Norwegian fishing industry consisted of primarily seasonal, coastal fisheries. Offshore catches (taken near Iceland, West Greenland, Bear Island, Spitsbergen and the Barents Sea) amounted to just 10–15 per cent of Norway's total catch of fish in the 1950s. However, in the 1960s this changed, and by 1965 the offshore fishery contributed 36 per cent



Source: Johansen 1989.

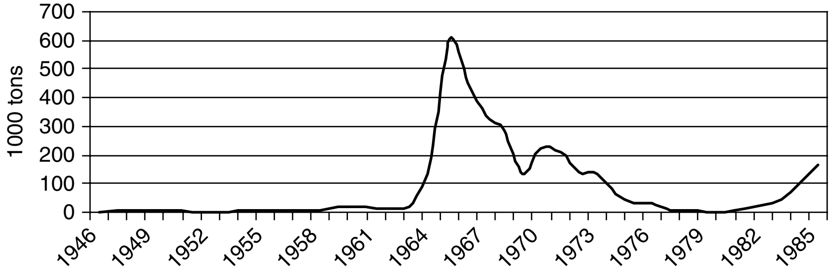
Figure 2.12 Participation in the North Sea herring fishery

of the total catch. The major structural changes were along the west coast of Norway. The growth in the mackerel and herring fishery in the North Sea increased rapidly during the 1960s. New fishing technology made it possible to exploit fish resources far from the coast. Significant advances were made in the ability to detect fish (for example, asdic and other echo sounding technologies), in manoeuvring technology, such as side propellers or bow thrusters, and in processing technology, such as pumping the fish on board. Also, larger boats with greater holding capacity were built, and new technology for cooling and for laying out and hauling in the seine was developed. The introduction of the power block in the early 1960s increased significantly the efficiency in the herring and mackerel fisheries. For the big purse-seines the power block technology reduced the need for fishers by as many as 7–8 per boat, reducing the need and demand for labour by almost a half within a few years.

Figure 2.12 shows participation by Norway in the North Sea herring and mackerel fishery, and the increase in the number of purse-seiners more than 80 feet long. It took 3–4 years to rebuild the purse-seine fleet and to equip it with the power block during the 1960s. Norway's first year of operating with modern purse-seine technology in the North Sea was 1964.

From an Open Fishery to Regulations

The great efficiency of the power-block purse-seine technology was soon seen in the North Sea herring fishery. About 1960 the Norwegian fleet landed 17000 tons of North Sea herring. Then, in the first year of the



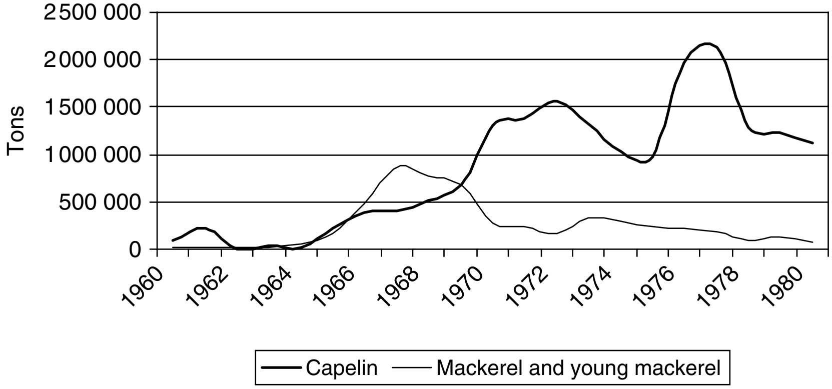
Source: Statistics Norway: Fishery Statistics.

Figure 2.13 Norwegian landings of North Sea herring during the period 1946–85

power block in the North Sea, the Norwegian purse-seine fleet caught 186 000 tons of herring; in just one year, the annual catch rose further to about 600 000 tons. The power block purse-seine technology was much more efficient than the North Sea trawling technology previously used. The result was predictable with such efficiency gains: more effort was attracted into the North Sea fisheries. The effort of other nations taking part in the fishery also burgeoned, notably Denmark, Sweden, Holland and West Germany, and concerns of overexploitation were soon voiced, especially with regard to the North Sea herring fishery.

Total landings of North Sea herring increased from 744 000 tons in 1960 to about 1 580 000 tons in 1965, though it then dropped to about 1 116 000 tons in 1966, and even further in 1967 and subsequent years. Figure 2.12 shows that the entry rate into the North Sea fisheries was high. Financial institutions complained to the government, wanting to restrict entry, so in the late 1960s some restrictions on the import of steel boats older than ten years and wooden boats older than five years were implemented.

The herring fishery in the North Sea declined dramatically during the 1970s, and in 1977 was closed completely. What were the reasons for the decline? In 1967, Danish fishery biologists had complained that exploitation of the North Sea herring was too heavy, but their Norwegian colleagues did not agree with them. What was clear to all, however, was that recruitment of herring was weak during the years 1964–1968. It is difficult to pinpoint the main cause of the poor recruitment, though variously it has been attributed to changes in the ecosystem causing irregular and low recruitment, and/or to overexploitation of the resource, hypotheses similar to those advanced for the collapse of the Atlanto-Scandian herring in 1957. In any case, in 1969 Norwegian fishery biologists came out in favour of protecting the young year classes of herring.



Source: Statistics Norway: Fishery Statistics.

Figure 2.14 Catches of capelin and mackerel 1960–1980

Looking for Alternative Resources to Exploit

Besides the North Sea herring fishery, winter herring was an extremely important fishery for the Norwegian purse-seine fleet in the first few years of the 1960s. The decline and subsequent collapse of the herring fishery, both winter herring and the North Sea herring, encouraged the purse-seine fleet to look for alternative resources to exploit, so shortly after the decline, it intensified its exploitation of mackerel in the North Sea. The growth of that fishery was similar to the development of the herring fishery in the North Sea some years earlier, and catches peaked in 1968. Thereafter, landings decreased except for a short boom in 1973 (Figure 2.14).

Other fishing nations also increased their levels of exploitation of mackerel in the North Sea, and the need to establish an annual allowable catch was generally accepted. Fishery biologists stressed that the total catch was greater than their recommended catch, but at this point in time, international cooperation on fisheries management had not yet been institutionalized. After the decline of the mackerel fishery, the purse-seine fleet was 'rescued' by a new fishery, for capelin (winter and summer) off the coast of Finnmark county. Almost all the capelin catch was processed by the fishmeal and oil industry. The capelin fishery in summer was farther away from the coast of Finnmark than the winter fishery.

The relatively fast growth of the winter capelin fishery (Figure 2.14) led to its regulation in 1970, with initially a general prohibition on fishing for the species west of 0° longitude during the period 1 June to 15 July. Further regulations were implemented after 1971, and the capelin fishery became one of the most regulated fisheries. The tragedy of the commons[2] (Hardin 1968) gradually unfolded and ultimately became recognized by everyone in the fishing industry. Because of the dramatically enhanced fishing technology and the situation after 1970, fishery biologists came to understand that they had to play the role of 'watchdog', to preclude overexploitation and potential collapse of this fishery.

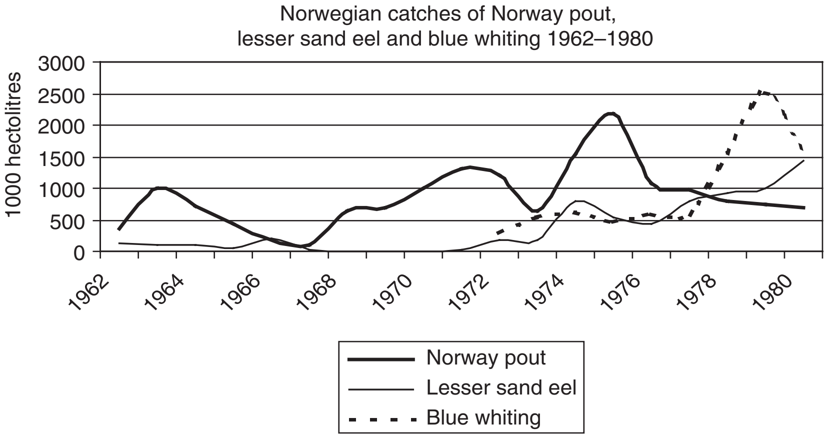
EXPANSION IN NEW FISHERIES – SAND EEL, NORWAY POUT AND BLUE WHITING

During the late 1950s, especially after the collapse of the winter herring fishery, new fisheries continued to evolve, for example the sand eel fishery, which was started by Danish fishers. The oil and fishmeal industry was searching for substitute input after the collapse of the winter herring fishery in the late 1950s, so sand eel was seen as a suitable alternative and many purse-seiners were rebuilt in the early 1960s so they could trawl for sand eels in the North Sea.

In 1959/1960, Norwegian fishers also began to exploit Norway pout west and south of Jæren in a trawl fishery that was also originated by Danish fishermen. The Norwegian fishing fleet landed 90 000 tons of Norway pout in 1963, rising within ten years to about 180 000 tons. However, the variability in the Norway pout fishery was relatively high, because the lifespan of Norway pout is about three years, so as a species it is similar to sprat and capelin in terms of its biological productivity.

During the 1970s it was quite easy to get trawling concessions for sand eel and Norway pout, and in 1978, 405 fishing boats had concessions to exploit the two species. Trawling for capelin off the coast of Finnmark county in winter and for Norway pout and sand eel in the North Sea during summer was apparently a lucrative combination. This pattern of operation was typical for medium-sized purse-seiners, which were converted to trawling because they were not big enough to install power block technology. Many North Sea trawlers also harvested blue whiting as an incidental catch in the Norway pout fishery. Figure 2.15 shows the landed quantity of Norway pout, sand eel and blue whiting during the period 1962–1980.

The Norwegian purse-seine fleet also participated in the sprat fishery in the North Sea, near Scotland and northern England, in the early 1970s, and landings increased dramatically. However, the combination of falling prices of sprat in the raw fish market and in cans resulted in an economic crisis for the sprat fishers along the west coast of Norway. A concession system for the sprat fishery was therefore approved in 1979. When it was reduced and regulated, seining for saithe (of medium size) increased. There



Note: 1 hectolitre is about 100 kilograms.

Source: Statistics Norway: Fishery Statistics.

Figure 2.15 Norwegian catches of 'new' species in the North Sea for the oil and meal industry

was also an increasing demand for fillets of saithe at that time. In the late 1960s, 1000–1500 tons of saithe were landed in Hordaland county annually, but these landings increased to more than 4000 tons by 1972 and peaked at 6000 tons in 1981.

In Hordaland county, purse-seine fishers also participated in the drift net fishery because they had time to do so between the herring seasons. Concessions were introduced into the drift net fishery in 1978.

Aquaculture and Oil

Many fishers became active in the development of the Norwegian aquaculture and petroleum industries during the 1970s. The combination of a decline in some fisheries, and technological development in fisheries in general, decreased the number of job opportunities for fishers, and many of them found employment in fish farming and in the growing petroleum industry along the west coast of Norway.

TWO FISHING COMMUNITIES – HOW THEY ADAPTED

All communities on the west coast of Norway were to some extent negatively impacted by the collapse of the herring fishery in the late 1950s.

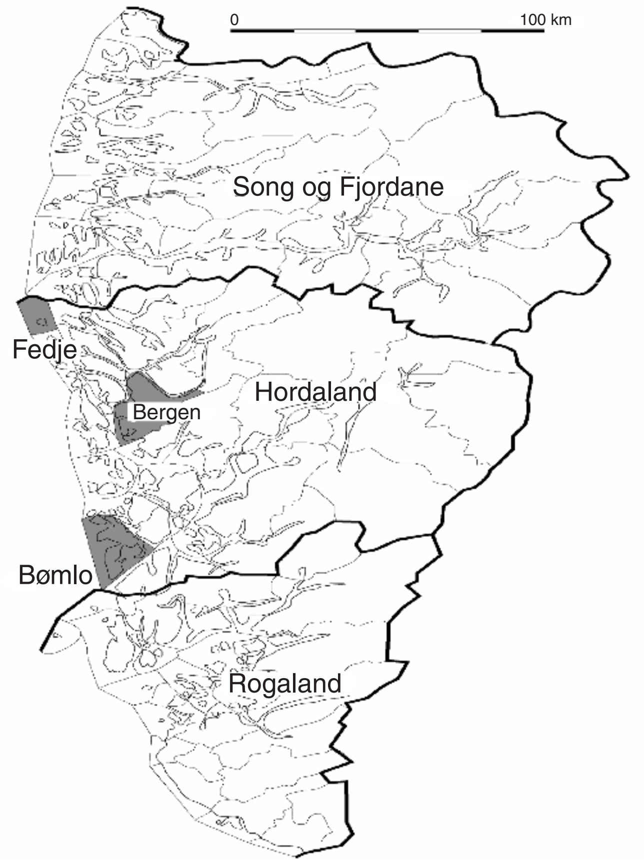


Figure 2.16 Bømlo in the south and Fedje in the north of Hordaland

Bømlo and Fedje are two small fishing communities, located respectively in the extreme south and north of Hordaland county (Figure 2.16). The collapse and changes in the fisheries during the late 1950s and 1960s significantly affected both communities, but they adapted and evolved differently.

The structure of the Bømlo and Fedje Fishing Fleets in 1958

In 1958 the composition of the fishing fleet in Bømlo was: 50 fishing boats <30 feet, 31 fishing boats 30–50 feet, 19 fishing boats 60–90 feet and 8 fishing boats >90 feet. In the same year, the fleet in Fedje comprised

49 fishing boats <50 feet, 27 fishing boats 30–60 feet, 10 fishing boats 60–90 feet and 1 fishing boat >90 feet.

Traditionally, Bømlo has had a relatively large and versatile fishing fleet, and the number of fishers has remained relatively stable over the past 20 years. Fedje, in contrast, has experienced concomitant emigration and a reduction in the number of fishing vessels and fishers.

Bømlo

Diversification characterizes the fishing community of Bømlo. From lobster fishing along the coast to the distant-water herring fishery off Iceland, Bømlo fishermen have participated in a wide variety of fisheries. Bømlo's fishing population is made up of many full-time fishers, plus many who combine fishing with farming or other occupations. In the early 1950s, the fishing industry provided about 70 per cent of Bømlo's total tax income.

After World War II, the fishers of Bømlo made the transition from land-seine to purse-seine technology relatively quickly. With a well-established modern purse-seine fleet, Bømlo managed to survive the tumultuous decline of the herring fisheries during the late 1960s.

In contrast to many other fishing-dependent communities, Bømlo avoided reducing its purse-seine fleet during the 1970s. Moreover, it succeeded in becoming the largest trawler municipality, supplying sand eel, Norway pout and blue whiting to meal and oil processors. Fishers in Bømlo were also successful in developing a modern longline fishery for ling, tusk and cod, an enterprise that was not seen in other areas of Hordaland county. In addition, Bømlo built up a relatively extensive and differentiated coastal fishing fleet: (i) seine-fishing for saithe; (ii) drift net fishing for salmon and mackerel; (iii) pot fishing for eel, crab and lobster; and (iv) net and longline fishing for ling, tusk, cod and so on. Overall, Bømlo, through modernization and endeavour, has been able to weather turbulent times in the fisheries and to remain a stable and successful fishing community.

Fedje

Fedje has a history of dependence on fisheries, between 60 and 70 per cent of the tax income in the municipality being generated traditionally by the fishing industry. In the late 1940s, four out of five workers were employed in the fishing industry. During the heyday of the winter herring fishery, Fedje was one of the central harbours for the fishing fleet. However, with the collapse of the herring and other fisheries, Fedje became less important as a fishing harbour.

Drift nets and land-seines were the traditional gears used by Fedje fishers, and they did not shift to purse-seines in the 1950s as was the case for the fishers in Bømlo. One important reason was that relatively big purse-seiners (90–100 feet or more) were not useful for the alternative fisheries which fishers in Fedje relied on then, for example, tuna fishing, whaling and fishing for basking sharks. The traditional fishing technologies in combination with a fishing fleet that included only a few boats >60 feet, made it difficult to make the transition to modern purse-seine technology. Hence, when the herring moved farther north after World War II, Fedje did not have the large mobile purse-seiners to pursue them, or to take advantage of other distant-water purse-seine fishing opportunities. Moreover, when the purse-seine fishery became regulated in the 1970s, it became almost impossible to enter it. As a result, Fedje became less important as a fishing harbour to its own inhabitants, as well as to outsiders who might have found it beneficial to land their catches there.

Although Fedje failed to make major investment in modern purse-seine technology, it did embark on a programme of expansion during the 1950s and 1960s to support other promising fisheries: (i) a seine fishery for saithe; (ii) a North Sea trawl fishery for sand eel, Norway pout and blue whiting; (iii) a basking shark fishery; and (iv) whaling. In the early 1950s it established a canning factory, and fishery-related economic activity expanded during the 1960s with the establishment of a cold storage plant, a fishmeal factory and various other fishery infrastructures. However, the promise of those fisheries was relatively short-lived, and by the end of the 1970s seine-fishing for saithe was gone, trawling in the North Sea for blue whiting, sand eel and Norway pout had effectively ended, and whaling was marginal or prohibited. The Fedje cold storage plant went bankrupt in 1983, and the fishmeal plant in 1986.

The introduction of the concession system for regulating the fisheries added to the woes of the fishing industry in Fedje. The problem was that Fedje fishers were rather unreceptive when invited to apply for concessions for the different developing fisheries. Fishers from Fedje also lost concessions or fishing rights because they did not use them. This lack of interest made the situation in Fedje's fishing industry even worse.

Given its history, it would seem that the fishing industry in Fedje had the same capacity to adapt to changes in Norway's west coast fisheries as did that of Bømlo. However, the fishing industry in Fedje did not adapt to the changes in the same way. Historians attribute this 'failure' (compared with Bømlo) partly to the development of the Norwegian economy in general then. Many fishers, after the collapse of the herring fishery and the decline of alternative fisheries, went ashore to take advantage of employment opportunities in other industries. From the comparison of the two municipalities, we conclude that the ability and willingness to diversify in the face of the uncertainty and risk associated with the availability of fish stocks results in a much more stable and productive fishing industry than one dependent on narrowly specialized fishing boats and on a minor group of species.

CONCLUSION

In this chapter we have described how the collapse in the herring fishery in the late 1950s affected the fishery-dependent communities along the west coast of Norway, especially in Hordaland county. The changes were on a major scale, perhaps even spectacular and disastrous, and took place over a relatively short period of time. Negative effects of climate change on fisheries are not likely to be any greater or to be more rapid, so this narrative should provide some valuable insights as to what to expect and how to respond to future environmental impacts on fisheries. Needless to say, of course, climate change may also affect some fisheries in some areas in a positive way. Such changes pose much less of a strain, but they might nevertheless be a mixed blessing, because some fish stocks may decline or disappear while others rise. In the latter case this may necessitate structural changes that create difficulties, as in the case of a sudden, major decline.

In retrospect, the effects of the herring collapses on Norwegian fishing communities were generally soon overcome. New fish stocks were found to replace the herring that had disappeared. A major leap in technology (the power block and the asdic) was clearly the culprit in the collapse of the herring: in a very short period of time it became possible to increase the catches many times over, and there was no management mechanism in place then which could limit what was essentially an international, open access fishery. However, the new technology also made the transition to new stocks possible. The purse-seine, the power block and the asdic all helped the move to the new fisheries for mackerel and capelin. Some communities were, however, more successful than others in making the transition, and there was a transition not just to new fisheries, but to other occupations. The collapse of the herring coincided with a transition to a greater occupational specialization; the fisher-jack-of-all-trades was on his way out, giving way to more full-time employment in existing and newly developing industries. It was also a period of rapid economic growth, which eased the absorption of redundant fishers into other occupations.

The collapse of the herring stocks slightly preceded the emergence of two new growth industries of late 20th century Norway, the fish-farming industry and the petroleum industry. Many people who left the fishing industry or who would otherwise have looked for a career there found their way into those new industries. The impact of the herring stock collapses was undoubtedly much softened by the fact that they took place in a period of economic growth and rising employment.

What does this hold for future structural change of a negative kind, such as might emerge from climate change? The Norwegian economy is currently more prosperous than it has ever been, and thus in a better shape to bear the costs of any structural change, be it in the fishing industry or elsewhere. However, there are other trends that might pull in the opposite direction and make any such changes more difficult. Firstly, there are probably no 'new' stocks to which the fishing fleet can be directed, in case of a collapse of today's major fish stocks. Secondly, the labour market has in some ways become less flexible. Skills of various kinds are in high demand, and those without skills, or with special skills not easily portable to other industries, may have greater difficulties in finding new employment than the redundant fishers of the 1960s and 1970s.

APPENDIX 2.1

Species in the categories presented in Figures 2.1 and 2.2

Categories	Species included in the categories
Crustaceans	Crab, lobster and deep water prawn
Cod-like fish	Cod, haddock, saithe and various related species
Mackerels	Mackerel, capelin, horse mackerel, blue whiting, sand eel and Norway pout
Flatfish	Halibut, Greenland halibut, plaice, witch
Herring and sprat	Sprat, Icelandic herring, North Sea herring, small herring, fat herring and winter herring
Other	Mackerel shark, spiked dogfish, catfish, Norway haddock (redfish), tuna, blue ling, ling, wild salmon and trout

NOTES

1. For a definition of winter herring fishery, see the subsection 'The different types of herring'.
2. Tragedy of the commons covers situations where free access to a valuable resource, for example a fish stock, leads to overexploitation and depletion.

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3. Sharing the herring: fish migrations, strategic advantage and climate change

Rögnvaldur Hannesson

INTRODUCTION

In the 1970s, most coastal states established 200-mile exclusive economic zones (EEZs), after the concept had been sanctioned by the then ongoing Law of the Sea Conference. In cases where the continental shelf does not extend more than 200 miles from shore, fish stocks confined to the waters of continental shelves (for example cod, plaice) were enclosed by the EEZs. Such stocks do, however, often migrate extensively, so in cases where two or more countries shared the same continental shelf, the stocks became the shared property of the countries between whose zones they migrated. Stocks that inhabit surface waters (for example mackerel, herring, tuna) usually migrate more widely than bottom-dwelling fish, and often move through the high seas outside the 200-mile EEZ of any country. In cases where the continental shelf protrudes out of the exclusive economic zone (for example the Grand Banks, Barents Sea, Bering Sea), even bottom-dwelling fish are not totally confined within EEZs.

Successful management of transboundary stocks requires that the countries having an interest in them agree on how they are to be shared and managed. In the late 1970s, Norway and the European Union (EU) agreed to share the stocks that migrate between their zones according to the 'zonal attachment' of each stock.[1] Zonal attachment can be defined and measured in various ways, and precisely how it is done can be controversial. Some fish may be spawned in the EEZ of Country A, not becoming fishable until they move into the EEZ of Country B, so the question arises whose fish they really are. Other fish may feed and grow in the EEZ of Country C, but be fishable mainly in the EEZ of Country D. In the agreements between the European Union and Norway, zonal attachment was based on the presence of the fishable part of the stocks in each party's zone during the years 1974–1978 (Engesæter, 1993). Different approaches have been applied to such calculations. One uses biomass multiplied by the time that migrating stocks spend in each country's EEZ (Hamre, 1993). This method was applied to share the capelin stock that migrates between the EEZs of Greenland, Iceland and Jan Mayen, an island under Norwegian sovereignty (Engesæter, 1993), but instead of biomass, the approach could be based on growth of the stock (Hamre, 1993).

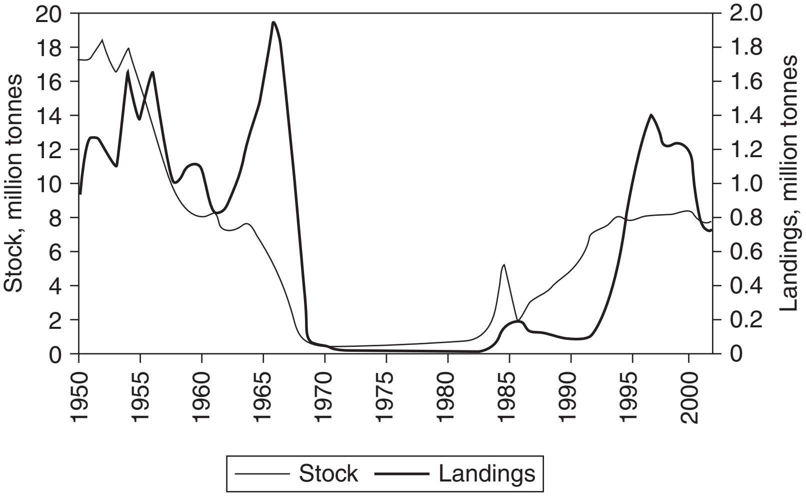
For most stocks in the North Sea[2] the agreement between Norway and the EU seems to have held up well, even if the zonal attachment principle is not necessarily compatible with the incentives of all affected countries (Hannesson, 2004). Problems have arisen, however, over North Sea herring. Herring biomass is much influenced by environmental factors, and its migratory behaviour changes as it vacillates between abundance and scarcity. Thus, when the stock recovered in the 1980s from its virtual crash around 1970, it started to migrate farther north and to a greater extent into the Norwegian EEZ. As a consequence, Norway became deeply concerned about having just a 4 per cent share of the quota on the basis of the zonal attachment of the stock during its depressed years. For some time no agreement was in force between Norway and the EU, and Norway fished the stock at will within its own zone after the herring moratorium was lifted in 1984. In 1986 a new agreement was concluded, giving Norway shares of 25, 29 or 32 per cent, depending on the size of the spawning stock (Engesæter, 1993).

As the North Sea herring example indicates, it is difficult to agree shares of fish stocks whose migratory behaviour and accessibility within individual countries' EEZs changes with stock size. If a stock is confined to a particular country's EEZ when its size is small, that country could have a strong incentive to keep it small in order to prevent it from becoming accessible to others, especially if the other interested parties are recalcitrant in agreeing to a cooperative management plan. In any case, it is likely that the 'core host' country would have a clear advantage when it comes to sharing the stock, because the minimum share it would be willing to accept would be determined by what it would obtain if it had the stock to itself.

Norwegian spring-spawning herring appears to be a stock of this nature. The stock spawns off the coast of Norway, and the juveniles feed in the Norwegian EEZ, and in the Russian EEZ when the stock is sufficiently large. The mature stock migrates into the so-called Ocean Loop, an area not covered by the EEZ of any country, and into the Faeroese and Icelandic EEZs (see Figure 2.3). These migrations appear to be related to how large the stock is. This chapter analyses the strategic advantage Norway might have from being in a position to reserve the stock for itself by fishing it down to the level above which it starts to migrate out of the Norwegian EEZ. This does not necessarily imply that Norway would have an advantage in not sharing the stock with others, but the issue is likely to have a bearing on how the catches from the stock should be shared and what interpretations should be given to the zonal attachment principle, or even whether that principle is of any value in determining how the stock should be shared. Lastly, this chapter considers how a change in ocean climate, such as might result from global warming, could affect Norway's strategic position and the sharing of the stock. An increase in ocean temperature east and north of Iceland would most likely increase the extent of migration of the stock by increasing the supply of plankton in the area where the herring stock is now rarely found, but where it was abundant in the middle of the last century.

NORWEGIAN SPRING-SPAWNING HERRING

The story of the Norwegian spring-spawning herring over the past 50 or so years is a dramatic one. It is thought that the stock in 1950 may have reached almost 20 million tonnes. Figure 3.1 shows the stock's estimated abundance and the catches. In the 1950s and 1960s, annual catches generally exceeded 1 million tonnes, probably substantially more than the stock could sustain. In the late 1960s it crashed, and biomass in 1972 is thought to have been less than half a million tonnes.



Source: ICES (2003), Table 3.3.3.

Figure 3.1 Stock size and catches of Norwegian spring-spawning herring in the years 1950–2002

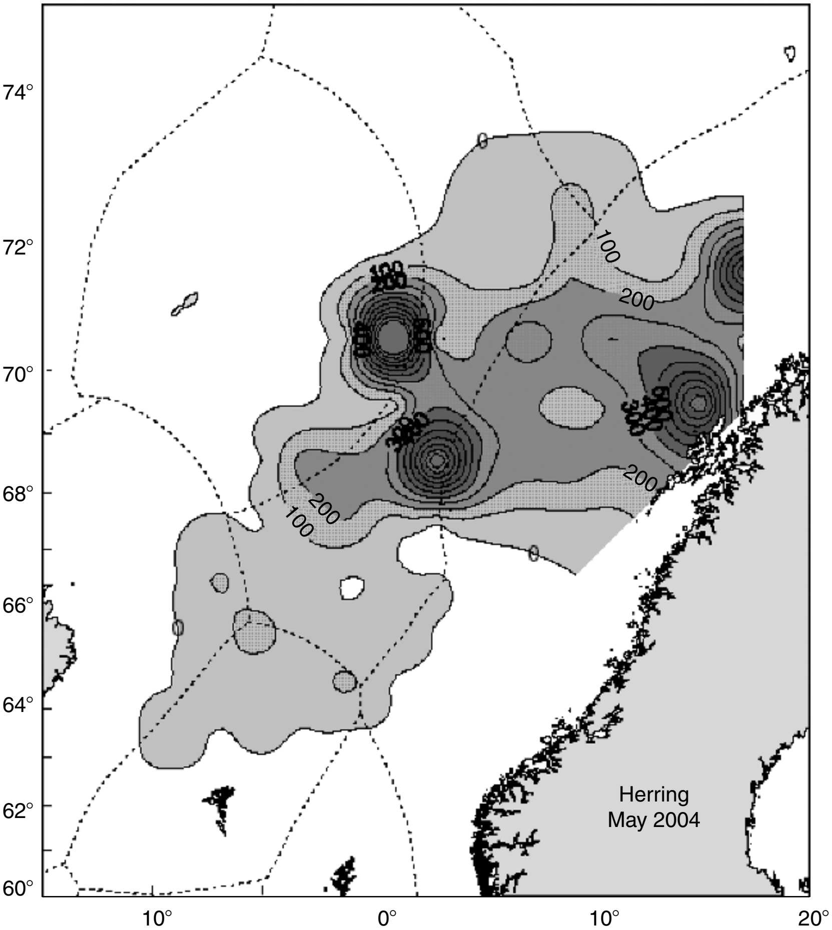
The migrations of Norwegian spring-spawning herring are complex, but are undoubtedly influenced by conditions in the ocean and, apparently, also stock size. Until the early 1960s the stock foraged off the north coast of Iceland in summer (May–September), wintered east of Iceland (November and December), and migrated to the west coast of Norway to spawn in February and March.[3] The young, immature year classes foraged in Norwegian coastal waters and in the Barents Sea. The summer feeding was in relatively warm 'Atlantic' water, which in summer is rich with plankton (specifically the copepod *Calanus finnmarchicus*). Then, cooling of the East Iceland Current in the mid-1960s pushed the boundary between Arctic and Atlantic waters eastwards (Malmberg, 1969; Malmberg and Jónsson, 2002), and the feeding migrations in spring and summer were diverted north towards Spitzbergen. After the crash around 1970 the stock ceased to migrate out of the Norwegian EEZ, wintering around Lofoten, but for the most part spawning farther south off the coast of Møre (see Figure 2.3). It is clearly tempting to explain the less extensive migrations of the herring stock after 1970 by a lessened need for food by a smaller stock.

After the collapse, fishing of the stock was banned, except for a small Norwegian catch of just 7000–20 000 tonnes annually. In 1985 the stock showed signs of recovery, and boats from Russia began to fish the stock. As it recovered further, after a short hiatus, catches by nations other than Norway also increased. From 1994, fishing boats from Iceland and the Faeroe Islands operated on the stock, and a year later boats from member countries of the EU began to do so as well.

The increasing fishing activity of nations other than Norway was inspired by the fact that, as stock size increased, fish began to migrate out of the Norwegian EEZ. This 'spillover' first occurred into the Soviet zone in the Barents Sea, but as the stock increased more it started to migrate into the high seas and into the Faeroese and Icelandic EEZs. The Faeroese and the Icelandic fisheries were partly inside their EEZs, but were also outside, in the so-called 'Ocean Loop' (see Figure 2.3). Fishing by EU countries was also in the Ocean Loop, because the stock rarely migrates into the EU's EEZ. Figure 3.2 shows the spring migrations of the herring stock in recent years.

Increased fishing by nations other than Norway led to efforts to control the fishery through international agreements. In 1999 an agreement was reached among all interested parties on the total quota to be allowed and its distribution. In 2003 this agreement fell apart, however, owing to disputes among the countries involved about the division of the total catch quota.

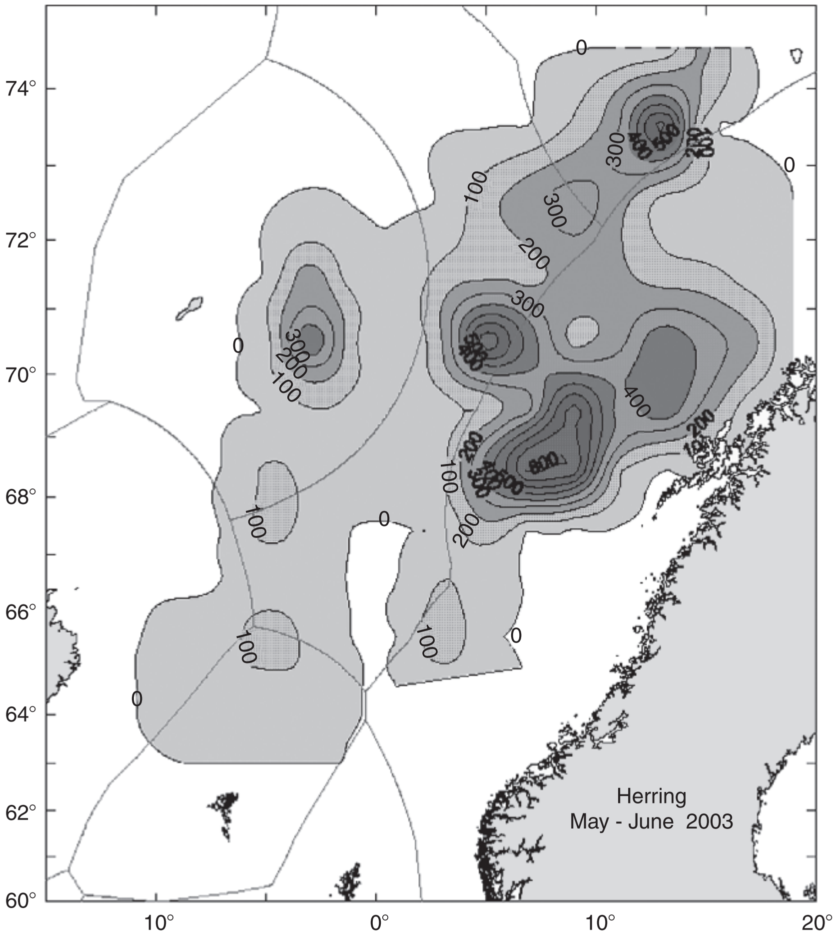
As the stock migrates further and wider the larger it is, the Norwegian share of the total catch depends on the size of the stock. The Norwegian share of the total catch taken from the stock is shown in Figure 3.3,



Source: Report on surveys of the distribution, abundance and migrations of the Norwegian spring-spawning herring, other pelagic fish and the environment of the Norwegian Sea and adjacent waters in late winter, spring and summer, various years. ICES, Copenhagen.

Figure 3.2a Spring distribution of the Norwegian spring-spawning herring, 2004

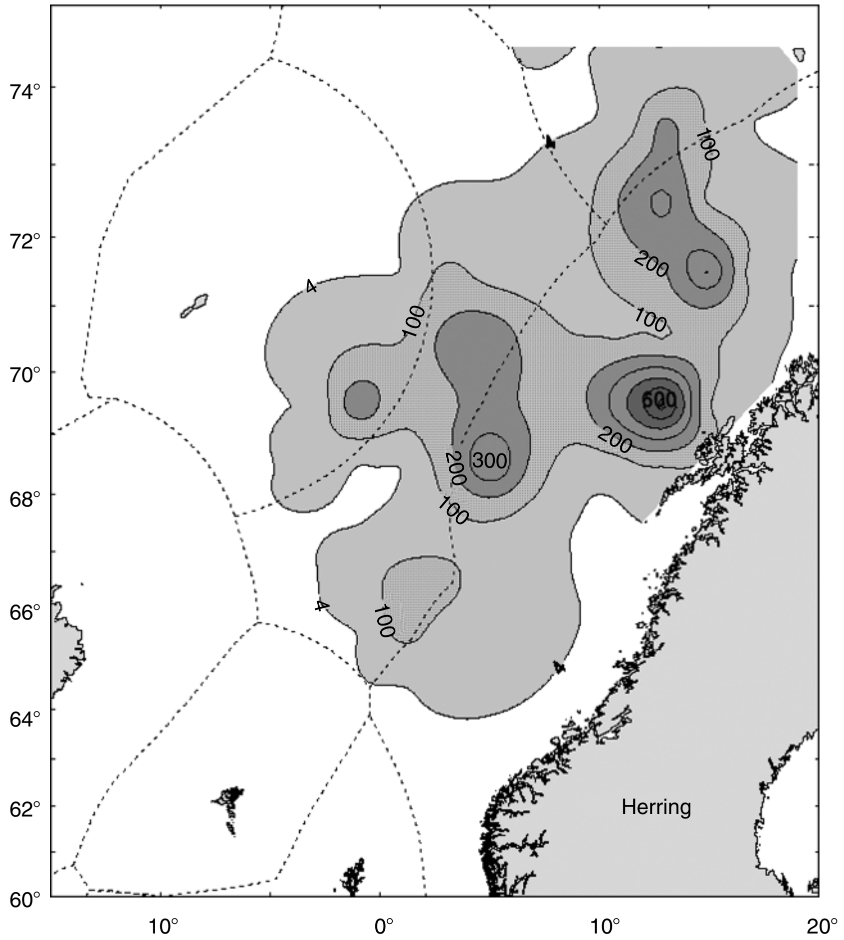
together with the stock size. The share fell to about 85 per cent in the late 1980s when the stock exceeded 2 million tonnes, then stayed relatively constant despite continued growth until it fell rather precipitously to about 60 per cent in 1995. Since then the stock has stayed relatively constant at about 8 million tonnes, and so has the Norwegian share of the



Source: See Figure 3.2a.

Figure 3.2b Spring distribution of the Norwegian spring-spawning herring, 2003

catch. It would seem, therefore, that Norway has a dominant strategic position in the game concerning herring and could elect to fish down the stock to a level that would prevent it from migrating outside its EEZ. While this is not necessarily the best strategy for Norway to follow, it is likely to be important for the way in which the spoils of the cooperative play would be divided.

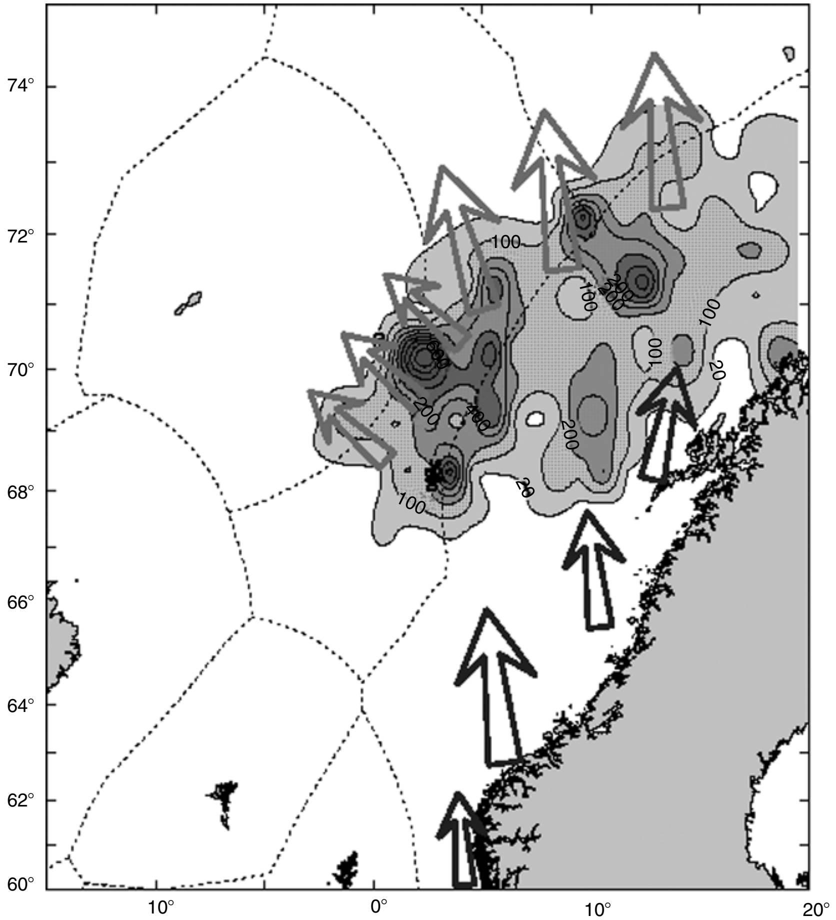


Source: See Figure 3.2a.

Figure 3.2c Spring distribution of the Norwegian spring-spawning herring, 2002

PREVIOUS LITERATURE ON THE HERRING GAME

Two recent papers address the exploitation of Norwegian spring-spawning herring from a game theory perspective (Arnason *et al.*, 2000; and Lindroos and Kaitala, 2000). Both focus on the forming of coalitions among the exploiting nations and on the scope for attaining a globally cooperative

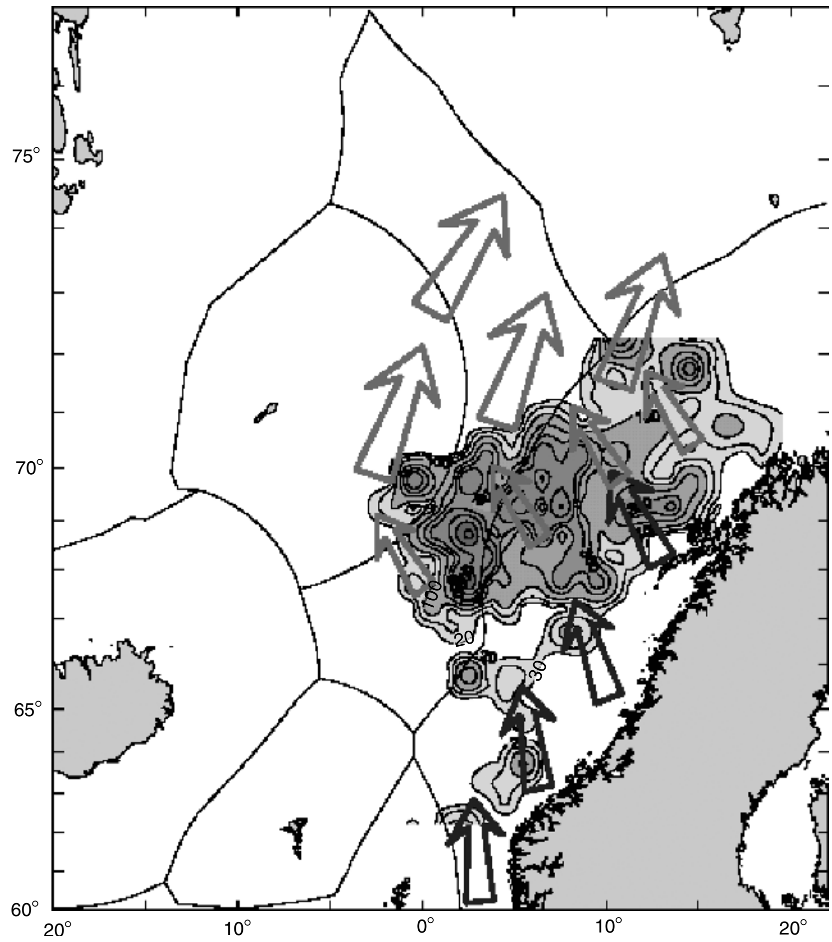


Note: The arrows show the inferred migration pattern in April (inward arrows) and June (outward arrows).

Source: See Figure 3.2a.

Figure 3.2d Spring distribution and migrations of the Norwegian spring-spawning herring, 2001

solution, but they take different approaches. Lindroos and Kaitala (2000) identify three players, Norway/Russia, in whose zones the stock spawns and the young fish grow, Iceland/Faeroe Islands, into whose zones the stock migrates, and the EU countries, which can fish the stock in international waters outside 200 miles. They use a year-class model and assume that the

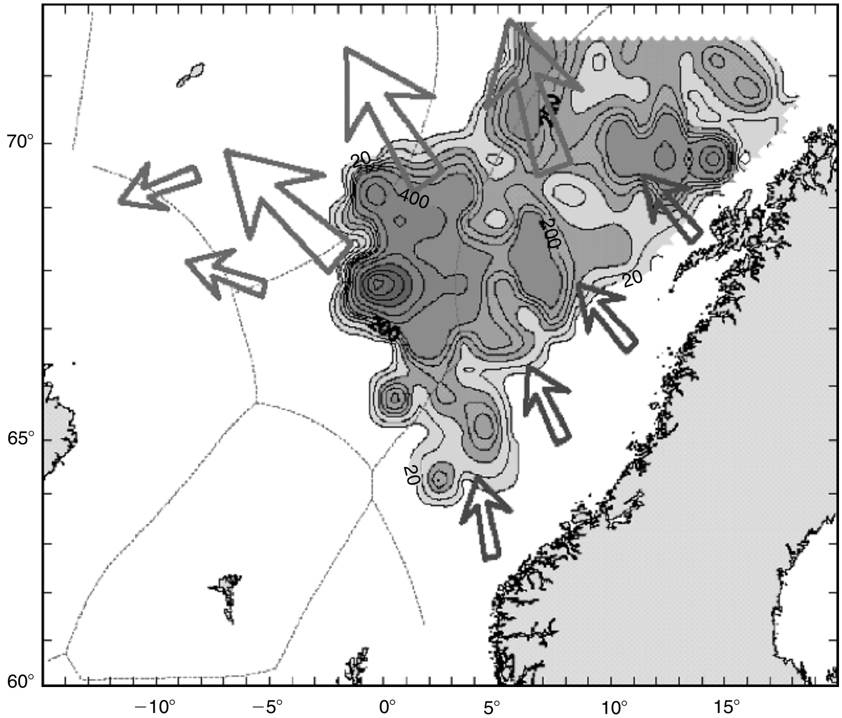


Note: See Figure 3.2d.

Source: See Figure 3.2a.

Figure 3.2e Spring distribution and migrations of the Norwegian spring-spawning herring, 2000

unit fishing costs are inversely related to the exploited stock, an assumption that is probably unrealistic for herring. They find that a fully cooperative solution is unlikely except if the fishery is rather inefficient (low catchability coefficient). Arnason *et al.* (2000) use a general biomass model and consider explicitly the migrations of the stock between the economic zones of different countries, as well as its migration into the international area outside 200 miles. They assume that the cost of catching herring is unrelated



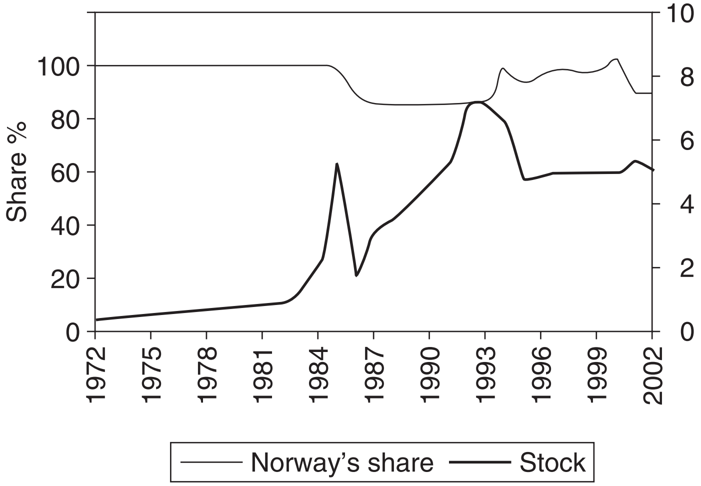
Note: See Figure 3.2d.

Source: See Figure 3.2a.

Figure 3.2f Spring distribution and migrations of the Norwegian spring-spawning herring, 1999

to the size of the stock, but is related to the distance of the fishing locations from the home port of the boats. They identify five players, Norway, Russia, Iceland, the Faeroe Islands and the EU, and find that a globally cooperative solution would require side-payments to Norway, and that many of the potential coalitions in the game would not be viable.

Neither of these papers considers the possibility that the stock would be confined to the Norwegian EEZ provided it is small enough, making it possible for Norway to act as a sole owner and, in its own interest, conserve the stock just below this critical level. This would seem sufficient to prevent a nearly total depletion of the stock that would occur in some scenarios of the Lindroos and Kaitala model. The model of Arnason *et al.* gives advantage to Norway by letting the stock originate in the Norwegian zone, resulting in a need for side-payments to Norway in order to make her interested



Source: ICES (2003).

Figure 3.3 The Norwegian share of the total catch of Norwegian spring-spawning herring and the size of the stock, 1972–2002

in a cooperative solution. These side-payments could take the form of Norway being allowed to fish in the economic zone of other countries (in the spatial model Arnason *et al.* (2000) use, no country can fish in the EEZ of another unless explicitly permitted to do so). In the context of the present chapter, this would mean giving Norway a greater share of the stock than its zonal attachment would dictate.

A recent paper by Bjørndal *et al.* (2004) considers the option of keeping the herring stock in Norwegian waters by fishing it down to 500 000 tonnes, assumed to be critical for migration. This is less than appears consistent with the recent past, and the importance of varying this level, which is in any case uncertain, is not investigated. Bjørndal *et al.* conclude that this option is unattractive for Norway and in fact less attractive than open access with closures. The paper by Bjørndal *et al.* simulates various forms of cooperative strategies and open access and builds on the papers by Arnason *et al.* (2000) and Lindroos and Kaitala (2000). A paper by Sissener and Bjørndal (2005) considers the effects of climate change on the migrations of herring.

This chapter applies a general biomass model of the same type as Arnason *et al.* (2000) and focuses on the consequences of the migratory behaviour of the herring stock being dependent on its size. Although the model in this chapter is applied to one specific stock, the whole approach is suggestive of what might happen in other cases where one country has an advantage similar to Norway's in this case.

THE MODEL

The herring fishery is modelled in discrete time, where fishing occurs at the beginning of each period and growth after the fishery is over, but depending on the size of the stock left after fishing. Formally,

$$X_{t+1} = G(S_t) + S_t$$
$$S_t = X_t - Y_t$$

where X is the stock at the beginning of a period, Y is the catch of fish during that period and $G(.)$ is the surplus growth of the stock (natural growth minus losses). This general biomass model is used for simplicity; in reality the herring stock consists of several year classes, and the surplus growth of the stock consists of the growth of all year classes in the stock in any given time period. In Appendix 3.1, a year-class model meant to reflect long-term (average) conditions is discussed and contrasted with the general biomass model.

The most popular general biomass growth equation is the logistic:

$$G = rS(1 - S/K) \tag{3.1}$$

Used in the above, we obtain the equation to be estimated:

$$X_{t+1} - S_t = \alpha S_t - \beta S_t^2$$

where the parameters of the logistic equation are $r = \alpha$ and $K = \alpha\ /\beta$. A variant of the logistic equation is the asymmetric logistic:

$$G = rS[1 - (S/K)^{\gamma}]$$

The Ricker equation, even if developed for a recruitment relationship, may also be used as a general biomass growth function:

$$X_{t+1} = S_t \exp(a(1 - S_t/K)) \tag{3.2}$$

which can be estimated in logarithmic form as

$$\ln(X_{t+1}) - \ln(S_t) = \alpha - \beta S_t$$

where the parameters are $a = \alpha$ and $K = \alpha/\beta$, where a is the intrinsic growth rate and K is the carrying capacity of the environment.

Data on stock size (X) and catches (Y) of Norwegian spring-spawning herring since 1950 are published in Table 3.3.3 of ICES (2003). From these data, we can calculate the stock left behind after fishing (S):

$$S_t = X_t - Y_t .$$

Table 3.1 shows the results of estimating the logistic equation (3.1) and the Ricker equation (3.2) (t-values are in parenthesis), as well as the implied values of K. Results are shown for the entire period and for two sub-periods. The dividing line between the two sub-periods is the early 1970s, when the stock had collapsed and radically changed its migratory behaviour. It can therefore be argued that the growth parameters of the stock most likely also changed after the early 1970s. The carrying capacity implied by both equations is similar for the entire period and for the period 1972–2002. The carrying capacity for the latter period is close to 10 million tonnes, close to what Arnason *et al.* (2000) found from a similar approach. The results for the entire period after 1950 indicate about twice as large a carrying capacity, which could be due to the much wider migration of the stock in the 1950s and 1960s. This is supported by the logistic equation for the period 1950–1972, which yields a much larger carrying capacity for that period, about 30 million tonnes. The results obtained for the Ricker equation for that sub-period are, however, nonsensical, implying a negative value of the parameter a. In the 1950s the stock was between 10–20 million tonnes (Figure 3.1).

It is evident from the low values of the R^2s in Table 3.1 that neither of the two equations can explain much of the annual growth in the stock. The growth of the herring stock is more influenced by variability of recruitment (that is, the size of new cohorts of fish being added to the stock). This variability is due to environmental factors that are poorly understood and that no model of stock-dependent growth can be expected to explain. That fact notwithstanding, a stock-growth model could make some sense for describing what happens under average conditions. Figure 3.4 shows the calculated annual growth ($G_t = X_{t+1} - S_t$) of the stock and the functions estimated for the period 1972–2002.

Figure 3.4 also shows an asymmetric logistic growth curve estimated by an optimization routine minimizing the sum of squared deviations between the calculated values of G and the values implied by the growth function. The skewness parameter is quite large (6.0), implying a surplus growth function heavily skewed to the right and substantially greater maximum growth than the other two curves, but a smaller carrying capacity. The year-class approach discussed in Appendix 3.1 indicates, however, that the

Table 3.1 Estimation of a surplus growth function for Norwegian spring-spawning herring

Period	Logistic				Ricker			
Year	α	β	K	R^2	α	β	K	R^2
1950–2002	0.14407	−0.00725	19.9	0.012	0.13912	−0.00793	17.5	0.020
	(2.64)	(−1.74)			(2.61)	(−1.09)		
1950–1972	0.06020	−0.00181	33.2	0.037	−0.03806	0.00577	6.59	0.075
	(0.76)	(−0.33)			(−0.84)	(1.28)		
1972–2002	0.36310	−0.03653	9.94	0.040	0.23553	−0.02158	10.9	0.040
	(1.71)	(−1.14)			(2.61)	(−1.09)		

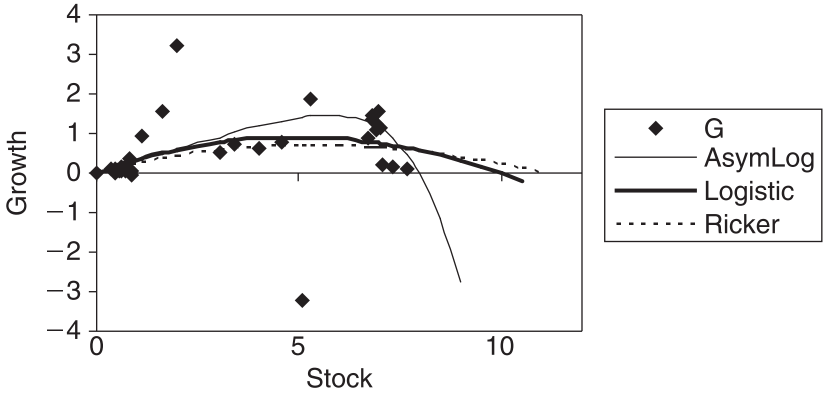


Figure 3.4 Calculated annual surplus growth (G) of Norwegian spring-spawning herring (diamonds) and three surplus growth functions; ordinary logistic, asymmetric logistic and Ricker

surplus growth curve could be skewed to the left and not to the right. In the following we will use the symmetric logistic function with rounded-off parameters of $r = 0.36$ and $K = 10$, because the Ricker function seems to give a maximum sustainable yield that is too low and the asymmetric logistic a maximum sustainable yield that is too large as well as a curve that is skewed in the wrong direction.

The Norwegian share of the stock, or zonal attachment, (u) is assumed to be determined from:

$$u = \begin{cases} 1 & \text{if}\, S \leq \bar{S} \\ 1 - \dfrac{v(S - \bar{S})}{(K - \bar{S})} & \text{otherwise} \end{cases}$$

where $\bar{S}$ is the critical stock level left after fishing, at which size the stock does not migrate out of the Norwegian EEZ, and v is the maximum share of the stock that others could obtain. Hence, the Norwegian share of the catch falls uniformly from 1 as the stock increases from the critical level $\bar{S}$ to the carrying capacity level K, where $u = 1 - v$. This is a simplification of the stepwise relationship produced by the catches in recent years and shown in Figure 3.3.

The herring is a schooling fish, and therefore it may be expected that the unit cost does not depend on the stock size.[4] Under those circumstances, the optimal exploitation of the stock is independent of prices and costs, except that the price must be high enough to cover the unit cost. If the price depends on catch volume, this must hold at the margin. If the objective is maximization of the current value of the fishery over an infinite time horizon, the general condition for optimality is that the marginal rate of surplus growth of the stock should be equal to the rate of interest. Here time-discounting will be ignored for simplicity, it being assumed instead that the goal is to maximize the annual rent from the fishery.

THE COMPETITIVE SOLUTION

Let us look first at a competitive solution where there is no cooperation between the parties and each takes the stock the other parties leave after fishing in their EEZs and in international waters as given. We simplify the setting to two players, Norway and the others. A justification for this is that, in the competitive setting, it turns out that the competitive players have incentives never to leave behind any of the stock that migrates into their zones or into international waters. This is because the stock they do not fish does not stay in their zone, or in international waters, but migrates back into the Norwegian zone, where it winters. This apparently accords with the present behaviour of the stock. At the beginning of the next period (the next spring) the stock migrates and spills over into other countries' zones and into the international area outside 200 miles, according to the share parameters u and v explained above.

Maximizing the rent per year in a steady-state equilibrium entails, for Norway (N) and the others (O) respectively, maximizing each party's share of the sustainable yield:

$$u(S)[G(S)+S]-S_N$$
$$(1-u(S))[G(S)+S]-S_O$$

where $S=S_N+S_O$. The first order conditions are

$$u(S)[G'(S)+1]+u'(S)[G(S)+S]-1=0$$
$$(1-u(S))[G'(S)+1]+u'(S)[G(S)+S]-1=0$$

Obviously these cannot be satisfied simultaneously, except if the stock is shared evenly among the two parties ($u=0.5$). While Norway will not fish all of the stock in her zone, the others will take everything that moves into their zone or on to the high seas. They will still be able to benefit from what Norway does not fish, because some of the stock that emerges at the beginning of each period will migrate out of the Norwegian zone, provided that the stock Norway leaves behind after fishing is sufficiently large. In the non-cooperative setting, the others do not get a high enough return from any fish they might choose not to catch.

Figure 3.5 shows how the optimal stock for Norway to leave behind after fishing changes as the maximum share of the others (v) increases. For a sufficiently high maximum share for the others ($v > 0.6$), Norway keeps the stock down at the level below which it does not spill over into the others' zones or on to the high seas ($\bar{S} = 2$). As v falls below 0.6, Norway fishes a lesser proportion, and some of the stock migrates out of its zone. If the stock always stayed in the Norwegian EEZ no matter how large it was, Norway would operate as a sole owner and leave behind half of the virgin stock after fishing.

Before examining the cooperative solution, let us look briefly at how the non-cooperative solution changes as a result of varying other parameters. Figure 3.6 shows how the optimal stock to be left behind by Norway varies with the level at which the stock starts to migrate out of the Norwegian EEZ. If this critical level is 3 or more, the optimal Norwegian strategy is to keep the stock at this critical level and prevent it from migrating out of its zone. If the critical level is 2 or less, it is optimal for Norway to leave behind slightly more than this (the critical level appears close to 2 in Figure 3.3).

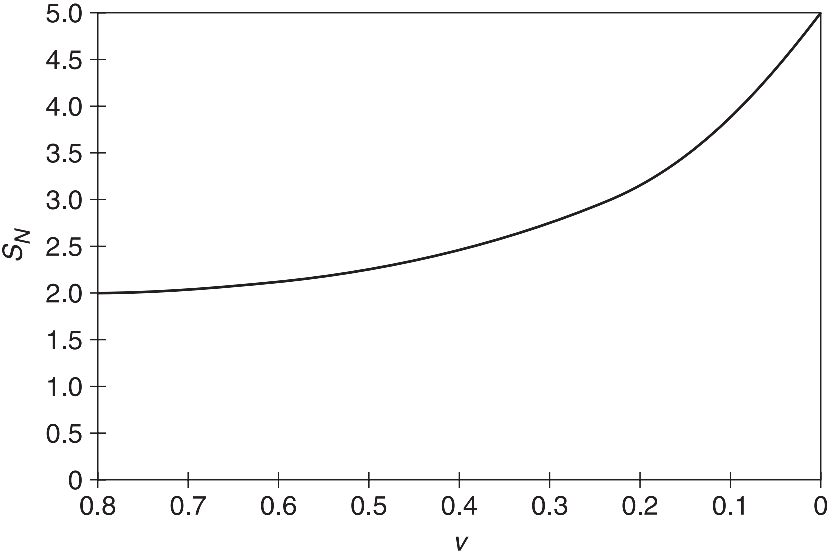


Figure 3.5 The stock (S_N) optimal for Norway to leave behind in a competitive solution as a function of the maximum share (v) other players could get of the stock

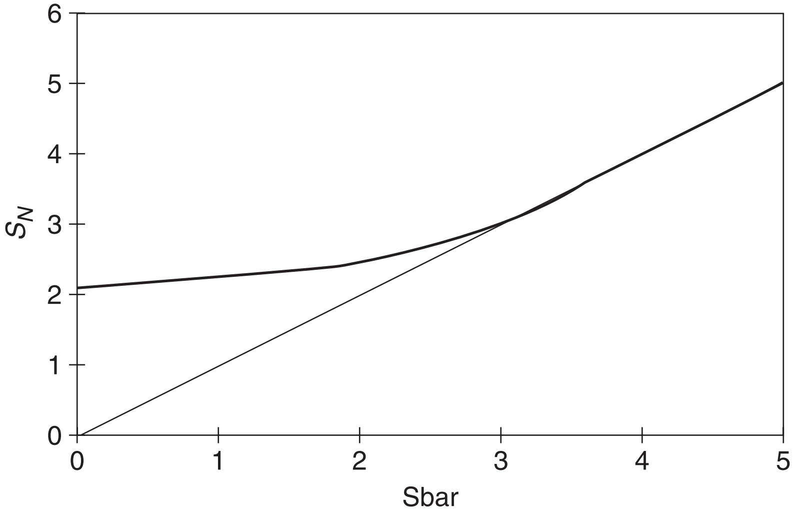


Figure 3.6 The stock (S_N) *optimal for Norway to leave behind in a competitive solution as a function of the critical level* ($\bar{S}$) *at which the stock starts to migrate out of the Norwegian EEZ*

Hence, if the most likely critical level is 2 and the maximum share others could ever take is about 40 per cent, which seems to correspond to current practice and circumstances, then the optimal non-cooperative strategy for Norway would be to keep the stock slightly above this critical level and to allow limited migration of the stock out of her own zone.

Now consider the sustainable catches in the non-cooperative solution. Figure 3.7 shows how the catches of Norway and the others depend on the maximum share (v) the others would ever have of the stock, given that $\bar{S} = 2$. What is particularly interesting is that the catches taken by the others will be greatest if their maximum possible share of the stock is low (between 10 and 20 per cent). This is so because, if they have a low share, the dominant player (Norway) will have a strong incentive to leave behind a large stock after fishing, because she will reap most of the benefits of this herself. On this unfished portion, the others are able to free-ride. As the maximum stock the others will ever have increases, their actual catches decrease, because Norway then has an incentive to fish down the stock in order to reduce herring migrations out of her zone. If the maximum share of the others is 70 per cent or more, the best strategy for Norway would be to fish the stock so heavily that it never migrates out of her zone.

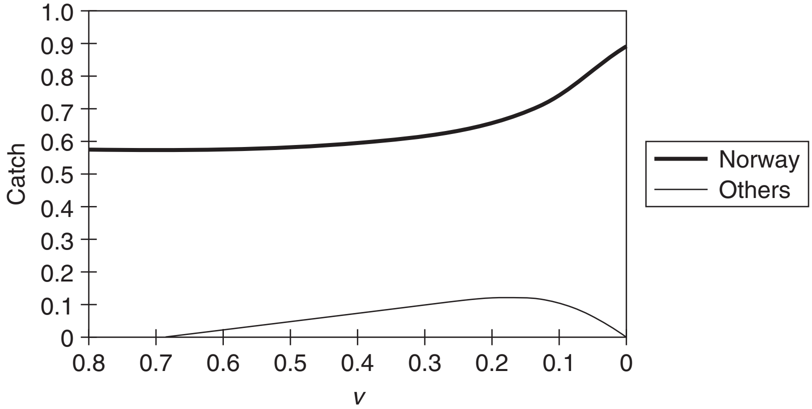


*Figure 3.7 The catch taken by Norway versus other players in a competitive solution as a function of the maximum share (*v*) the other players would ever have of the stock*

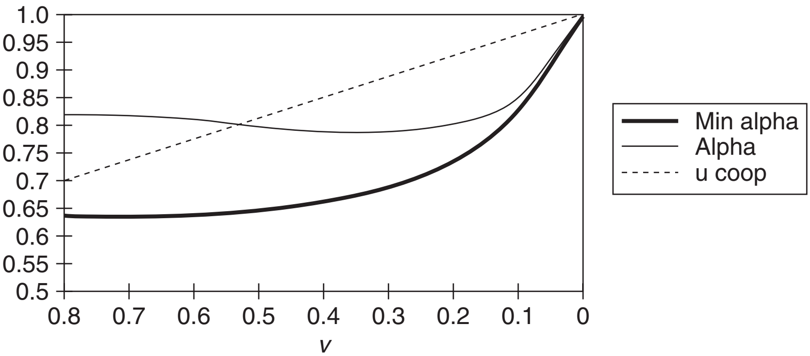
THE COOPERATIVE SOLUTION

In a cooperative solution the parties would maximize the sustainable yield, which in this case is 0.9. Agreement would have to be reached on how the gains from this were to be shared. One possible solution is the Nash bargaining solution, by which the parties first maximize the gain to be obtained, then share it evenly. The opportunity cost of the cooperative solution is the catch obtained in non-cooperative play, which is clearly much higher for Norway than for the others. Sharing the gains evenly thus implies

$$0.9\alpha - Y_{N,N} = 0.9(1-\alpha) - Y_{N,O}$$

where α is Norway's share of the catch and $Y_{N,N}$ and $Y_{N,O}$ are the catches of Norway and the others in a non-cooperative solution. A likely question would then be what the share going to Norway would have to be at a minimum to make her interested in concluding an agreement on cooperation. Clearly, Norway's share of a cooperative catch would have to give her at least as great a catch as she would obtain from a non-cooperative solution. This defines the minimum Norwegian share:

$$\min \alpha = Y_{N,N}/0.9$$



Note: Alpha indicates Norway's share when benefits are split evenly; Min alpha is the minimum share that must be offered to Norway to make her better off than without cooperation; and *u* coop is the share of the stock in the Norwegian zone in a cooperative solution.

*Figure 3.8 The share of catch going to Norway in a cooperative solution as functions of other players' maximum share (*v*) of the stock*

How these two shares depend on the others' maximum share of the resource is illustrated in Figure 3.8. Norway would have to receive at least 60 per cent of the sustainable yield, and an even split of the benefits would give her a share of about 80 per cent, though both shares approach one as the stock becomes more confined to the Norwegian zone. These shares bear no direct relation to the so-called zonal attachment of the stock, that is, how much of the stock is found inside the two players' economic zones. In a cooperative solution ($S = 5$), the share of the stock in the Norwegian EEZ (the line *u* coop) would increase linearly from 70 per cent when $v = 0.8$ to 100 per cent when $v = 0$, while 97 per cent or more of the stock would be in the Norwegian zone in a non-cooperative solution.

CLIMATE CHANGE

One possible effect of a warming of the ocean north and east of Iceland is that the conditions for herring would improve there. In the 1950s and 1960s and earlier, herring migrated towards the north coast of Iceland and supported a substantial fishery there. Its disappearance from those areas need not be attributed solely to overfishing, but could also have been due to a cooling of the waters there. In the 1960s, both temperature and salinity there dropped precipitously, and at the same time herring stopped migrating so far west (Malmberg, 1969). Shortly after the dip, both temperature and salinity recovered, but apparently not quite to the previous level (see Figure 3.2 of Malmberg and Jónsson, 2002). Nevertheless, the herring have not resumed their previous migrations, but it is possible that a further rise in temperature could stimulate a return to previous patterns of migration.

Three different, but not mutually exclusive, consequences for the herring stock may be envisaged to result from a rising seawater temperature north and east of Iceland. First, the migrations of Norwegian spring-spawning herring could extend farther west and make the stock more accessible for exploitation by non-Norwegian fishermen. Second, conditions for growth could improve, so the maximum stock size could increase. This, in turn, could generate more extensive migration. Third, local spring-spawning herring stocks could emerge. Prior to the herring collapse, some herring spawned off Iceland and the Faeroe Islands, and there are reports that herring spawned off Greenland in the 1930s, a period with a substantially higher temperature in those waters than in later years (Vilhjálmsson, 1997).[5] These local spawning stocks have long since disappeared, but a summer-spawning herring stock remains at Iceland in a healthy condition.

What are the implications of a larger carrying capacity of the environment and more extensive migrations for the sharing of herring? Leaving aside the possibility that local spawning stocks would emerge, suppose that the carrying capacity doubles to $K = 20$. From the earlier discussion, it may be recalled that the stock was between 10–20 million tonnes in the 1950s. This assumption need not, therefore, be unrealistic, even if it is way beyond today's stock size, which is in any case below the carrying capacity of the environment.

In the model used in this chapter, the effect of this change would be to encourage further migration of herring out of the Norwegian EEZ. Figure 3.9 shows that it would still be optimal for Norway in a non-cooperative solution to deplete the stock to a rather low level in order to discourage migration out of its zone, but the stock left after fishing in each period would be higher in this case than if productivity was not so high. Hence, Norway would not deplete the herring stock in her EEZ right down to the critical level of 2 where it ceases to migrate out of the zone.

Another and more troublesome effect of better conditions (greater productivity) is that the other parties would probably have to be offered a more lucrative deal in order to ensure their participation in a cooperative solution. Improved conditions would imply that those parties obtain a larger share of the benefit from the stock left unfished by Norway, but they still have no incentives to leave behind any of the stock in their own zone or in international waters. Figure 3.10 shows the critical share of the maximum sustainable yield that the other parties would have to obtain in order to play

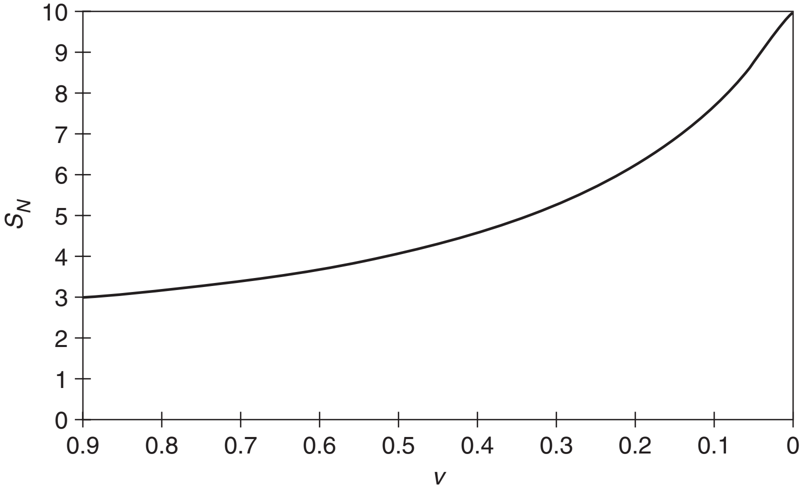
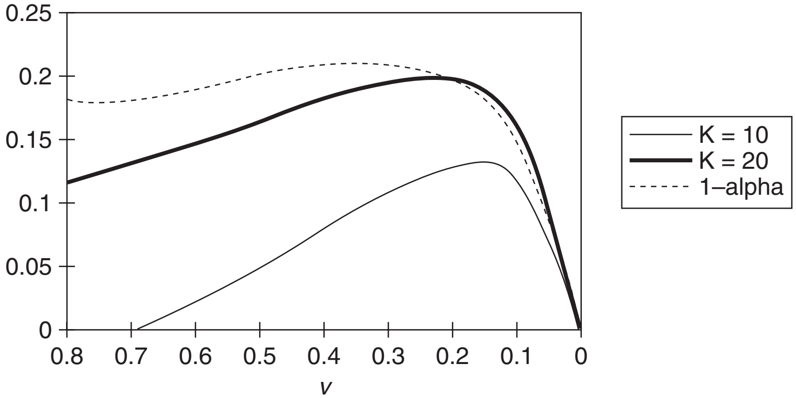


Figure 3.9 The stock (S_N*) optimal for Norway not to fish in a competitive solution as a function of the maximum share (*v*) other players could get of the stock when the carrying capacity is 20 million tonnes*



Note: The solid lines show how the minimum share others will have to be offered in order to accept a cooperative solution varies with their maximum share (v) of the stock, for present ($K = 10$) versus possible future ($K = 20$) carrying capacity. The dotted line ($1 - \alpha$) shows the Nash bargaining solution with the present carrying capacity ($K = 10$).

Figure 3.10 Minimum shares for other countries as a function of their maximum share of the stock cooperatively (this share is calculated in a way analogous to min α above). This critical share rises substantially as K increases from 10 to 20. The figure also shows the other parties' share ($1 - \alpha$) of the cooperative sustainable catch prior to improved conditions in the ocean (that is with $K = 10$) when the gains from cooperation are evenly divided. For some values of v, the critical share under the new conditions exceeds the Nash bargaining share in the cooperative solution under the old conditions, implying that the other parties would gain from reverting to a non-cooperative play unless the share they obtained in a cooperative solution was revised.

Improved conditions for the herring stock are therefore likely to put agreements on cooperation under strain, especially because such secular changes in resource growth may be difficult to distinguish from year-on-year variability, which is substantial and has been ignored here; lasting agreements would have to be concluded on the basis of long-term average conditions, but would come under strain as such conditions changed. The consequences of such breakdowns could be dramatic. Suppose, for example, that the parties have believed in $K = 10$ for some time but that the others have now concluded that the area has become more productive and that they should obtain a bigger share of the stock. After the breakdown, both parties would be likely to revert to non-cooperative play; Norway would reduce its escapement to something like 2–3 million tonnes, the others would catch all they could get, and the stock left unfished would fall by 50 per cent or so, from 5 million tonnes to 2–3 million tonnes, despite the greater productivity and carrying capacity.

ACKNOWLEDGEMENTS

I am grateful to Trond Bjørndal and Sam Herrick for comments on an earlier version.

APPENDIX A3.1: THE SURPLUS GROWTH MODEL VERSUS A MULTI-YEAR-CLASS APPROACH

As mentioned in the main text, the Norwegian spring-spawning herring stock consists of several year classes. It is of interest to check the logistic growth function against the surplus growth that would emerge from a more realistic multi-year-class model. This appendix considers the yield of an average year class of herring over its lifespan. This is the same as the annual yield from a stock in a steady state.

The herring stock consists of 16 or more year classes. Natural mortality, maturity and exploitation pattern are taken from Table 3.4.4 of ICES (2003). A logistic growth function was used to express weight at age (w_t), where t denotes age and $t_0 = 3$:

$$w_t = \frac{w_\infty}{\dfrac{w_\infty - w_{t_0}}{w_{t_0}} e^{-a(t-t_0)} + 1}$$

The parameters of the function were estimated from the weight at age observed for the stock in 2002 (Table 3.2.2.2 of ICES, 2003). The estimation was done by an optimization routine minimizing the sum of squared deviations between the observed weight at age and the calculated weight. The parameters estimated were t_0, a and w_∞. Figure A3.1 shows the values generated by the weight function and the sampled weight at age. The agreement is not poor, but the sampled weight at age follows a somewhat curious pattern, indicating an uneven growth over the lifespan of the fish. This could be due to the age groups having experienced different growth conditions during their lifetime. Therefore, a better result could be expected from fitting the growth curve to the weights of individual cohorts. From Table 3.2.2.2 of ICES (2003), it is possible to identify 40 cohorts, starting at age three. The parameters of the growth curve did not differ much using these data, as can be seen from Table A3.1. In the following, the estimates for the 2002 stock will be used.

Figure A3.2 shows the development of the weight at age of 40 year classes, starting with the year class of 1947 (3-year-old herring in 1950).

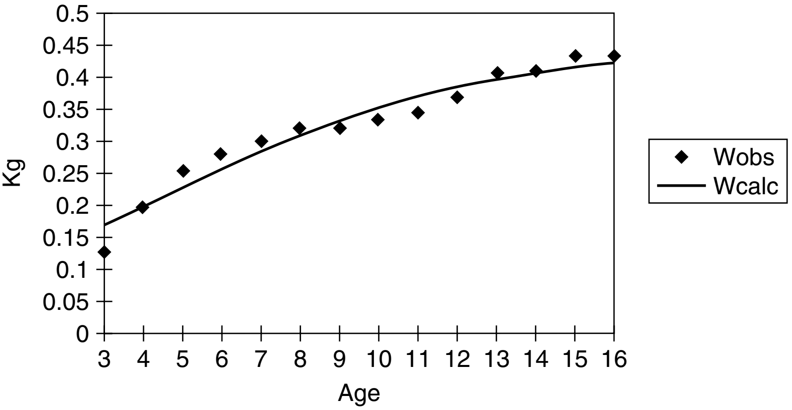


Figure A3.1 Observed weight at age for the stock in 2002 and the logistic weight-at-age function

Table A3.1 Value of the parameters of the logistic weight-at-age function estimated from observations in 2002 and from weights of 40 year classes

	w_{t0}	a	w_∞
Stock 2002	0.1702	0.2624	0.4468
40 year classes	0.1837	0.2715	0.4385

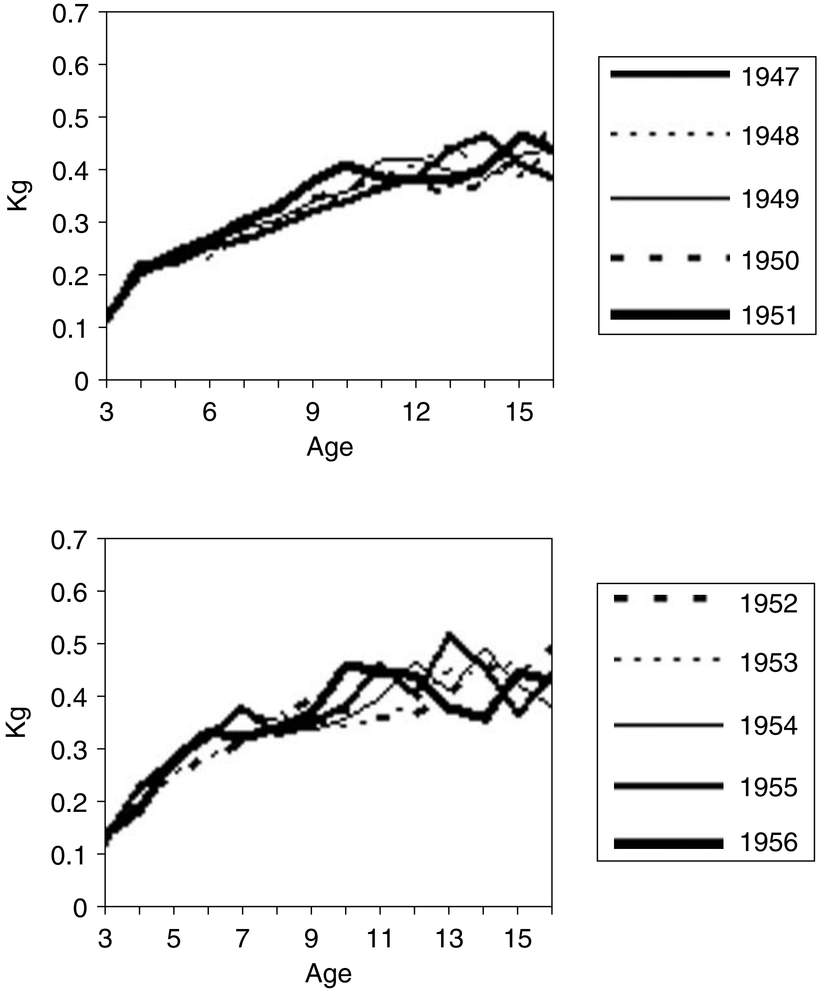
Clearly, the growth of fish can be highly variable; it is not uncommon to see the age-specific weight of a year class drop as it grows older. This could be due to sampling error, but undoubtedly growth conditions vary from year to year. From about 1970, when the year class of 1961 was nine years old, growth apparently became much more irregular; this was the time when the herring stock collapsed. In the 1970s and 1980s, the growth of older year classes was apparently greater than earlier; year classes gained more in weight and the maximum weight at age also apparently rose. It is tempting to conclude that growth is density-dependent; in this period the stock was extremely small (see Figure 3.1). From around 1980, when the 1967–1971 year classes were ten years old or more, there was a sudden and sharp reversal in growth; weight appears to have declined with age for an extended period. It is less tempting here to invoke density-dependence, because the stock grew slowly until the late 1980s. Since about 1990, individual growth has been more regular and similar to what it was in the 1950s and 1960s, and the weight at age has also been lower than in the 1970s and 1980s, about the same as it was in the 1950s and 1960s. Apparently stock size and behaviour are becoming closer to what prevailed in the middle of the previous century, although the migrations are less extensive.

A crucial step in obtaining a surplus growth curve is the link between recruitment of young fish and the size of the spawning stock. As already stated, recruitment of young fish is highly variable; it can vary by an order of magnitude for the same spawning stock. The reasons for this are not well understood, but apparently are related to fluctuations in the marine environment. Figure A3.3 shows the number of recruits plotted against the spawning stock; no correlation is apparent.

A log-linear relationship between the number of recruits and the size of the spawning stock is

$$R_{t+1} = aS_t^{\beta}$$

where R is recruitment and S is the size of the spawning stock. Estimation of the log-linear form gave the results in Table A3.2. This implies a linear



Source: ICES (2003) Table 3.2.2.2.

Figure A 3.2 Weight at age of year classes from 1947–86

relationship between spawning stock and recruitment, but we need a concave recruitment function to obtain meaningful results. The model starts with a given number of recruits (*R*). Depending on total fish mortality (*Z*), these recruits produce a certain spawning stock biomass, referred to here as relationship *f*(*Z*). This spawning stock biomass must produce the number of

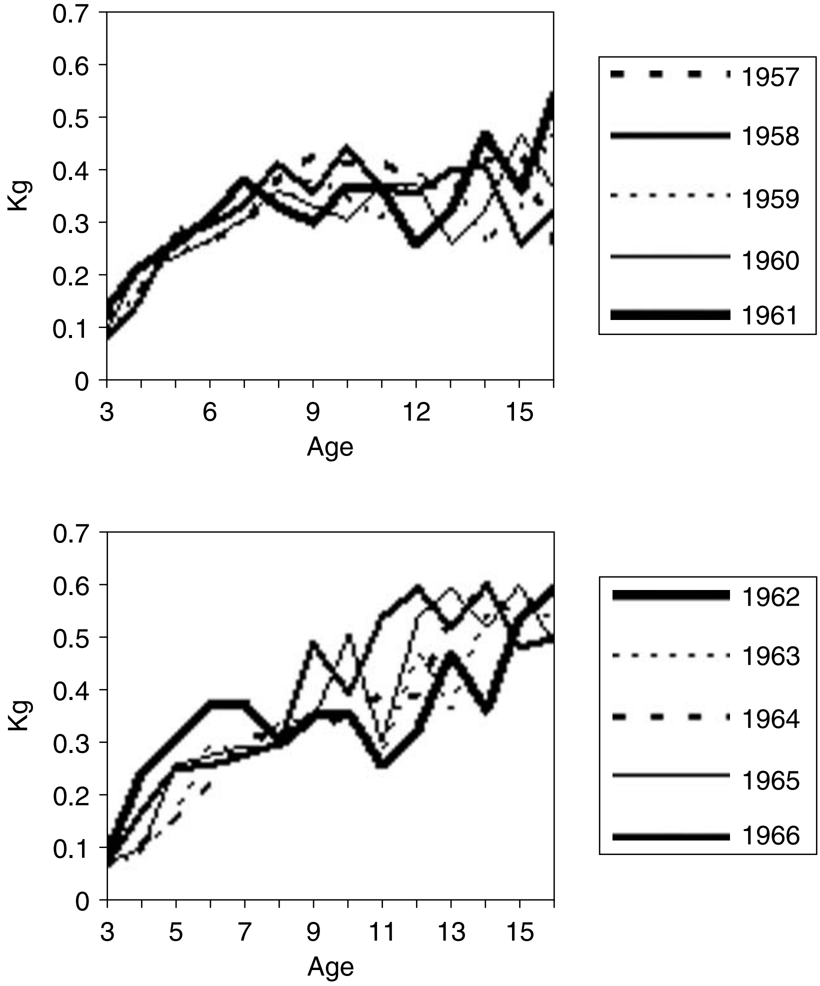


Figure A 3.2 (continued)

recruits we started with through the recruitment function $R(B)$. We thus seek the solution of

$$B - f(Z)R(B) = 0$$

For a given level of Z, $f(.)$ is a constant, so in order to obtain a solution, $R(B)$ must be non-linear, and for a meaningful solution, $R(B)$ must be

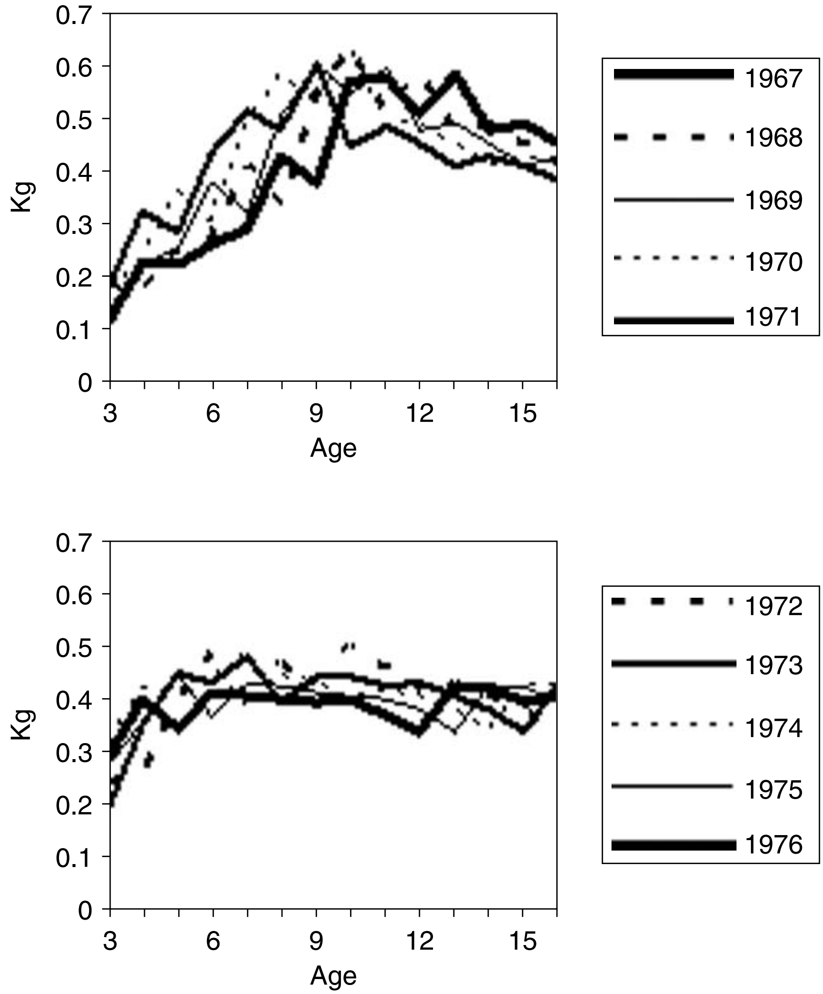


Figure A 3.2 (continued)

concave in B. A frequently used concave recruitment function is the Beverton-Holt function:

$$R = \frac{aB}{1 + B/b}$$

A logarithmic form of this function was estimated with an optimization routine minimizing the sum of squared deviations between observed and

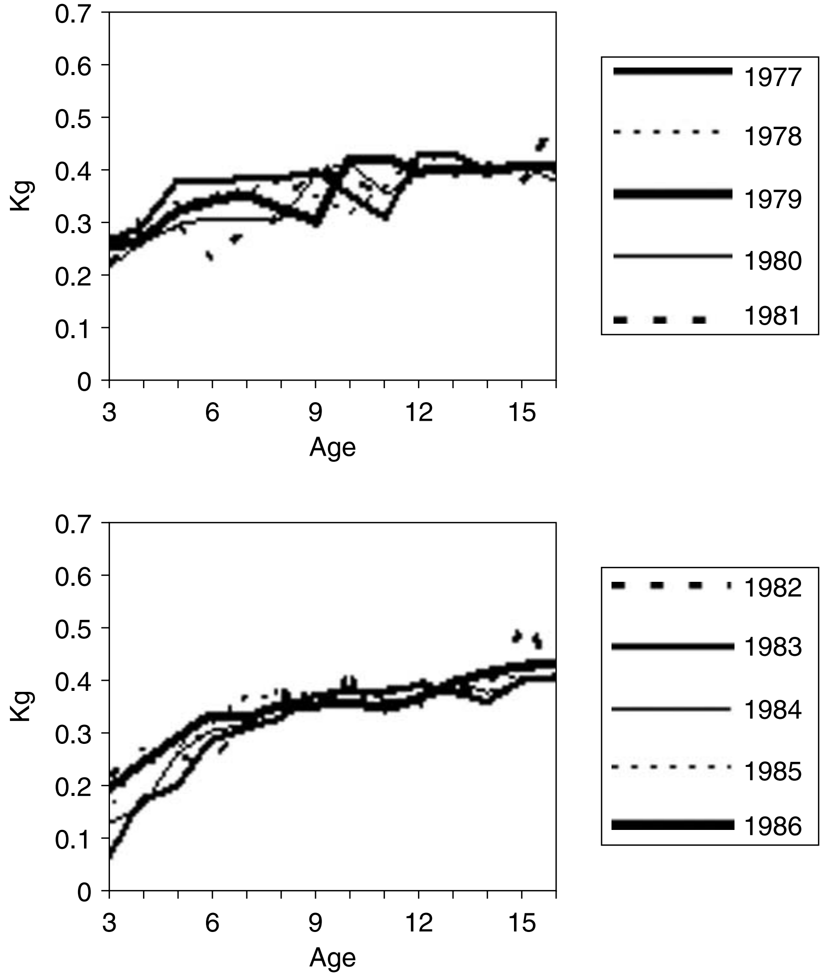
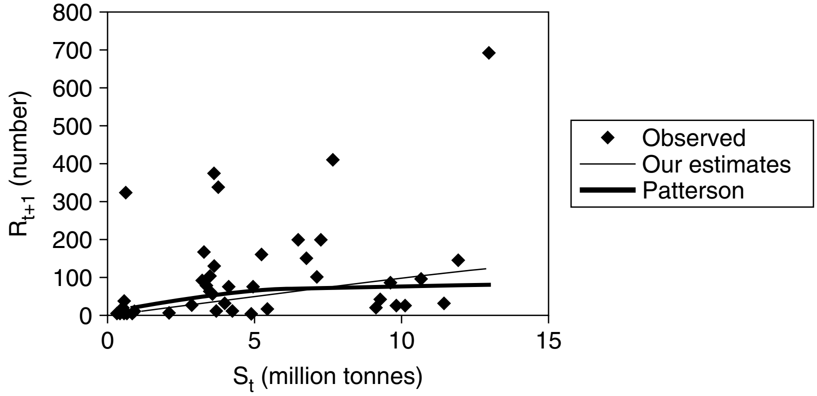


Figure A 3.2 (continued)

calculated recruitment. The parameter values obtained were $a = 11.8$ and $b = 54$. These values differ greatly from the values obtained by Patterson ($a = 32.459$ and $b = 3.044867$), reported in Lindroos and Kaitala (2000, p. 326). Figure A3.3 shows Beverton-Holt recruitment functions with both sets of parameters, together with the observed recruitment. The function using Patterson's estimates is more curved and appears easier to reconcile with some facts about the fishery, as discussed below.



Source: Lindroos and Kaitala (2000), for Patterson's estimates.

Figure A3.3 Recruitment of fish and size of the spawning stock, and two Beverton-Holt recruitment functions fitted to the data

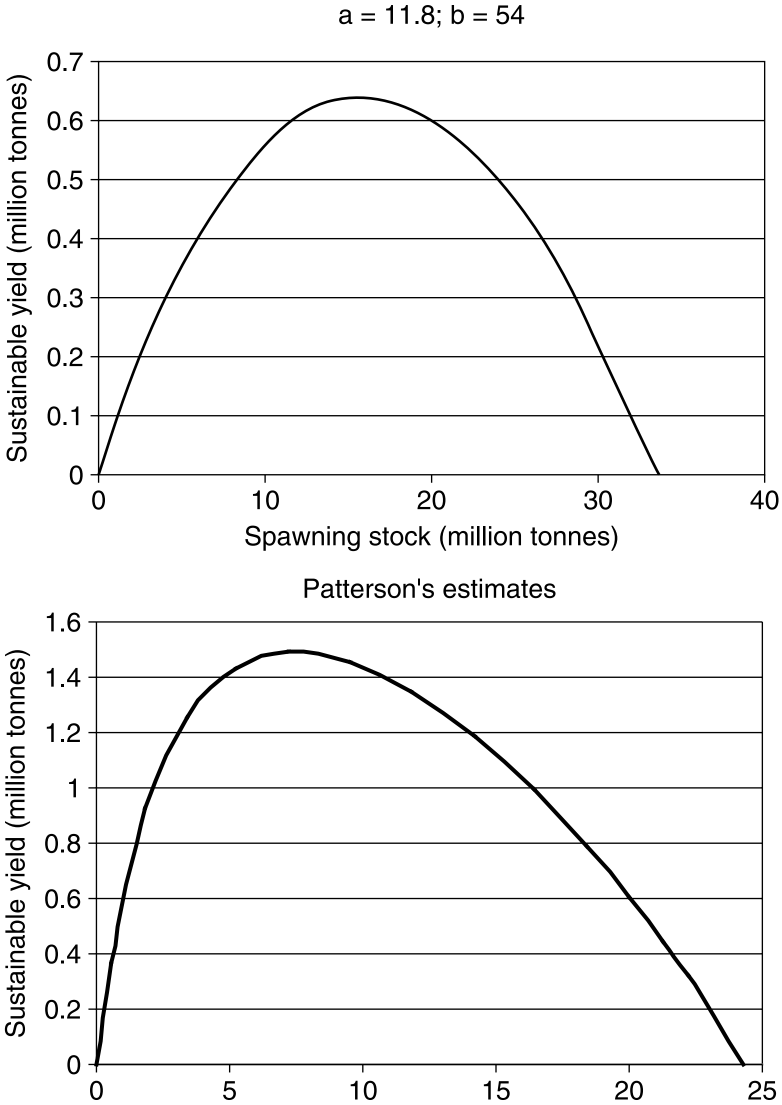
Table A3.2 Estimates of a log-linear recruitment function

α	β	R^2
2.3934 (10.90)	1.0053 (6.70)	0.46

Note: *t*-values in parentheses.

Source: ICES (2003), Table 3.3.3.

Surplus growth as a function of spawning stock biomass,[6] using the Beverton-Holt recruitment function with both sets of parameters, is shown in Figure A3.4. The two curves are remarkably different. The one using the parameters reported above is nearly symmetrical, but implies a rather large maximum biomass (>30 million tonnes) and a low maximum sustainable yield (<650 000 tonnes). The natural mortality assumed for the zero age group until it reaches age 3 is rather high, 2.5, while the total mortality of the zero age group until it reaches age 3, according to Table 3.4.4 in ICES (2003), is 1.8. Using this latter number implies a still greater and less realistic



Source: Lindroos and Kaitala (2000), for Patterson's estimates.

Figure A3.4 Sustainable yield (surplus growth) as a function of spawning stock biomass using a Beverton-Holt recruitment function with the parameter estimates reported in this paper and those of Patterson

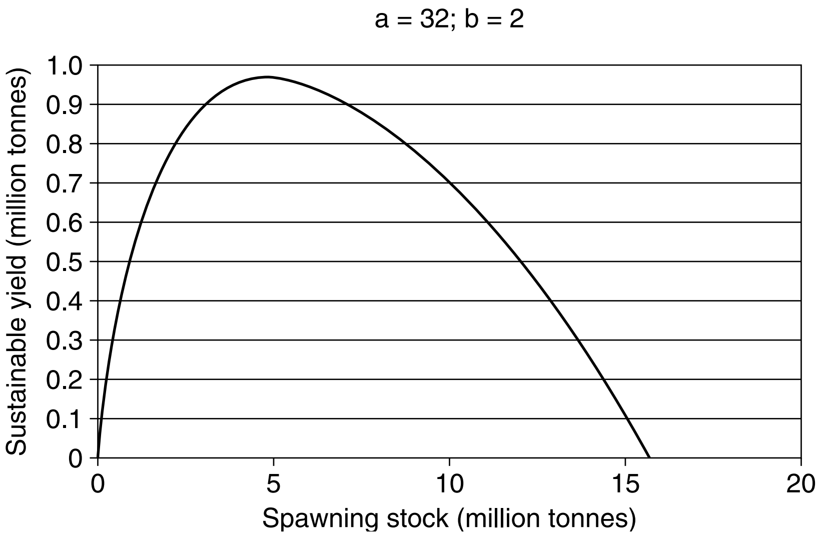


Figure A3.5 A realistic sustainable yield curve produced by a year-class model

maximum biomass. More seriously, the fishing mortality needed to maximize the sustainable yield is extremely small (about 0.03), and a fishing mortality of 0.09 would be enough to wipe out the spawning stock. This is way below the fishing mortality of recent years, which has been at about 0.2 or more.

The sustainable yield curve obtained with Patterson's estimates is markedly skewed to the left. The maximum sustainable yield is markedly higher than in the logistic model, whereas the supporting spawning stock and maximum spawning stock are both lower. The spawning stock biomass providing maximum sustainable yield is about 8 million tonnes, close to the actual biomass in recent years. The unexploited biomass (the carrying capacity) is almost 25 million tonnes, much more than the stock was in 1950. The maximum sustainable yield is more than twice as high as with the other set of parameters, 1.5 million tonnes. The fishing mortality needed to produce maximum sustainable yield is 0.25, which is not unreasonable, although perhaps a little high. The total mortality assumed for the 0-age group until it reaches age three was as in Table 3.4.4 of ICES (2003).

While Patterson's estimates can be more easily reconciled with the facts of the herring fishery, it is possible to find parameter values for the Beverton-Holt recruitment function that would perform even better in this respect and be more in tune with the logistic model used in the main text. Figure A3.5 shows the sustainable yield curve emerging with $a = 32$

and $b = 2$. It has a maximum spawning stock of about 5 million tonnes, like the logistic model, but a larger carrying capacity, because the curve is skewed to the left. It produces a maximum sustainable yield of about 960 000 tonnes, which is slightly more than the logistic model. The corresponding fishing mortality is slightly above 0.2, close to the fishing mortality in recent years. Hence, it is possible to find parameter values that reasonably reconcile the logistic model and the multi-year-class approach. The fundamental problem is, however, what really are normal, average conditions for a fish stock influenced by such enormous environmental fluctuations as is the Norwegian spring-spawning herring. The logistic equation is not necessarily much worse in that respect than the apparently more realistic multi-year-class approach.

NOTES

1. On the concept of zonal attachment, see ICES (1978) and Engesæter (1993).
2. Cod, haddock, saithe, plaice, whiting, sprat and herring (Engesæter, 1993).
3. According to the Report of the Scientific Working Group on Zonal Attachment of Norwegian Spring-spawning Herring, November 1995. The report was produced by a group of marine biologists from Norway, Iceland, the Faeroe Islands and Russia. It has not been published formally, but is obtainable in draft form from the marine research institutes involved in those countries. See also Vilhjálmsson (1997).
4. In a study of the North Sea herring, Bjørndal (1987) reported results that imply a very weak dependence of unit costs on the size of the stock.
5. This complex of stocks used to be called Atlanto-Scandian herring.
6. It was assumed that the spawning stock had been exposed to 20 per cent of the total mortality, as in Table 3.4.4 of ICES (2003).

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4. Rise and fall of the herring towns: impacts of climate and human teleconnections

Lawrence Hamilton, Oddmund Otterstad and Helga Ögmundardóttir

INTRODUCTION

Time plots of catches by fisheries for small pelagic species often show a characteristic pattern. The fishery builds up to a sharp peak of high catches, then drops steeply as the resource becomes scarce. This pattern might occur only once in a fishery's history, or several times with a separation of decades. It is not uncommon for a different small pelagic species to become more abundant, providing a new fisheries target, after the formerly most-prized species vacates its niche. Similar spike-and-collapse patterns can take place in the substitute fisheries as well.

Population volatility appears widespread among small pelagic species. As relatively short-lived forage fish, they experience intermittent strong year classes. Spawning and migration cycles are sensitive to annual-scale variations in ocean environment or climate. Fishing pressure can accentuate this volatility. The characteristic spikes of pelagic fishery catches represent not simply peaks in abundance, as it has been tempting to assume. Rather, they are peaks in fisheries' success, an imperfect correlate of abundance. Unsustainable peaks can result from intensified fishing effort, market demand or technological innovations, even while abundance itself declines. Intensification temporarily masks decline, but catches eventually come down too – often with a crash. Some dramatic failures of twentieth-century pelagic fisheries occurred when rising fisheries pressure coincided with falling environmental conditions, a double blow against a resource.

Order-of-magnitude fluctuations in small-pelagic stocks have consequences on land, where families, enterprises and communities depend on the resource. The human dimensions of pelagic-fishery troubles have been particularly prominent in the case of Norwegian spring-spawning herring, a once-vast stock that during the first half of the 20th century supported fishing communities around the northeast Atlantic, then almost vanished in a late-1960s collapse. With the collapse, herring towns lost their main resource, and faced an urgent need to find other livelihoods. The societal aftermath, as well as the build-up, shows some common elements across different places. Here, we illustrate with the stories of some individual communities – Siglufjörður, a North Iceland village that boomed briefly as the 'Herring Capital of the World'; Seyðisfjörður and Neskaupstaður in the Eastfjords of Iceland, which succeeded Siglufjörður as the centre of Iceland's herring boom during its final stage in the 1960s; and Råkvåg, a quieter Norwegian village where centuries of herring fishing ended with the collapse.

The fisherfolk of these and many other herring towns pursued essentially the same large migratory stock. Adverse environmental shifts around Iceland, together with overfishing (putting pressure on different herring life stages and during different seasons) on both Norwegian and Icelandic grounds, reduced this common stock by more than 95 per cent. Three decades later, the stock had regained only a fraction of its former size and range (for an overview, see Vilhjálmsson, 1997). The shared fates of Icelandic and Norwegian herring fisheries reflect their shared resource.

Signs of synchrony among more distant pelagic fisheries, for example Atlantic and Pacific herring, have also been observed, but their causes are less obvious. One class of explanations looks for teleconnections through global or hemispheric climate, which might impact Atlantic and Pacific ecosystems alike. We suggest an alternative or supplementary hypothesis. The correlations between Atlantic and Pacific fisheries might at least partly be due to humans, and in this respect not so different from what happened to the Atlantic herring towns.

NORWEGIAN SPRING-SPAWNING HERRING

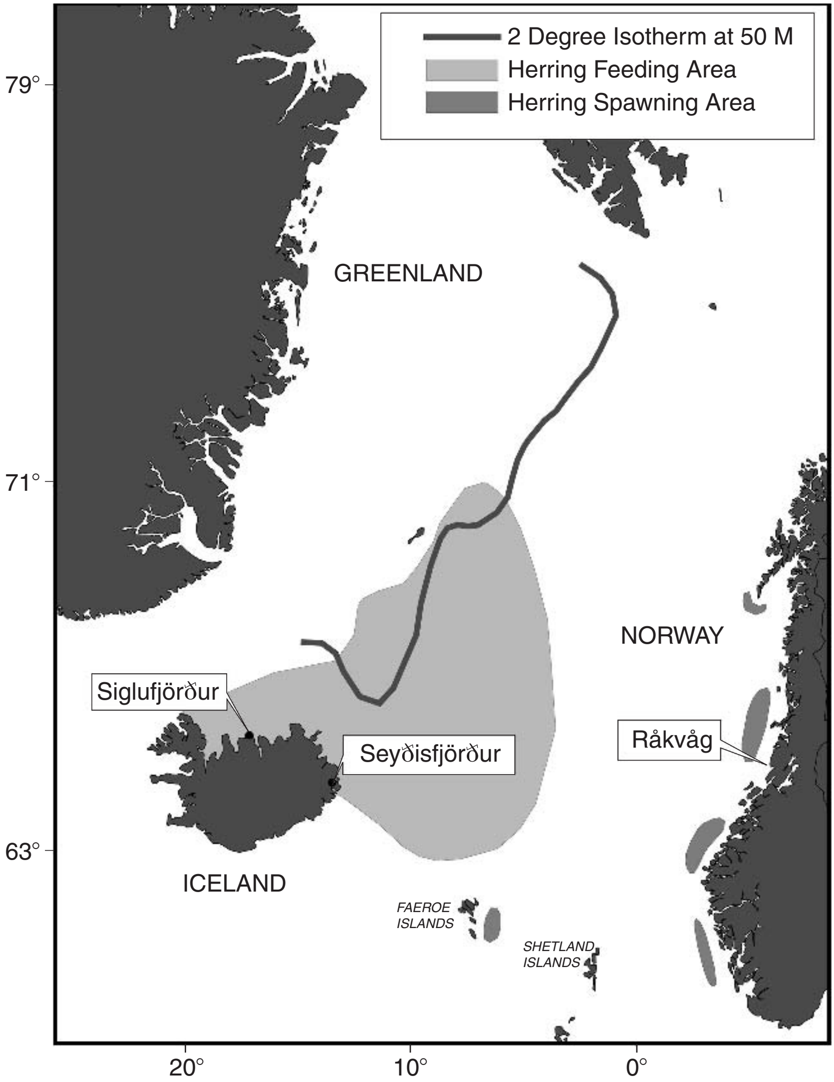
The Atlanto-Scandian herring (*Clupea harengus*) complex consists of several main stocks, the largest of which is Norwegian spring-spawning herring. Through the early and middle 20th century, the stock followed an annual migration around the northeast Atlantic. Typically, most of the stock spawned in spring along the coast of Norway and around the Faeroe Islands (Figure 4.1a). Larvae drifted north into the Barents Sea, and mature fish (and eventually, the younger recruits) made a westward migration to feeding grounds north and east of Iceland. The stock wintered in a small area east of Iceland, migrating eastwards again towards Norway for spring spawning (Vilhjálmsson, 1997). During the 1950s and 1960s, this pattern altered drastically, as the world's greatest herring resource almost vanished (Figures 4.1 b–d). It remained at low levels through the late 1980s, then noticeably started to recover – primarily in Norwegian waters.

Norwegian spring-spawning herring had been known and fished for centuries on their spawning areas along the Norwegian coast. In the mid-19th century, fishers from Norway discovered that the same herring were abundant on feeding grounds north and east of Iceland in summer and autumn (Figure 4.1a). Norwegians initiated a fishery that provided herring-salting jobs, and built wooden houses (an improvement over the Icelanders' mainly turf dwellings) in east Iceland towns such as Seyðisfjörður. Learning from the foreigners, Icelanders began their own fishing company in Siglufjörður on the north coast in 1880. Initially, fishing efforts were concentrated within the fjords. The first peak of Iceland's herring fishery faded in the late 1800s, as the climate worsened and prices fell on European markets (Sigurðsson, 1989).

Herring fishing recovered with warming conditions during the early part of the 20th century. For Iceland, this helped set in motion a remarkable climb from poverty to affluence. Larger vessels using the new purse-seine technology explored offshore feeding grounds and brought back unprecedented catches. The international fishery in Icelandic waters took between 10 000 and 25 000 tonnes per year during the first decades of the 20th century. Initially, Icelandic vessels accounted for only a small fraction of the catch in Icelandic waters, but after 1915 they became dominant. Total catches continued their uneven increase, reaching peaks above 200 000 tonnes several times in the 1930s and 1940s. These good herring seasons contributed to Iceland's achievement of economic, then political independence in the 1940s (Kristfinnsson, 2001).

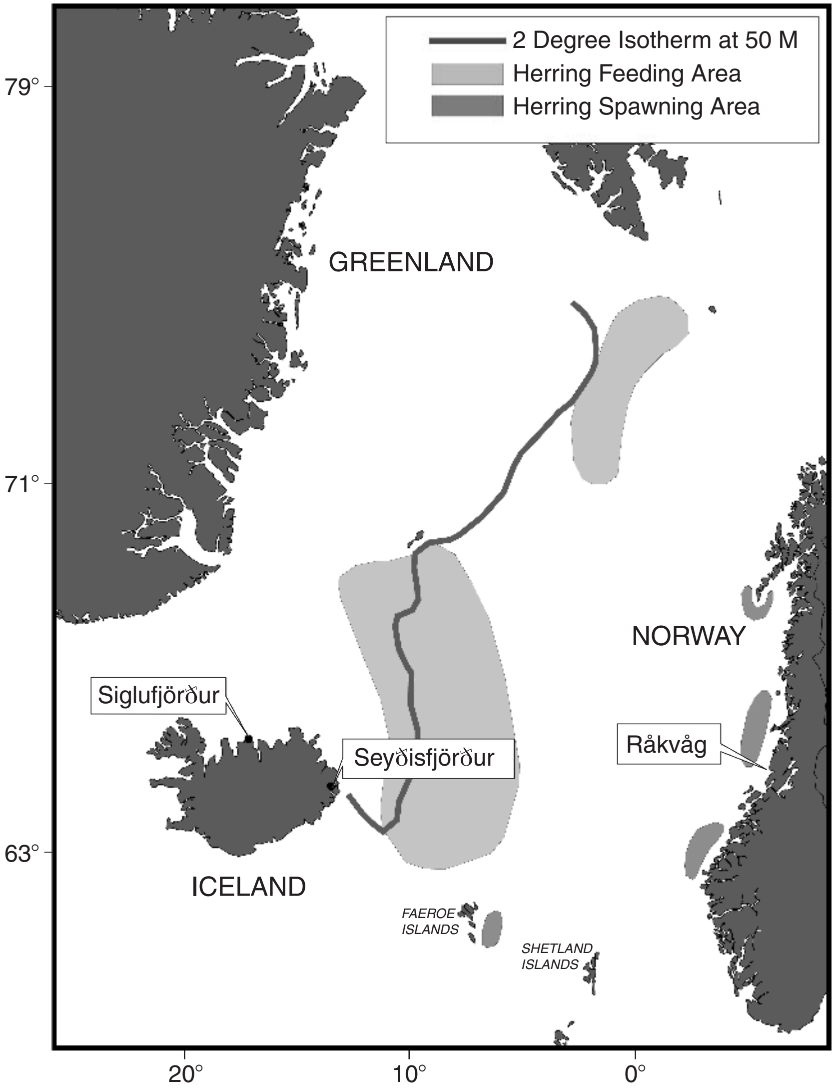
During these years, strong markets, improving technology and increasing effort led to rising success in exploiting the stock throughout the northeastern Atlantic. Total catches fluctuated around a general upward trend, exceeding one million tonnes per year during the 1950s (top graph, Figure 4.2). Expanding markets together with technological innovations – sonar to locate herring schools, and power-block-assisted purse-seines of nylon mesh to catch them – propelled a mid-1960s spike that reached almost two million tonnes.

Collapse followed quickly after this 'killer spike', as catches fell below 100 000 tonnes in 1969 and 10 000 tonnes in 1973. In retrospect, it was clear that the golden years had been times of unsustainable overfishing. Estimated spawning biomass of the spring-spawning stock declined from 14 million tonnes in 1950 to less than half a million tonnes in 1972. As the lower plot of Figure 4.2 shows, the 1960s spike in catches occurred at a time when biomass had already dropped by 74 per cent in just 16 years. Rising catches combined with falling population size to produce an abrupt jump



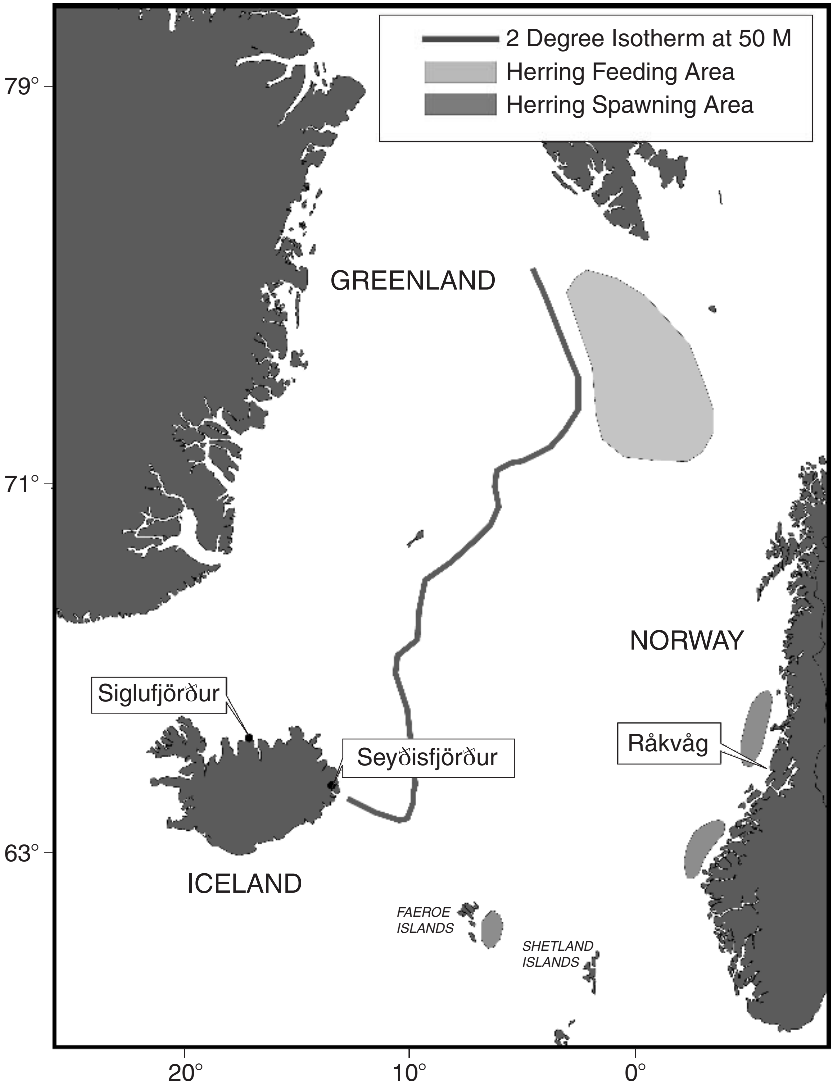
Source: Maps by Cliff Brown, after Vilhjálmsson (1997).

Figure 4.1a 'Traditional' feeding and spawning areas of Norwegian spring-spawning herring



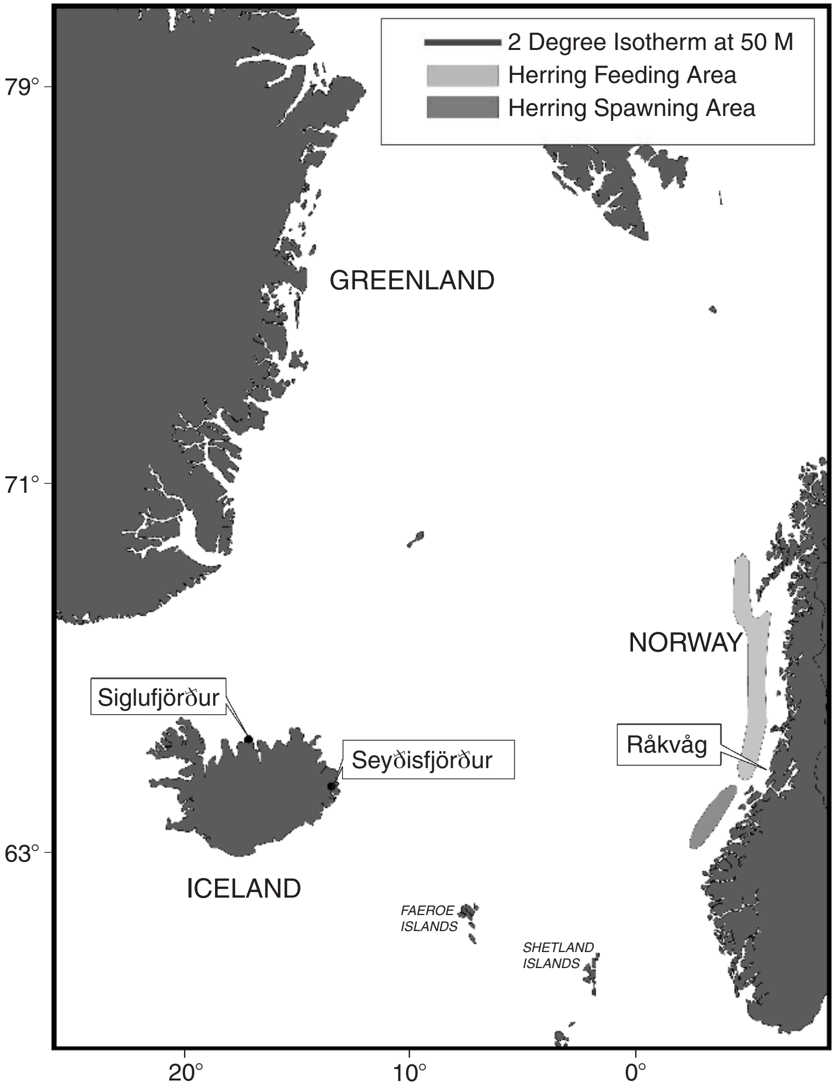
Source: See Figure 4.1a.

Figure 4.1b Feeding and spawning areas of Norwegian spring-spawning herring, 1965–1966



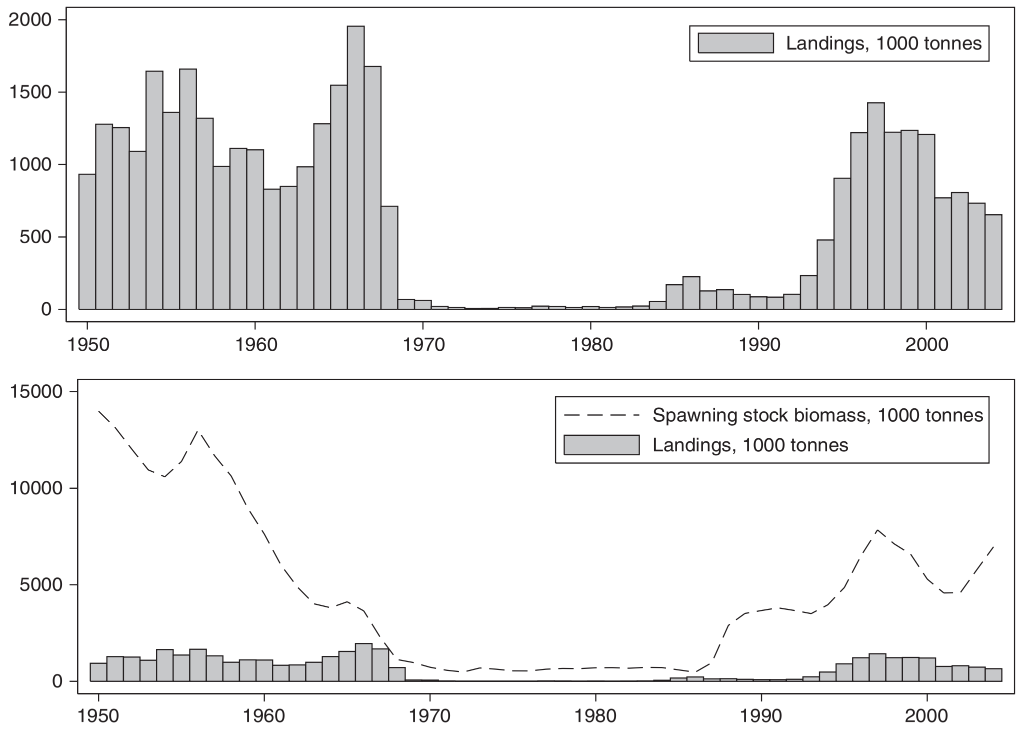
Source: See Figure 4.1a.

Figure 4.1c Feeding and spawning areas of Norwegian spring-spawning herring, 1967–1968



Source: See Figure 4.1a.

Figure 4.1d Feeding and spawning areas of Norwegian spring-spawning herring, 1972–1986



Source: North East Atlantic Fisheries Commission (2004).

Figure 4.2 Norwegian spring-spawning herring total landings (top) and spawning stock biomass (bottom), 1950–2004 in fishing mortality, effectively killing off the resource. Only a coastal remnant of the stock survived around Norway (Figures 4.1b–c). The temporary disconnect between biomass and catches is a key observation from Figure 4.2, and we return to this point later, as it applies to a different ocean.

Overfishing drove the steady decline of herring biomass after 1950. Climate change, however, played a role in the crucial decade of the 1960s. From 1920 until 1965, during the herring fishery's best years, relatively warm conditions prevailed over the northern North Atlantic. Cold, low-salinity Arctic surface water, which formed a boundary for the herring feeding area, generally stayed north of Iceland, as indicated by the 2°C isotherm in Figure 4.1a. In 1965 there was a sudden change, and this front shifted southeast (Figure 4.1b). Northwesterly winds associated with a prolonged negative state of the North Atlantic Oscillation (NAO) drove unusual volumes of polar surface water and ice through Fram Strait into the Greenland and Iceland Seas (Dickson *et al.*, 1988; Hurrell, 1995; Belkin *et al.*, 1998). The cold, stratified water reduced phytoplankton production, and hence the zooplankton on which herring needed to feed (Astthorsson *et al.*, 1983; Astthorsson and Gislason, 1995, 1998). The north Iceland feeding grounds became a virtual desert (Vilhjálmsson, 1997).

The herring thus lost a main feeding area while under intense fishing pressure – annual removals exceeding one million tonnes. This combination of overfishing and environmental change led to a total collapse lasting more than two decades. On land, the herring towns faced crisis.

SIGLUFJÖRÐUR, NORTH ICELAND'S 'HERRING CAPITAL'

In 1890, Siglufjörður was a small-scale farming and fishing settlement, home to fewer than 100 people. Its mountainous surroundings, near-Arctic climate and remote location, at the northern end of the Tröllaskagi peninsula (66.1°N, 18.9°W), made it a poor site for commerce or farming. People fished for cod from small boats and kept a few sheep, or if fortunate, a cow. They gathered hay for the livestock, and sometimes cultivated small potato or vegetable gardens. Ships could land in the fjord, where a small store provided them with goods, and by doing so opened economic possibilities, such as selling salted cod, for local people who were not landed farmers.

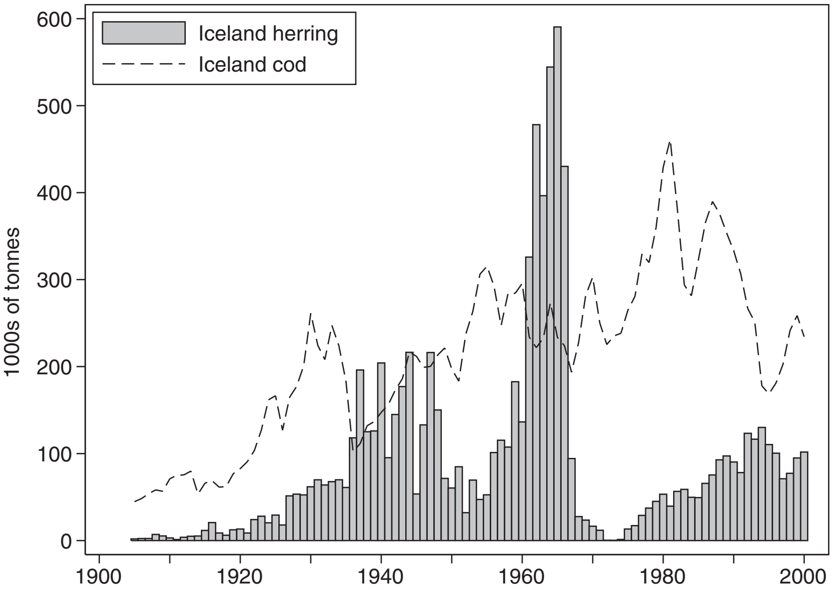
Siglufjörður's location turned out to be perfect for herring, however. During the 20th century, when Norwegian spring-spawning herring were found to be abundant on feeding grounds north of Iceland (Figure 4.1a), Siglufjörður boomed as the 'herring capital of the world'. It grew to become the fifth largest town in Iceland, and an engine pulling the national economy. Then, as the herring declined and ocean/climate change forced herring feeding grounds off to the east (Figure 4.1b–c), Siglufjörður declined too. Eventually, Siglufjörður faded back into minor status, struggling with the common fishing-town dilemma: what to do next, when the best fish are gone?

The story of the herring capital's rise and fall has been described in social-historical terms by Sigurðsson (1989, 1990), and more recently in an interdisciplinary paper by Hamilton *et al.* (2004a). The Herring Era Museum in Siglufjörður, which earned the European Museum Forum's Micheletti Award in 2004, tells the town's story through a rich collection of exhibits, photographs, reconstructions and documents (see Kristfinsson, 2001).

Herring north of Iceland became known to Norwegian fishers in the late 19th century, and in 1903, Norwegians arrived in Siglufjörður to pursue the resource. They established Iceland's first processing factory, reducing herring to fishmeal and oil; the first salting line, to produce high-value fish for human consumption; and the first storage facility, for products awaiting export. Icelanders were hired to work in the new industry, drawn from Siglufjörður and elsewhere. Other herring towns (síldarstaðirnir) in North and East Iceland took part in this boom, but as more factories and salting lines were built, and a growing fleet of foreign and Icelandic vessels brought in the catch, Siglufjörður remained dominant. Through the early 1950s, Siglufjörður was salting more herring each year than the rest of Iceland combined (Sigurðsson, 1990). In several years the herring exports from Siglufjörður constituted more than 20 per cent of all exports from Iceland (Kristfinnsson, 2001).

Figure 4.3 shows the Icelandic herring catches from 1900 through 2000. The golden years of Iceland's herring adventure, boom times for Siglufjörður, form the prolonged but uneven first peak (late 1920s through early 1950s).

The initially labour-intensive fishery made substantial cash wages available to many people for the first time. Farmers denounced the herring towns as they watched their labourers depart for new opportunities. Young Icelandic women, the herring girls (síldarstúlkur), ignored warnings of sin and moved to Siglufjörður, taking arduous but well-paid jobs processing and salting the catch. The town's year-round population increased tenfold (from 144 to 1450) over the period 1903–1924, then doubled again by the late 1930s. The seasonal workforce, arriving with the herring from May through October, added several thousand more. Thousands of foreign fishers, when they came ashore, swelled the population even further. Figure 4.4 shows the dramatic rise and fall of year-round (winter) population in Siglufjörður over the course of the 20th century.

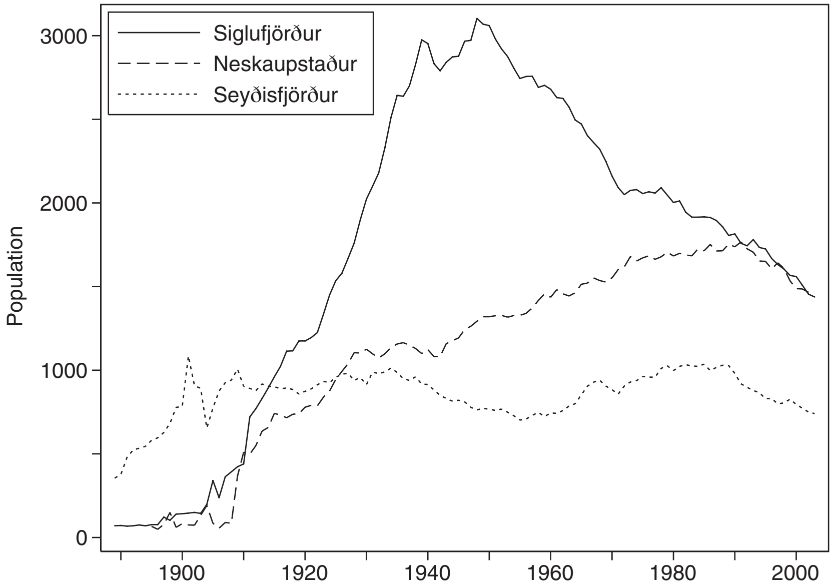


Note: The labour-intensive 'golden years' of the North Iceland herring fishery (late 1920s through early 1950s), and its final more industrial eastern stage (1960s), show up as separate peaks for the herring data. Compare these two peaks with the feeding-area shifts ('traditional' versus 1965–66, and 1967–68) shown in Figure 4.1.

Source: ICES (2001).

Figure 4.3 Icelandic herring and cod catch, 1900–2000

This largely young, unattached workforce found in Siglufjörður a previously unimaginable degree of economic and social freedom. There were opportunities for dancing, music, and entertainment – the town hosted 18 public bars in the 1920s (compared with only two today). There still exists a whole music genre, on recordings and in the older people's memories, of 'herring-waltzes' (síldarvalsar) from the herring years, many describing the atmosphere in Siglufjörður back then. The jobs themselves were demanding, driven by the pace of the fishery. When herring were landed, whatever the quantity, they had to be processed at once. Although work was hard, it was also rewarding to a degree that few participants, and virtually no women, had previously known. Labour in the herring fishery provided many young people from poor backgrounds with savings they would subsequently invest elsewhere in housing, education and new businesses, climbing in one generation into the middle class.



Source: Statistics Iceland (1997, 2002).

Figure 4.4 Populations of three herring towns: Siglufjörður in North Iceland, and Neskaupstaður and Seyðisfjörður in East Iceland

The freshest herring were salted and exported at good prices for human consumption. The refuse and remainder went to the factories for reduction into fishmeal and oil. At its peak, Siglufjörður boasted 27 salting stations and five fishmeal factories in operation, far more than other towns such as Raufarhöfn (ten stations, one factory), Akureyri (six stations) or Húsavík (five stations, one factory) in North Iceland, or their East Iceland counterparts such as Seyðisfjörður (nine stations, two factories) and Neskaupstaður (six stations, one factory) (Kristfinnsson, 2001).

Siglufjörður flourished while herring were plentiful in local waters. Good years became infrequent after 1953. At the same time, factories became more automated, requiring fewer workers. Boats travelled increasingly farther north and east to find the remaining fish, eventually towards Jan Mayen and Svalbard, out of range for small boats, and too far away to bring back herring fresh enough for salting. The golden years in Siglufjörður had been fading for more than a decade, before climatic changes in the 1960s finished the resource off. Through the 1950s and 1960s, the town's population fell steeply (Figure 4.4).

Resource declines caused by overfishing, environmental variation or both are a nearly universal experience among fisheries-dependent communities. When resources decline, some people migrate away, while the community left behind seeks a 'Plan B'. Alternative fisheries, or diversification to other species, often ones that were previously less valued or less abundant, tend to be the first idea. Alternative fisheries hold obvious attractions for fisherfolk, although they might employ fewer workers than the old fishery, and the 'new' species could be subject to depletion as well. Another idea tends to be tourism. The remote locations, rugged geography and narrow resource base of fisheries-dependent communities that could discourage other kinds of development might be turned into attractions to tourists.

Siglufjörður after the herring era has shown all these patterns. The herring resource, for which the town was built, had been eroded by overfishing, then collapsed with environmental change. Outmigration shrank the town's population. Alternative, more diversified fisheries have become mainstays of the smaller economy today – capelin (*Mallotus villosus*), a different small-pelagic species, are reduced for fishmeal and oil, while cod (*Gadus morhua*) and shrimp (*Pandalus borealis*) are landed for export and human consumption. The harbour is quiet and empty, compared with its heyday in the mid-20th century.

To move beyond fishing, Siglufjörður residents are lobbying for a new tunnel to the south, towards Eyjafjörður and Akureyri. One argument for the tunnel is tourism. Siglufjörður's vivid history as the herring capital, with its first-class museum and fine scenery, provide credible tourist attractions, but at present the long drive from the main road or towns remains problematic. A southern tunnel, however, would bring Siglufjörður within easy bus-tour range of Akureyri, the urban centre of North Iceland, with its airport, cruise ships and summer tourists.

For the cycle from small village to prosperous fishing centre, decline, and the search for an alternative, Siglufjörður provides the archetype. Other towns have followed similar cycles, although not often so starkly.

HERRING TOWNS OF EAST ICELAND

Icelandic herring catches, displayed in Figure 4.3, show two distinct high eras. The first was during the 1930s, 1940s and early 1950s. This comparatively low-technology labour-intensive era brought thousands of jobs to Siglufjörður and other herring towns. Then, as herring biomass shrank, and the remaining fish were found farther north and east, the fishery entered a new era. Low-tech inshore vessels could no longer reach the fish.

The great terminal 1960s spike of Iceland's second herring era reflected catches by a more industrialized, long-distance fleet based in East Iceland towns such as Seyðisfjörður (65.3°N, 14.0°W) and Neskaupstaður (65.2°N, 13.7°W). New post-war technologies, the power block, nylon nets and sonar, allowed massive catches and masked the resource decline. Larger ships ranged far to the east and north to find the fish (compare the chronology in Figure 4.3 with the corresponding maps in Figure 4.1).

Contemporary narratives about the last decades of Iceland's herring adventure describe the dramatic shift of fishing activity from North Iceland to the Eastfjords. Social activity, and the eyes of the nation, followed this shift as the herring retreated farther and farther east. When the fishery ended, almost in an instant (1968), the herring's earlier retreat looked in retrospect like a warning of what was just around the corner.

The excellent harbour of Seyðisfjörður had become one of Iceland's first herring ports when Norwegians started fishing there in the late 19th century. By 1901 the population passed 1000, compared with fewer than 150 in Siglufjörður (Figure 4.4). However, the herring catches there declined, and Seyðisfjörður grew no further, whereas Siglufjörður began to boom after 1910. In the mid-1930s two herring plants were built in Seyðisfjörður, and new vessels were purchased. The herring plants often had an insufficient supply of fish, so to raise catches, a trawler was allotted to the town by the government in 1946 (one of several distributed to Icelandic municipalities as a way to provide jobs). Herring salting resumed in 1950, after a lapse of 50 years. One plant was enlarged in 1956, then rebuilt in 1962, as Seyðisfjörður became more important (and Siglufjörður less so) in the east-shifting herring fishery. Seyðisfjörður processed massive volumes of herring during the fishery's terminal spike in the period 1962–1967 (see Figure 4.3), before the resource disappeared.

During the peak year of 1966, Seyðisfjörður processed some 150 000 tonnes of herring and salted 108 000 barrels. Hundreds of students, fishers and others came to work in this short-lived boom; similar opportunities no longer existed in Siglufjörður. Norwegian ships also came to fish, but they processed their herring onboard. Although the eastern boom involved far more fish per year than the northern boom ever had, it created fewer jobs owing to its more modern, industrialized methods.

Following the collapse, Seyðisfjörður's herring plants turned to alternative species. One became a cod freezing plant in 1969; another had little to do for five years until a fishery emerged for capelin. Unlike Siglufjörður, Seyðisfjörður also possessed significant demersal fish resources. In 1972, another trawler was purchased to fish for cod. Cod landings overall increased as herring catches dipped (Figure 4.3), and together with capelin this allowed Seyðisfjörður's population to continue growing after the herring crash, until cod too declined and outmigration became marked (see Figure 4.4).

There was a similar pattern in the nearby herring town of Neskaupstaður. Even during the herring era, many small boats in Neskaupstaður fished for cod and other demersal species, providing jobs in the absence of herring. Working in cod processing was socially stigmatized compared with working in herring, the bigger, more exciting fishery. The herring fishery demanded harder physical work, intense for short periods, but requiring limited skills. It appealed to younger workers, more so than the comparatively stable and technical cod fishery. Having the alternative of cod, however, left Neskaupstaður a way out of the herring crisis. In 1970 Neskaupstaður was among the first towns in Iceland to buy a stern trawler to fish cod, allowing it to bridge the gap between the herring era and what came afterwards – the trawler era.

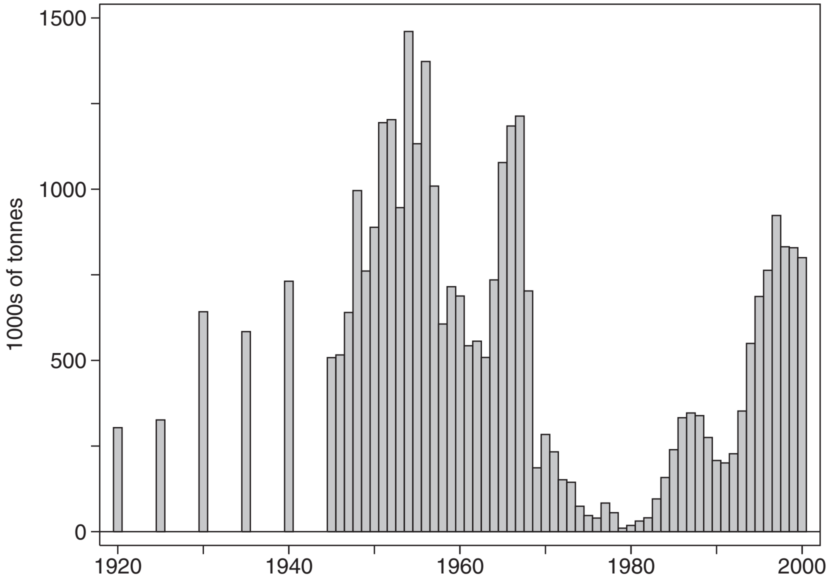
In Seyðisfjörður and Neskaupstaður, both the rise and the fall of the herring era came later and much faster than in Siglufjörður. Because East Iceland fisheries were more diverse and less labour-intensive, immediate socioeconomic impacts of the herring collapse were less harsh. However, when cod catches fell too, a few decades later, the Eastfjords towns were left in similar dire straits.

The herring collapse was a national shock, with impacts not confined to the herring towns. Unemployment increased around Iceland; net outmigration jumped during the years 1969 and 1970 to its highest levels since 1887 (Statistics Iceland, 1997). Herring and cod had been the economy's main pillars; the loss of one highlighted the nation's vulnerability to environmental forces, and the need for diversification beyond fish.

RÅKVÅG, NORWAY

Norwegian landings of spring-spawning herring have a much longer history than Iceland's, but follow a broadly similar 20th century path (Figure 4.5). Many Norwegian towns and villages also suffered as the herring declined, then collapsed. One example is the small rural community of Råkvåg (63.8°N, 10.1°E) on the Fosen peninsula in Sør-Trøndelag county, mid-Norway (Figure 4.1).

The first known period of economic growth based on the herring fishery in the Fosen area was from about 1590 to 1670. There are few data available concerning environmental conditions so long ago. Norwegian books about coastal history (*Trondheim Bys historie*, 1956; Sjurseth, 1961; Bjørkvik, 1972; *Bergen Bys historie*, 1979; Tufteland *et al.*, 1986), or about the history of herring fisheries in Norway (Fasting, 1960; Vollan, 1971) note



Note: For 1920–40, values at 5-year intervals are shown.

Source: ICES (2001).

Figure 4.5 Total Norwegian herring catch, 1920–2000

the boom of the herring fisheries then, and their subsequent collapse. There was evidently a shift in herring migration around 1600, because formerly successful herring fisheries in the area between Denmark and Sweden collapsed at the same time that herring fisheries in Norway bloomed. Danish and Swedish fishers applied to the King in Copenhagen to take part in the Norwegian herring fisheries. The Fosen peninsula with its two western fjords, Stjørna (where Råkvåg is situated, at the mouth of one river) and Bjugn, was the most successful area for herring fishing in the whole of Norway. The population in those two municipalities increased by 250 per cent in the period from 1610 to 1665.

The two fjords are part of a huge geological structure that starts in the west at the Norwegian continental shelf, rises above water around the fjord area and the river basins, and continues far inland as a wall of old bedrock. This westbound geological structure (fault) tends to capture efficiently what comes with the warm North Atlantic Current as it sweeps along the west coast of Norway. Some consequences are unpopular, such as the clouds that tend to collect at the inner end of the fjords. Other consequences are more favourable: the strong circulation of warm seawater into the narrow fjords so that, when herring are abundant, the fish tend to be trapped there as well. One reason for the success of these two municipalities in the old herring fishery relates to characteristics of the shore-seine, which was the most efficient fishing gear at that time. One end of the seine had to be anchored to land, usually at a pier. As soon as the seine was circled around the herring, the other end was dragged to land again either by men or horses. The landscape of Stjørna and Bjugn is ideal for a shore-seine, and every year when the herring arrived, farmers and tenants alike stopped other tasks to concentrate on fishing.

From about 1670 to 1890, more 'normal' conditions prevailed for Norway's herring fisheries, punctuated by occasional local or regional booms and collapses. Some places, especially on the west coast, specialized in herring and were the first to explore new technology. Others, like Råkvåg, were content with the old fishing routines and waited for the herring to come to the places that suited their traditional gear.

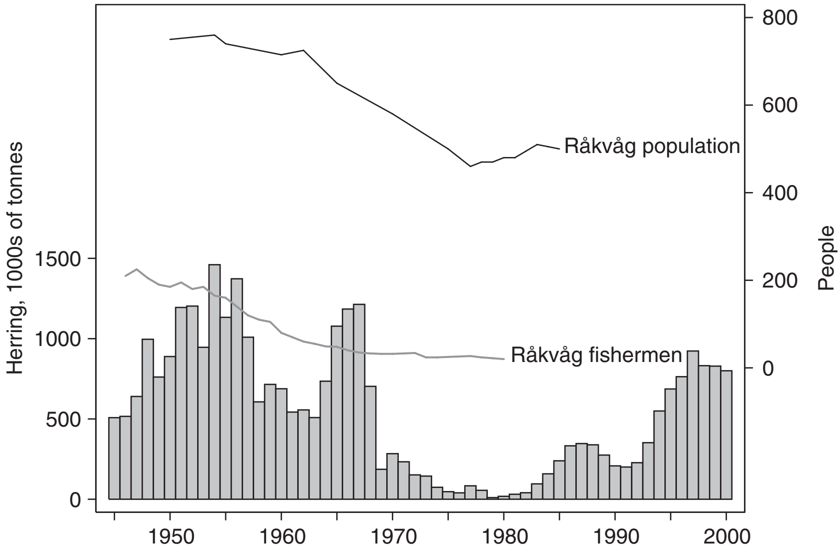
The second period of expansion in population and activities in the Råkvåg area came in the period 1890–1945, roughly the same growth period (and for much the same reason) as Siglufjörður, though not reaching the same size. Never before or since did Råkvåg become so important relative to the rest of the Fosen and Trøndelag area. This applied to both business and community life. From being an impoverished community with some tenant, fisher and farmer families, Råkvåg became the most affluent community on the Fosen peninsula. The population increased, both farmers and tenant families had a noticeable rise in living standards, and the community was enriched with more resident citizens and businessmen. It was a gradual development, with a wide spectrum of adjustments from the most dependent tenants to farmers gaining access to the ranks of the capitalists. In addition, businessmen came from other towns and established canning factories with their own export markets and boats. This created considerable competition for the local herring resources, and some conflicts arose between the new groups and the community's old élite. The long-standing landowners were still a dominant factor. They owned most of the land, and through these property rights they had full control of the local trading rights and most of the shared rights in the major shore-seine cooperative enterprises.

The best period for Råkvåg was between 1910 and 1920. Then, the concentration of herring came exactly to mid-Norway between Stadt in the south and Halten in the north, as it had done when the two fjords marked their position for the first time as rich herring sites. The herring were captured by the shore-seine cooperatives, usually 10–20 men. At its peak, the village of Råkvåg had 12 such cooperatives. Businessmen from all along the coast came with their vessels and bought the herring, to be salted onboard and transported in barrels to the markets. Because of its good harbour and rich community life, Råkvåg became the centre of this activity for the whole area. Some buyers established fish salting businesses at Råkvåg, increasing both activity and competition. The favourable market conditions during World War I motivated even more people to take part. In principle, shore-seine cooperatives have no chance in competition against the more mobile purse-seines, that can catch the herring long before it would reach the inner end of the fjords. However, the purse-seiners in the 1920s were mainly steam-powered vessels without hydraulic equipment, expensive and not very efficient. As markets became more problematic in the 1920s, many of these capitalist companies went bankrupt, while the shore-seine cooperatives still managed to survive. Independent of the increased competition at sea, which was moving to the disadvantage of Råkvåg, local fish-salting companies and a herring oil factory provided both women and men with work. In this way, the local fishing community managed to survive without dramatic change until the end of World War II.

The halt in most fishing during World War II allowed fish populations a chance to recover from fisheries depletion. The immediate post-war situation was probably as close to what might be considered ‘a natural state of fisheries’ as we saw during the whole 20th century. In this situation, the two fjords in Bjugn and Stjørna proved their reputation as the best herring fjords in mid-Norway. Råkvåg entered a new, more extensive, hectic and shorter period of growth. Herring were the basis for growth, and conditions in the fishery also caused the collapse. These conditions were not simply a matter of resources, however. Råkvåg was outmanoeuvred by more effective vessels and operating routines, as the fishery became more industrial, parallel to the second phase of Iceland’s herring era.

Already in the early 1950s Råkvåg had concentrated on fat herring, a seasonal fishery some time after Christmas, that provided the local salting businesses with their raw material. In the ordinary herring fishery (of spring-spawning herring) there was no more room for the shore-seine as the catches were taken far out at sea either by purse-seiners or by vessels with drift nets. Both these types of vessels used new hydraulic equipment, but purse-seiners became dominant when the herring almost disappeared after 1956. Purse-seiners’ new technology (especially the power block and nylon nets) allowed them to operate in deeper water and farther from shore, where the remaining fish were found. Again, Norwegian developments ran parallel to those in Iceland, half an ocean away.

Purse-seiners also tended to explore other resources as one stock declined. Consequently, they turned to fat herring in the late 1950s when spring-spawning herring became scarce. This shift by the seiners suddenly



Source: ICES (2001); Otterstad (1992).

Figure 4.6 Population and number of registered fishermen in Råkvåg, shown with the total Norway herring catch through the crisis years

eliminated the one niche left for the fleet and fish processors of Råkvåg. Figure 4.6 reflects the declines of both population and fishers in Råkvåg, alongside the herring collapse. As in Siglufjörður, Råkvåg's decline coincided with the expansion of the more capital-intensive offshore fishery, well before the terminal spike and collapse.

Despite the herring difficulties, the standard of living in Råkvåg was better than ever, and class differences were resolutely broken down. For the first time, however, the community suffered a merciless rationalization evaluation by the State. The question arose: what was the point of people living in places like Råkvåg? Norway found itself among the poorest countries in Europe in the late 1950s, which left little concern for maintaining a pattern of settlement based on vanishing resources. The situation was paradoxical. On one hand, fisher families were becoming relatively independent of the 'big men' and acquiring a material standard of which they had only dreamt. They had also put their own people into the system. On the other hand, they found that Råkvåg was being pushed out of active community life, and that they had to move to participate in the modern welfare state.

Up to the 1970s, it was possible to survive in Råkvåg by taking other occasional work in the service industries or commuting to industrial centres. In the latter part of the 1970s, politics became more favourable to the community, with the implementation of the so-called counter-cyclical policy. The economic prerequisite for this new policy was Norway's improved economy, based on oil drilling. The political prerequisite was the strong mobilization in rural communities in connection with the EEC referendum in 1972.

It is interesting to reflect that the National Insurance Scheme, which was introduced in 1967, would probably have moderated the economic collapse in Råkvåg if it had come in 1960. Because it came seven years after the worst crisis, it only rescued those who could no longer compete in the labour market (the old and the sick). Moreover, herring regulations in the 1970s virtually froze the situation as it was in fisheries at that time. Boat owners gained concession rights, and these rights were then considered more or less as an added value to the sale price of the vessels. It was also the case that the State in a way bought the right to reduce the catch capacity of the deep-sea fleet by compensating the owners with several million kroner when they transferred their vessels to another country. This guarantee was not available when the Råkvåg shipowners were being outmanoeuvred.

The situation in Råkvåg today depends on the State standing guarantee for continued settlement. In the last instance, both the municipality's activities and the main sources of residents' income (commuting and welfare) are dependent on State support. This leaves little basis for independent political initiatives. For the Fosen peninsula as a whole, much of the productive, marketable activity has disappeared. Rates of registered unemployment are low, but these do not measure what is actually produced.

In recent years, Råkvåg has seemed more conscious of this position of political dependence. Some of the younger leaders, who built their houses in the most promising period of the 1970s, view it as problematic that the authorities (municipality or State) can make political decisions that are disadvantageous to the local population. In connection with unpopular decisions, such as the destruction of a local salmon river to build a power station, or placing a waste deposit in the area, locals have come to realize that they have no political say. Business development such as tourism could strengthen their hand. Tourism has increased every year since local organizations and businesses began their summer arrangements in the mid-1980s. One can say that Råkvåg has awakened after years of slumber, but no longer as a fishing community. Rather, it appears as a museum for a vanished fishing community.

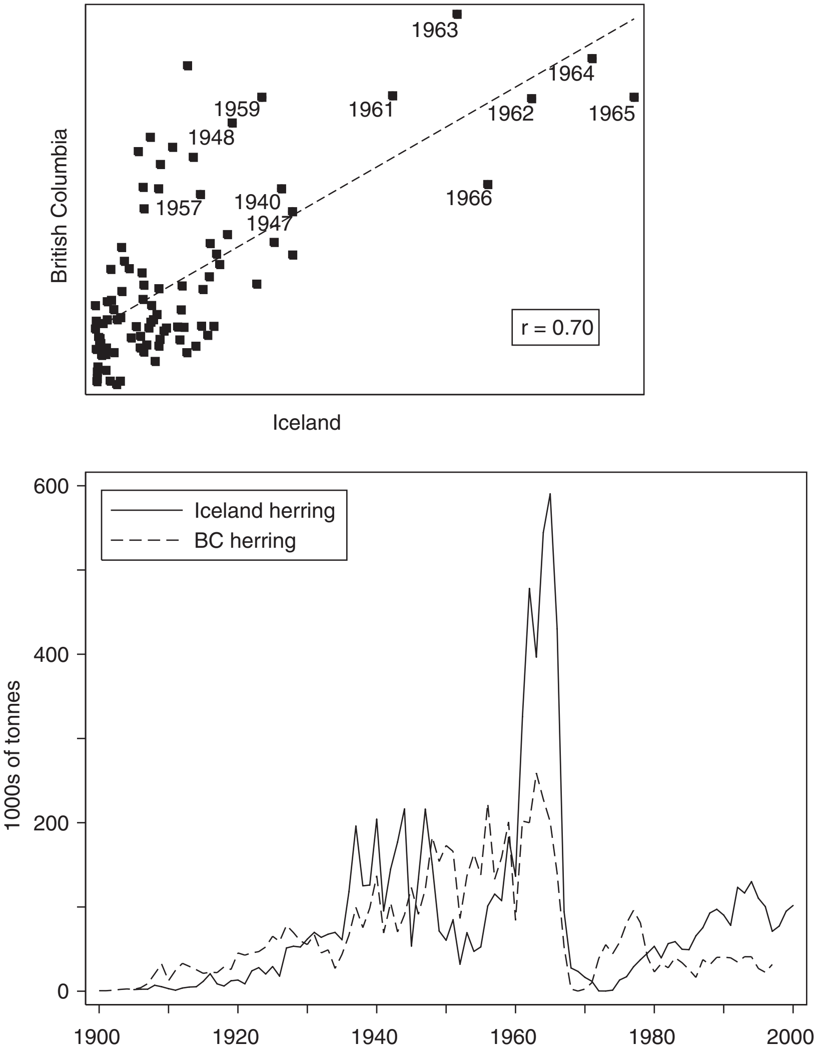
In Norway, there were many similar places with such Klondike experiences in herring. Many sank into a long-term crisis connected to the decline of the fishery in the 1950s and its total collapse in the late 1960s. A few communities, however, succeeded in maintaining their fishery through the years of virtually no herring. They managed by converting their activities to an alternative pelagic species, capelin, as in Iceland. Through this adaptation they stayed in business during the decades it took for herring stocks to rebuild. As a result, these few towns today are the herring élite, the only ones able to benefit from the reborn fishery. Råkvåg, on the other hand, experienced the collapse without state intervention, and hence lost its place in the fishery. In the decade from 1980 to 1990, the few remaining fishers in Råkvåg earned only about US$100 000 from fishing.

TELECONNECTIONS TO THE PACIFIC?

Correlations between small-pelagic fisheries catches have been noted across long distances, and even across oceans. The fisheries of Iceland and Norway are correlated because both involved the same migratory stock, Norwegian spring-spawning herring. Other correlations have less obvious causes, however.

Figure 4.7 shows 20th century catches of herring from waters off Iceland and British Columbia. Although oceans apart, the two series' correlation is 0.7, meaning that about half of their variance ($r^2 = 0.49$) is shared. Both series show nearly simultaneous take-offs in the late 1930s, and peaks followed by collapse in the 1960s. It might seem reasonable to guess that they are linked climatically, with changes communicated in some way through the atmosphere. This guess makes sense if we interpret fish catches as proxies for abundance, but Figure 4.2 demonstrated why such an interpretation is risky. Trends in Norwegian spring-spawning herring biomass and catches actually went in opposite directions during the 1950s and early 1960s. It was not abundance, but human factors, postwar technologies and markets, that drove catches to an unsustainable peak, quickly followed by collapse. Similar technologies and markets influenced the British Columbia herring fishery, and could have produced its similar, nearly simultaneous, collapse.

The scatterplot in Figure 4.7 reveals that the two fisheries' correlation derives mainly from the common boom from 1930 to 1970, and particularly from the high-catch outliers of the early 1960s. These high points represent artefacts of international markets and technology rather than signs of unprecedented abundance. Power-block assisted purse-seines of strong nylon mesh, guided by sonar to find schooling fish, were innovations after World War II that allowed rising catches at a time of falling stocks. In this example, it appears that human activities, instead of or in addition to



Note: Scatterplot indicates the linear relationship (correlation 0.70) between catches.

Source: ICES (2001); Marine Research Institute (2001); DFO (2004).

Figure 4.7 20th century trends for herring catches off Iceland and British Columbia climatic forces, caused the herring catches to follow parallel trends across different oceans.

In general, the hypothesis of 'human teleconnections' across spatially or ecologically distinct fisheries deserves serious consideration as we look for signals from climate. Technologies and markets have global reach, and act rapidly. Moreover, the ecological consequences of fishing down dominant species could well extend this reach to a wide range of non-targeted marine species, as well as to social systems on land.

DISCUSSION

Dependence on a vast shared resource built up the economies of herring towns around the north-central and northeastern Atlantic during the first half of the 20th century. During the post-war years, culminating in the early-1960s spike, the fishery rapidly drew down its resource. This killer spike disastrously coincided with an adverse climatic event, collapsing the fishery in the late 1960s. That pattern, of collapse resulting from climatic variation on top of overfishing, has characterized other fisheries crises (for example see Hamilton *et al.*, 2004b). The fisheries events reflect a more general proposition: climate changes tend to impact human affairs largely through interactions with resource use and distribution behaviour, which can reduce or exacerbate climate impacts. We should not expect to see simple, physical impacts from climate alone, unmitigated by social factors.

The importance of social factors becomes particularly clear if we compare neighbouring places (such as herring towns) that took divergent paths during a common ecological change. When inshore herring became less plentiful off mid-Norway or North Iceland, that ecological shift privileged port locations with better offshore access, and also those enterprises or individuals who invested in more technology- and capital-intensive fishing styles, which permitted them to range farther and deeper, pursuing whatever fish remained. This suggests a second general proposition: technological and capital intensification are common responses to resource depletion. In the short term, intensification succeeds, even while accelerating the depletion. Traditional, labour-intensive production styles cannot so easily extend their range, and hence suffer more immediately as resources thin.

A depleted species might rebound, or new species might take its ecological place. In the decades since the late-1960s herring collapse, industrial fisheries focused on capelin, while herring eventually began to recover. Resource recovery or substitutions did not return the old state of social affairs, however – a third general proposition from this study. The shift from labour-intensive to capital-intensive fisheries, more concentrated and in different locations, was not reversed. Places or enterprises that enjoyed ecological advantages, or made advantageous choices during transition times, later maintained their advantage through market or regulatory arrangements such as quota rights despite further ecological change (see Hamilton *et al.*, 2003, for a similar 'tale of two cities' from West Greenland). Social institutions can affect the longer term outcomes of change either by redistributing resources to the losers, or by reinforcing the new position of winners.

When their staple resource vanished, herring towns had to find other livelihoods, with varying degrees of success. Some combination of alternative fisheries, aquaculture and tourism constitute the standard alternative plan. Fishing towns often have historical and picturesque qualities that could well make them tourist drawcards, but development is limited by the supply of tourists, who are no more infinite than fish and can be scarce in remote coastal regions. Råkvåg's proximity to the urban area of Trondheim enabled it to become a regional attraction. Siglufjörður hopes for better connections to the relatively small urban centre of Akureyri, which in turn has natural and cultural attractions, transportation links and an established tourism industry that could send international visitors north to the old herring capital. Eastfjords towns such as Seyðisfjörður and Neskaupstaður are farther from tourist circuits and other attractions. At the time of writing, a cruise ship from Denmark and Norway regularly calls in Seyðisfjörður, providing income and jobs. Prospects for expanding East Iceland tourism substantially are uncertain, however. One possibility, the development of the dramatic highlands wilderness as an ecotourism destination, might have been set back by competing decisions regarding the lands.

Large-scale energy or mineral developments comprise yet another plan for some parts of the north, and they can evolve into a new main plan. The Kárahnjúkar hydroelectric megaproject in East Iceland, scheduled for completion by 2007, gives a striking example. This US$3 billion project, financed by Iceland's National Power Company, centres around a 190 m high rockfill dam under construction in the highlands northeast of Vatnajökull glacier. The dam will create a 57 km^2 reservoir above the canyon of the Jökulsá á Dal river, and divert the river's water through a 39 km tunnel to join the Jökulsá í Fljótsdal river to the east. A powerhouse by the Jökulsá í Fljótsdal will then generate the electricity.

Prime customer for the electricity will be a new aluminium smelter, under construction in the depressed former herring town of Reyðarfjörður (just south of Seyðisfjörður and Neskaupstaður, and in the same municipality as the latter). This smelter, itself costing more than US$1 billion, is financed by the US company Alcoa. It represents the largest private investment in

Iceland's history. The Icelandic government's political support and deep financial stake in Kárahnjúkar (as well as connected investments by national and local governments in infrastructure) have been justified by the estimated 1000 permanent jobs it might create, many at the smelter in Reyðarfjörður. The smelter and related development, it is argued, could stem the outmigration of young people that has been eroding the Eastfjords population since their fisheries declined. Through an indirect social path, fisheries troubles thus feed back to drive wholesale transformation of inland ecosystems and landscapes, as well as a former fishing fjord.

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5. An optimal harvest policy for the recently renewed United States Pacific sardine fishery

Samuel F. Herrick Jr, Kevin Hill and Christian Reiss

INTRODUCTION

Pacific sardine (*Sardinops sagax*) have at times been the most abundant of the coastal pelagic species (CPS) found in the California Current ecosystem. When the population is large it is abundant from the tip of Baja California (23°N) to southeastern Alaska (57°N). It is generally accepted that sardines off the west coast of North America consist of three subpopulations or stocks: (i) a northern subpopulation (northern Baja California to Alaska); (ii) a southern subpopulation (off Baja California); (iii) a Gulf of California subpopulation (Vrooman, 1964; Hedgecock *et al.*, 1989). A fourth, far northern subpopulation has also been postulated (Radovich, 1982).

Migratory behaviour also influences the distribution of sardine off the North American west coast. Pacific sardine probably migrate more extensively when abundance is high, moving north as far as British Columbia in summer and returning to southern California and northern Baja California in the autumn. When Pacific sardine abundance is low, there are no commercial quantities north of Point Conception. California tagging studies (Clark and Janssen, 1945) and data from the fisheries indicate that the larger fish move north. Migratory patterns appear to be complex, and the timing and extent of movement are apparently strongly influenced by oceanographic conditions (Hart, 1973) and stock biomass.

Pacific sardine supported the largest fishery in the eastern Pacific Ocean during the 1930s and 1940s. Sardine were taken along the coast of British Columbia, Washington, Oregon, California, and Mexico, with the bulk of the catch off California (Figure 5.1). Off southern and central California, a seemingly limitless resource and a huge demand for Pacific sardine products (canned sardine, sardine oil, meal and fertilizer) led to a boom fishery during the period. Landings along the entire coast peaked at >700 000

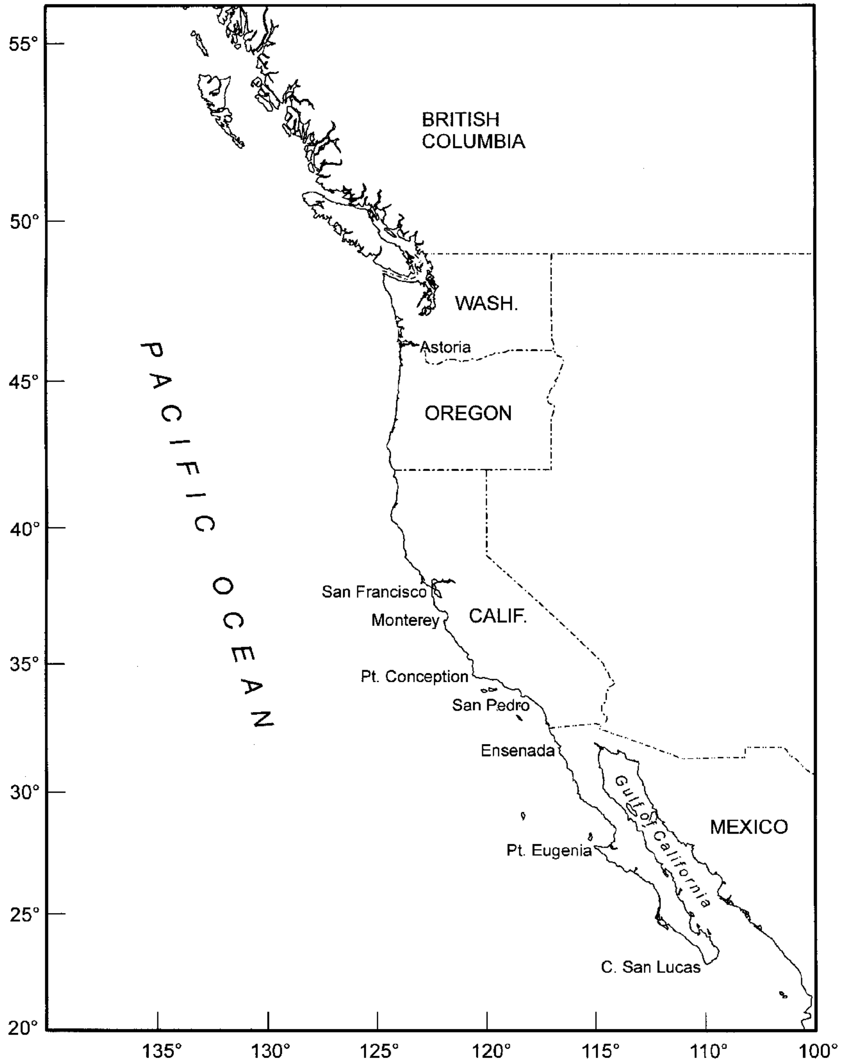

Figure 5.1 *Areas of Pacific sardine fishery activity along the North American Pacific Coast*

tonnes during the 1936–37 season, and continued at high levels throughout most of the 1940s, stimulated by the need for protein during and immediately after World War II. As Pacific sardine production soared, State biologists began to warn that the resource would be unable to sustain such a high level of landings. High catch rates persisted as the fishing fleet became much more proficient, and because a species like Pacific sardine tends to maintain a constant density at the centre of its geographic distribution while it contracts at its margins (MacCall, 1990).[1] Hence, catch rates do not decline as fast as abundance. Nonetheless, there was no mechanism in place to limit catches in accord with what the resource could sustain. Unsurprisingly, the fishery began a southward decline in the late 1940s when landings ceased in the northwest. Then, by the 1960s Pacific sardine landings were extremely low off California, which resulted in a moratorium on its directed fishery, starting in 1967.

Initially, the collapse of the Pacific sardine fishery was viewed as a classic case of overfishing: an overcapitalized industry, with too many fishing vessels employing advanced technology to harvest a fragile, if not a dwindling resource (Uber and MacCall, 1990). More recently, the collapse of the Pacific sardine fishery has been attributed to a combination of overfishing and an unfavourable environment. When the fishery peaked, ocean waters were beginning to cool, and the latter has been associated with lower biological productivity of sardine. This suggests that the collapse was partly due to a change in ocean temperature, linked to a change from warm to cold water off the Californian coast (Baumgartner *et al.*, 1992; Chavez *et al.*, 2003). This was not immediately evident, because the change is believed to be part of a natural, quasi-periodic event affecting the entire Pacific, in which sardine fisheries as far apart as California, Peru and Japan followed parallel courses of boom and bust.

Evidence of a vigorously recovering spawning biomass led California to lift its moratorium on Pacific sardine harvesting in 1986, and by the 1990s, extremely favourable biomass and environmental conditions were fuelling a rapid resurgence in Pacific sardine availability along the west coasts of the US, Mexico and Canada. In order to avoid the earlier experience off the US west coast, the US Pacific Fishery Management Council (PFMC) responded to the situation by instituting a fishery management plan (FMP) for coastal pelagic species (Pacific sardine, Pacific mackerel, jack mackerel, northern anchovy and market squid) in 1998 (Pacific Fishery Management Council, 1998).

In this chapter, we consider the biological and economic implications of the recent resurgence of Pacific sardine in the US west coast CPS fishery. We focus on the FMP's domestic harvest policy for CPS, particularly as it pertains to Pacific sardine. We review conditions surrounding the collapse of the historical sardine fishery and the renewed US, Mexican and Canadian fisheries for Pacific sardine, and examine the development of the FMP's harvest policy for Pacific sardine and how it addresses the factors attributable to the failure of the earlier fishery. Finally, we evaluate the policy implications of the harvest policy and offer some conclusions.

THE HISTORICAL PACIFIC SARDINE FISHERY, ITS COLLAPSE, AND EPILOGUE

The US Pacific sardine fishery developed in response to an increased demand for protein during World War I. The fishery expanded rapidly, and became so large that by the 1930s sardine accounted for almost 25 per cent of all fish landed in the US (Frey, 1971). Coastwide landings exceeded 350 000 tonnes each season from 1933 through 1934 to 1945 through 1946; 83–99 per cent of these landings were made in California, the balance in British Columbia, Washington and Oregon (Figure 5.2). In the late 1940s, sardine abundance and landings declined dramatically (MacCall, 1979; Radovich, 1981). The decline has been attributed to a combination of overfishing, environmental and policy factors, although the relative importance of these factors is still open to debate (Clark and Marr, 1955; Jacobson and MacCall, 1995).

Overfishing has strong economic underpinnings. The advent of sardine canning around the beginning of the 20th century created two new industries. Canning provided high quality, highly valued canned Pacific sardine for human consumption. The second new industry produced protein-rich animal feeds, fertilizer and oil from the reduction of the canning wastes. So valuable were these canned sardine by-products that processors were soon using whole fish and canning waste to produce fishmeal, oil and fertilizer. Concern over using whole fish instead of waste led to laws that allowed only those plants that canned sardine for human consumption to produce sardine by-products. Consequently, canners supplied canned sardine at or below cost to obtain enough waste and whole fish for reduction (Uber and MacCall, 1990).[2] World War II stimulated demand for Pacific sardine products, and annual landings averaged more than 500 000 tonnes from 1939 through 1945, more than twice the sustainable harvest quota called for by California biologists (Radovich, 1981; Uber and MacCall, 1990). The increase in landings was mainly due to an increase in the number of larger, highly mobile, more technologically advanced vessels, fishing farther offshore, that is more efficient units of fishing effort. By then, the economics of reduction and the institutional arrangements under which sardine were harvested (that is open access – common property) had led to extreme overcapitalization of the fishery as well as both economic and biological overfishing of the resource.

The declining years of the sardine fishery were also a period of cooling ocean water associated with lower biological productivity of sardine, and the contraction of a declining biomass southwards (McFarlane *et al.*, 2002). As a result, harvesting ceased completely in British Columbia, Washington and Oregon in the late 1940s, but significant quantities continued to be landed in California through the 1950s (Figure 5.2).

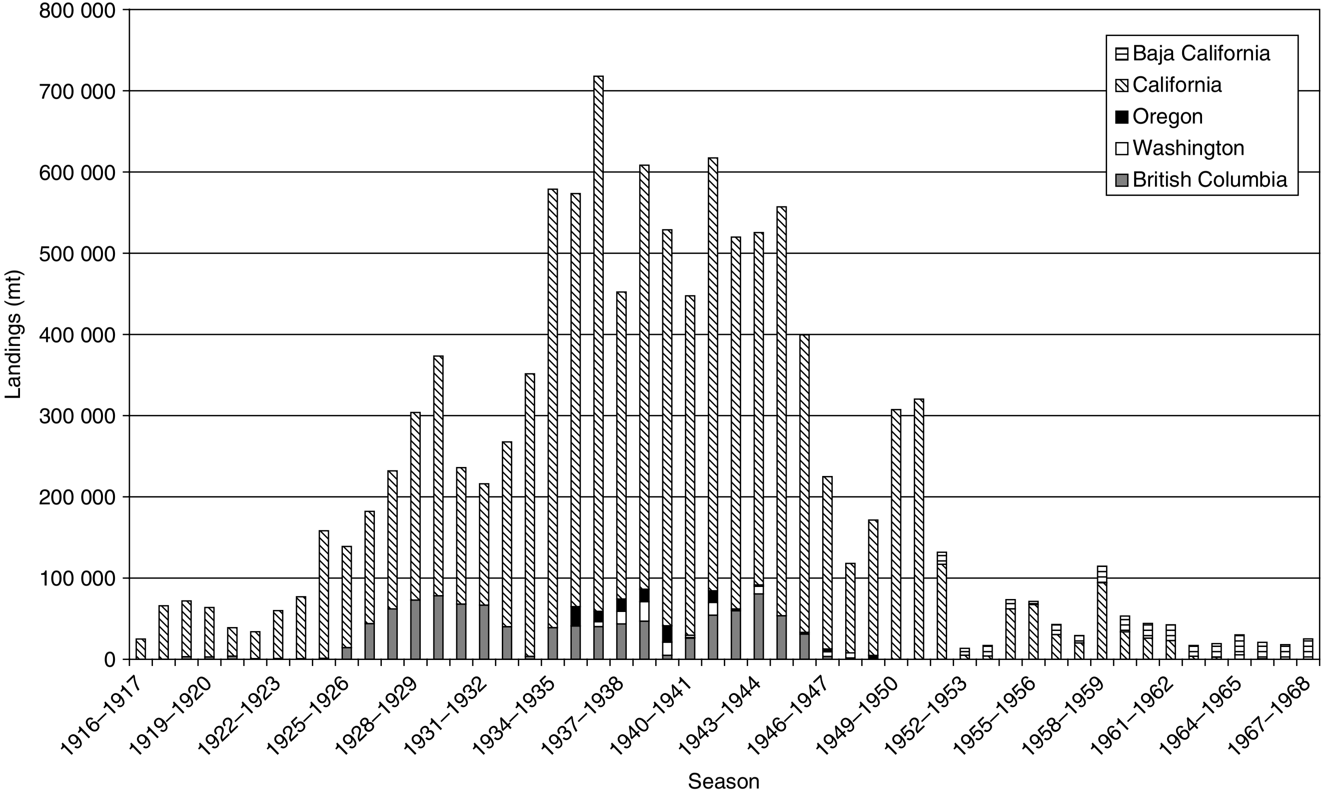

Figure 5.2 *North American Pacific Coast landings of Pacific sardine, 1916–1917 through 1967–1968 seasons*

These circumstances have been thought to give rise to a situation in which a gradual depletion of a resource is obscured by its behavioural response to the lowering of ocean temperature. This line of reasoning draws from the work of MacCall (1976, 1990) and Radovich (1973, 1976, 1981), in which it is argued that, as the reduced biomass contracts into a smaller area, it becomes more available there, and the fishery may not experience noticeable changes in catch per unit effort.[3] In the case of sardine, the idea that the resource was in some sense infinitely abundant follows from fishing on the centre of biomass concentration by increasingly efficient units of fishing effort. In other words, as it was shrinking, the biomass became denser and therefore more available to a fleet of highly efficient fishing vessels acting cooperatively to converge on a concentration.[4] Given this situation, one would expect to see a rather abrupt drop in Pacific sardine landings to indicate that the resource had become overfished (Figures 5.2, 5.3). This indeed appeared to be the case starting in the 1950s, as the reduced biomass became centred off southern California.

Radovich (1981) suggested that differences in agency perspectives played a role in the fishery's collapse. He pointed out that the then California Division of Fish and Game issued warnings that commercial exploitation of the resource could not increase without limit and advocated an annual harvest quota to prevent overfishing. This was consistent with its charge to provide for the restoration and preservation of fish in State waters. On the

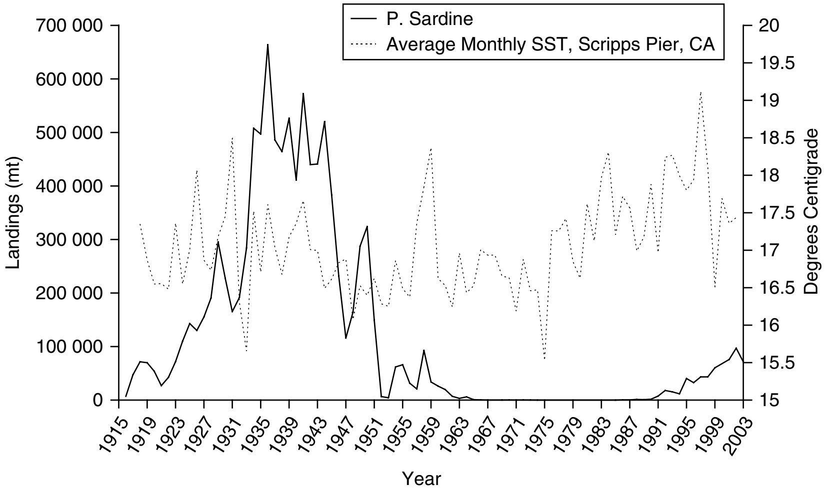

Figure 5.3 *US west coast landings of Pacific sardine, and Scripps Pier, California sea surface temperatures, 1916–2003*

other hand, the then Federal, US Bureau of Commercial Fisheries (BCF) was dedicated to developing and maintaining viable US fisheries, which endeared it to industry, but tended to put it at odds with the State. The BCF was inclined to look towards nature and not industry as the main cause of the Pacific sardine's decline. This divide in conservation-management philosophy led to a management stalemate, with no action taken to prevent overfishing.

What was the aftermath of the Pacific sardine fishery's failure? Before the collapse of the fishery, a typical operational pattern for purse-seiners entailed fishing for salmon off Alaska during June through September, sardine off the west coast from October to March, and market squid during April, May and often September, off southern and central California. Therefore, those who participated in the fishery during its prime had some options after it collapsed (Uber and MacCall, 1990). Highly specialized purse-seiners, those that were not able to switch to alternative gears, could fish for other CPS throughout the year, primarily northern anchovy, of which abundance increased after the Pacific sardine collapse, but also Pacific mackerel and market squid. They would also fish for tuna when available. Smaller, less specialized, vessels could also engage in the Dungeness crab pot fishery from November through February, and at other times of the year, the albacore troll or salmon troll fisheries. Some, particularly from the Pacific northwest, ventured into the emerging west coast groundfish fishery. Shifts to other fisheries were often accompanied by crew reductions.

Many vessels for which these were not viable options were sold, often at prices well below their original cost. Some larger vessels were purchased for use in the expanding king crab fishery off Alaska. Others were sold in the international market for use in small pelagic fisheries in countries such as Peru, Chile and South Africa. Chile and Peru also purchased much of the surplus sardine canning and reduction equipment. The relocation of cheap vessels and processing equipment to these countries was also accompanied by a transfer of corresponding technical, scientific and management expertise (Uber and MacCall, 1990). Consequently, much of the human capital that became surplus with the collapse of the Pacific sardine fishery became involved in the development of the sardine and anchoveta fisheries off South America. The transfer of existing capital, labour, technology and expertise greatly shortened the learning curve, and allowed those fisheries to develop much more rapidly and cost-effectively than if they were totally dependent on the host country for the necessary resources. In retrospect, this outcome might be perceived as an optimum economic strategy on the part of commercial US sardine interests, effectively mining the fish in a seller's market as quickly as possible, then shifting equipment and operations to South America to help develop the Peruvian anchoveta fishery, whose exploitation followed the California example, including its collapse in the early 1970s (McEvoy, 1986; MacGarvin, 2002).

The opportunity for vessels, equipment and personnel to find employment opportunities in other domestic ventures and overseas greatly helped to lessen the economic damage from the collapse of the US Pacific sardine fishery. However, this did not obviate the need to address the 'tragedy of the commons' that the collapse of the Pacific sardine fishery represented. More broadly, one of the most important lessons learned from this experience was that the development of industrial fisheries usually proceeds much more rapidly than the difficult political process of developing and instituting sound conservation and management policies designed to prevent a 'tragedy of the commons'. In effect, the difficulty in preventing, or remedying, such a situation serves as a perverse subsidy to those having a commercial interest in fishery resources. On the other hand, if cycles in Pacific sardine abundance are strictly environment-driven, then a strategy of mining the resource on a first-come first-served basis may be entirely reasonable for the US.

This lesson was not lost on state and federal fishery agencies. As signs of a Pacific sardine renaissance began to appear, California and then the US government took steps to prevent reoccurrence of a Pacific sardine 'tragedy of the commons'.

OPTIMUM HARVEST POLICY

Renewal of the Pacific Sardine Fishery

In the late 1970s and early 1980s, the California Department of Fish and Game (CDFG) began receiving anecdotal reports about the sighting, setting, and dumping of 'pure' schools of juvenile sardine, and the incidental occurrence of sardine in other fisheries, suggesting increased abundance (California Department of Fish and Game, 1986). In 1986, the State lifted its 18-year moratorium on Pacific sardine harvest on the basis of at-sea-survey and other data indicating that the spawning biomass had exceeded 18 144 tonnes (20 000 short tons), the State's lower limit for a directed fishery. In accordance with its Pacific sardine recovery plan, California established annual directed quotas[5] for Pacific sardine, beginning in 1986.

As the Pacific sardine biomass increased through the 1990s, fishing opportunities in northern areas increased, and Pacific sardine began to appear off Oregon, Washington, British Columbia (Figure 5.1), and in international waters more than 200 miles offshore of southern California. There were few regulations pertaining to Pacific sardine harvests in Oregon and Washington, and catches off Oregon, Washington and outside state waters could not be managed effectively under existing Californian regulations. Mexican harvests were increasing, raising concerns that they might be sufficient to eliminate the Pacific sardine recovery even in the absence of a US fishery. Given these developments, consolidation of responsibilities for Pacific sardine and all CPS under a single federal, fishery management plan would make management of the directed Pacific sardine fishery more efficient and effective.

In addition, cooperative, transboundary conservation and management of CPS resources shared by the US, Mexico and Canada was recognized as paramount. Yet, while there were informal collaborations by researchers from the US, Mexico and Canada on key scientific issues, no formal, cooperative arrangement for international management of sardine and other CPS resources was in place. Management of all CPS under a single federal FMP would greatly facilitate cooperative international and interstate management and scientific work. However, experience with the anchovy fishery indicated that management agreements and programmes for collaborative scientific work in the absence of an FMP and direct federal involvement would be difficult to achieve. In the meantime, each country continued unilaterally to manage its own CPS fisheries.

In June 1997 the PFMC approved Amendment 8 to its northern anchovy FMP, which expanded its scope to include the entire CPS fishery. The Council's decision was strongly endorsed by CDFG and the US National Marine Fisheries Service (NMFS). This action was based on supporting documentation provided by the CDFG that described increases in the abundance, distribution and catch of Pacific sardine, as well as market squid, while at the same time there were insufficient resources available at State level for management. A CPS FMP development team was formed and directed to begin work on Amendment 8, the CPS FMP, which was implemented at the beginning of 2000 (Pacific Fishery Management Council, 1998).

From a management perspective, the most significant feature that CPS fisheries share is their propensity for change, in both resource availability and market demand, in both the long and the short term.[6] Given these circumstances, it becomes extremely difficult for any management regime to produce stable yields of Pacific sardine over time. Natural events may lead to short and prolonged periods of biomass decline, and given the vagaries of the domestic Pacific sardine fishery, and in the markets for Pacific sardine products, managers of CPS fisheries should expect, and plan for, considerable variation in abundance and yields. Foremost in this regard is the need to develop a harvest policy for each CPS that (i) prevents biological overfishing by curtailing fishing during periods of low abundance to protect the long-term health of the stock; and (ii) that addresses overcapitalization and economic overfishing by managing harvesting activities to ensure that the allowable harvest from the stock provides the greatest range of social and economic benefits to the public at large. Below we describe the Council's approach towards addressing these issues in the CPS FMP with respect to Pacific sardine: biological overfishing through its environment-based maximum sustainable yield (MSY) harvest control rule for Pacific sardine, and overcapitalization through its limited entry programme for the CPS finfish fishery.

MSY Control Rule: Optimum Yield from the Resource

The MSY control rule for Pacific sardine defined in the CPS FMP (Pacific Fishery Management Council, 1998) is environment-based and tuned to the importance of sardine within the ecosystem. It is intended to prevent sardine from being overfished and to maintain relatively high and consistent catch levels over the long term. To achieve this, the control rule has been formulated to reduce the exploitation rate as the Pacific sardine biomass declines, and as the oceanic environment becomes less favourable for an expanded Pacific sardine biomass. The harvest formula (Hill *et al.*, 1999) is:

$$H_{t+1} = (BIOMASS_t - CUTOFF) \times FRACTION \times US\ DISTRIBUTION$$
$$s.t.\ H_{t+1} <= MAXCAT$$

where H is the US harvest target level,[7] $BIOMASS$ is the estimated biomass of fish at the beginning of the season, $CUTOFF$ is the lowest level of estimated biomass at which directed harvest is allowed, $FRACTION$ is the fraction of the biomass above $CUTOFF$ that can be taken by the fishery, and $US\ DISTRIBUTION$ is the percentage of harvestable biomass in US waters.

The estimated $BIOMASS$ is from an age-structured population simulation model that uses fishery-dependent and -independent data to estimate annual Pacific sardine abundance, year-class strength and age-specific fishing mortality (Hill *et al.*, 1999). $CUTOFF$ protects the stock when biomass is low; explicitly accounting for the importance of sardine as forage for other species of fish, marine mammals and seabirds. $FRACTION$ explicitly accounts for environmental impacts on the stock biomass.

Compared with the other CPS, the biological productivity of Pacific sardine is most strongly affected by environmental variation. Favourable and unfavourable periods or 'regimes' for Pacific sardine tend to occur in phases of about 50 years (Baumgartner *et al.*, 1992). Therefore, periods of low abundance for Pacific sardine will probably take place even in the absence of a fishery. This makes choice of the control rule parameters *CUTOFF* and *FRACTION* more complex in the case of Pacific sardine.

In addition to the *CUTOFF*, *FRACTION* and *US DISTRIBUTION* parameters, the MSY control rule also incorporates a maximum harvest level constraint *MAXCAT* ($H <= MAXCAT$). *MAXCAT* is used to guard against extremely high catch levels attributable to errors in estimating biomass, to reduce year-to-year variation in catches, and to avoid overcapitalization during short periods of high biomass and high harvest. *MAXCAT* also prevents the catch from exceeding MSY at high stock levels, and spreads the catch from strong year classes over a wider range of fishing seasons.

In the MSY control rule for Pacific sardine, *FRACTION* is a proxy for F_{MSY} (that is the fishing mortality rate for deterministic equilibrium MSY). The value for *FRACTION* is based on recent ocean temperatures, taking into account the fact that Pacific sardine biological productivity is higher under oceanic conditions associated with relatively warm water temperature. The relationship between F_{MSY} for Pacific sardine and ocean temperature is:

$$F_{MSY} = 0.248649805\ T^2 - 8.190043975\ T + 67.4558326$$

where T is the average sea surface temperature at Scripps Pier (La Jolla, California) during the preceding three years. The MSY control rule for Pacific sardine sets the control rule parameter *FRACTION* equal to F_{MSY}, except that *FRACTION* is never allowed to exceed 15 per cent or to be lower than 5 per cent (Figure 5.4). The upper and lower levels for *FRACTION* are policy decisions based on social, economic, ecological, environmental and biological criteria. In this context H then becomes the 'optimum yield'[8] from the fishery.

The current MSY control rule for Pacific sardine (Conser *et al.*, 2003) sets H for the northern Pacific sardine stock based on an estimate of biomass for the entire stock, with: *CUTOFF* equal to 150 000 tonnes, *FRACTION* at 15 per cent, and *MAXCAT* equal to 200 000 tonnes. The *US DISTRIBUTION* is 87 per cent, the estimated proportion of *BIOMASS* in US waters based on analyses of data from aerial spotters.

Limited Entry – Efficient Utilization of Target Yield

Based on the circumstances leading up to the collapse of the historic Pacific sardine fishery, the Council was concerned that with the resurgence of Pacific sardine, the CPS finfish fishery (Pacific mackerel, Pacific sardine, jack mackerel and northern anchovy) would become overcapitalized faster than management authorities could react, given the rate of recovery of

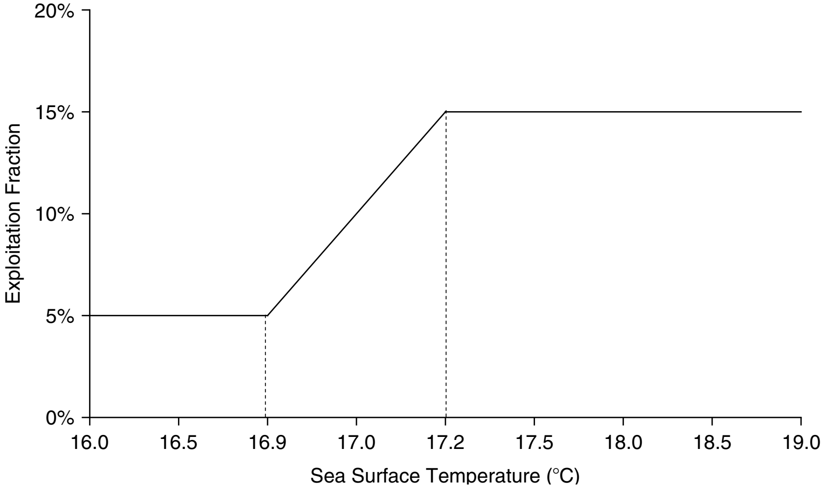

Figure 5.4 Environmentally based US Pacific sardine harvest control rule fraction (for any given year, sea surface temperature is the running average sea surface temperature at Scripps Pier, La Jolla, California for the proceeding three years). Thus the years are three preceding a given year

sardine and ensuing market opportunities. To prevent this from happening, the Council instituted a limited entry programme for the finfish fishery south of 39°N.[9]

Initially the Council considered several CPS finfish limited entry fleet-size options based on the number of vessels accounting for different proportions of the total CPS finfish landings south of 39°N (Pacific Fishery Management Council, 1998). The Council's preferred option was for a limited entry fleet consisting of the 70 vessels that accounted for 99 per cent of the total CPS finfish landings during the five-year qualifying period for limited entry, 1993–1997. While the Council recognized that an economically efficient finfish fleet would probably be smaller in number (40 vessels accounted for 95 per cent of finfish landings during the qualifying period), the 70-vessel fleet was considered less disruptive in terms of impacts on fishing communities. Moreover, any effort to achieve a smaller, highly specialized, more efficient CPS limited entry finfish fleet could have significant repercussions in terms of reducing harvesting capacity in alternative fisheries in which conventional CPS finfish vessels participated, namely those for market squid and tunas. Although 70 vessels were expected to qualify, only 65 vessels actually acquired permits for the limited entry fishery, either directly or through transfer.

Subsequently, the Council decided to establish a harvesting capacity goal for the limited entry finfish fishery, where fleet size would be based on the number of vessels capable of harvesting the long-term expected annual combined harvest guideline for CPS finfish. This initiated an output-orientated data-envelopment analysis (DEA)[10] which was conducted to estimate harvesting capacity for each of the 65 vessels constituting the CPS finfish limited entry fleet (Pacific Fishery Management Council, 2002). Two measures of finfish harvesting capacity per trip were derived for each limited entry vessel. The first measure was based on the maximum recorded landing for each vessel over the period, and approximates a vessel's physical capacity, its maximum output per trip. Physical capacity is typically related to a vessel's hold capacity, and reflects the fisher's opinion regarding one or more extraordinary occurrences in the fishery: (i) periods of peak availability of fish; (ii) unique environmental conditions that enhance effort production; or (iii) extreme demand for output. The second measure of harvesting capacity was based on each vessel's average finfish landing over the period, and approximates output, landings per trip, under what are considered usual or normal operating conditions. This concept of capacity incorporates the fisher's expectations concerning typical variations in resource availability, environmental conditions and output demand, and in this sense is a technological economic measure of harvesting capacity (Walden *et al.*, 2003). Each vessel's calculated gross tonnage[11] was used as a proxy for the vessel's capital stock, the fixed input, in the CPS finfish DEA.

Annual harvest capacity for each vessel was its per-trip capacity multiplied by the amount of effort (trips) each vessel was expected to generate during the year. As with physical and normal measures of harvest capacity per trip, the amount of effort a vessel produces during the year can be considered in terms of what is possible from a purely technological or engineering standpoint, and that which reflects normal resource availability, environmental conditions and market conditions. The former can be thought of as physical effort, the latter as normal effort. Each vessel's physical effort was the maximum number of finfish trips per year observed over the period 1981–2000. Each vessel's normal effort was the average number of per year trips over the period. Accordingly, each vessel's annual physical harvesting capacity was calculated as its physical capacity per trip multiplied by its maximum number of trips per year (physical effort), and each vessel's annual normal harvesting capacity was calculated as its normal capacity per trip multiplied by its average number of trips per year (normal effort). Summing annual vessel capacities provides an estimate of annual harvesting capacity for the finfish limited entry fleet. The CPS finfish limited entry fleet's physical annual capacity was estimated as 538 824 tonnes, its normal capacity as 111 417 tonnes.

To determine the long-term expected aggregate CPS finfish target harvest level, a time-series of finfish biomass estimates for each finfish species was assembled for the period 1932–2000. The MSY and harvest target level control rules from the CPS FMP were applied to each species' annual biomass estimates for each year in the period to obtain annual target harvest levels in current time equivalents (Figure 5.5). The long-term expected, or average, aggregate finfish harvest level was 108 306 tonnes. The peak aggregate CPS finfish target harvest based on the same series was 273 507 tonnes.

The results of the DEA indicated that the existing 65 vessel CPS finfish limited entry fleet would achieve the harvest capacity goal; normal harvesting capacity approximating the expected long-term annual aggregate finfish harvest target level. The limited entry fleet's physical harvesting capacity would be more than sufficient to harvest the annual maximum potential, peak period, amount of finfish. This 'excess finfish capacity' may be much more important on a per-trip rather than an annual basis in terms of satisfying peak period conditions.[12] Moreover, this 'excess finfish capacity' may well be directed towards the harvest of market squid and tuna, or other non-CPS finfish opportunities. Therefore, while there may be some unrealized efficiency gains in the CPS finfish fishery, given the range of opportunities available to CPS finfish fleet, and other non-efficiency considerations, maintaining the harvesting capacity of the current limited entry fleet of vessels appears consistent with the achievement of an optimum harvest policy for CPS.

Domestic Allocation of the Acceptable Biological Catch – Full Utilization of the Target Harvest Level

Traditionally, the US commercial fishery for Pacific sardine has been in the waters off California and comprises a southern subarea fishery that includes the fleet based primarily in the San Pedro-Los Angeles area, and a northern subarea fishery primarily based in Monterey Bay (Figure 5.1). However, with improving environmental conditions and an expanding biomass, sardine have recently become available in commercially fishable quantities in the Columbia River plume off Oregon and Washington,[13] and the Pacific Northwest fishery has re-emerged (Figure 5.6).[14] Each regional fishery targets Pacific sardine, and in the California fisheries, mackerel, market squid and tunas when available. Ex-vessel landings in all sectors are driven by domestic and international market forces for sardine, as well as the availability and markets for other species of economic benefit to Pacific sardine vessels and processors. The US harvests and processes Pacific sardine for non-reduction use, which includes: (i) direct human

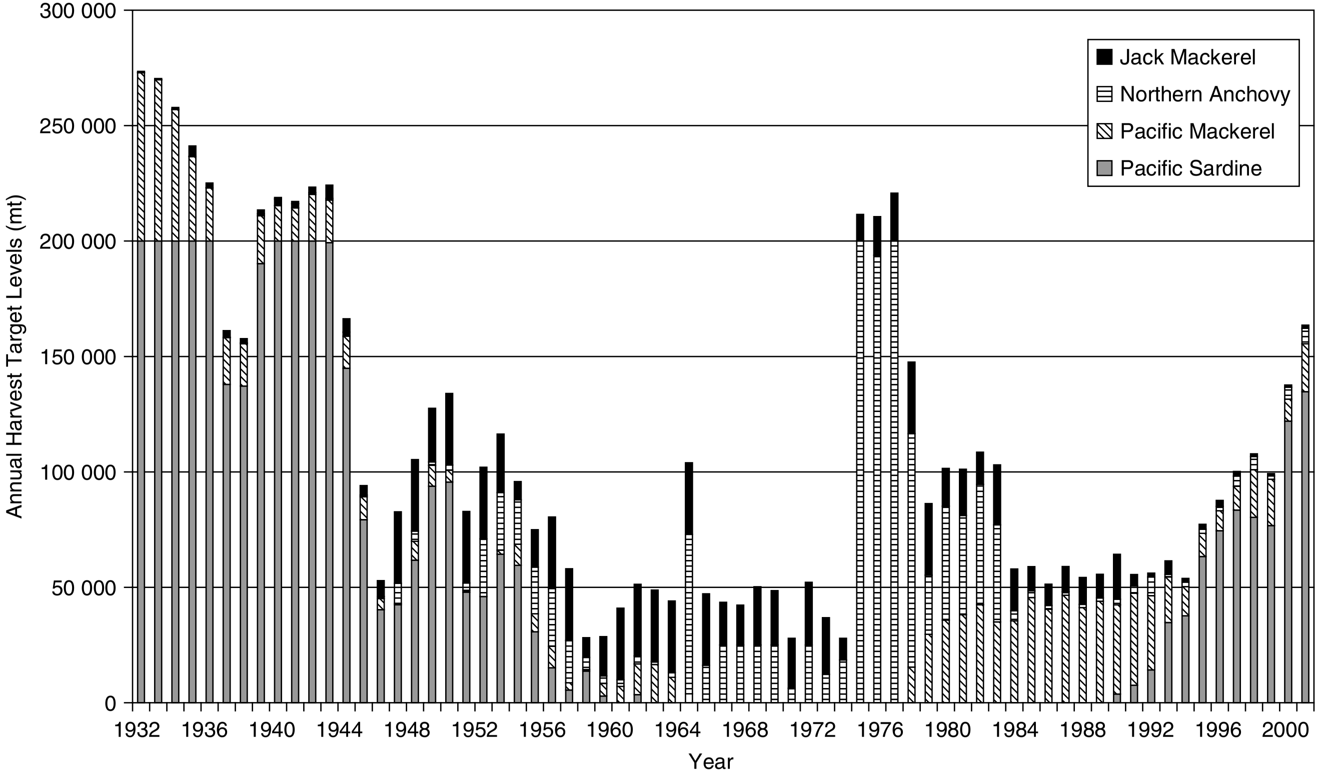

Figure 5.5 Estimated US West Coast CPS finfish annual harvest target levels, 1932–2000

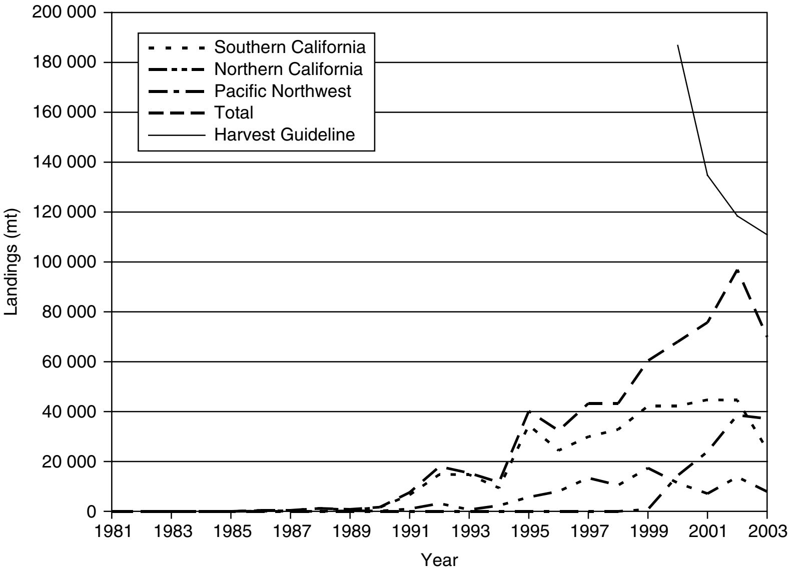

Figure 5.6 US West Coast Pacific sardine landings and harvest guideline (mt), 1981–2003

consumption; (ii) aquaculture feed (whole fish); (iii) bait for commercial fisheries; and (iv) sport-fishery bait.

While the Pacific sardine harvest control rule in the CPS FMP establishes a coastwide acceptable biological catch (ABC) within the US west coast exclusive economic zone (EEZ), each regional fishery tends to harvest sardine according to a schedule dictated by their availability, by the availability and markets for other species of economic importance, and by weather and ocean conditions. Generally, the southern Californian fishery starts harvesting Pacific sardine from 1 January, and harvests increase steadily throughout the year. The northern Californian fishery starts in August (tied to market squid availability) and increases through January or February of the following year. The Pacific Northwest has a much more abbreviated season, which starts in June and generally concludes in October, when weather and ocean conditions make fishing difficult or impossible for purse-seiners, and less productive because Pacific sardine schools are harder to locate (Figure 5.7).

Because of the seasonal nature of the northern and southern Californian fisheries, California's existing north-south allocation framework for the coastwide Pacific sardine ABC was retained in the FMP to ensure that the northern Californian sector received a reasonable fishing opportunity.

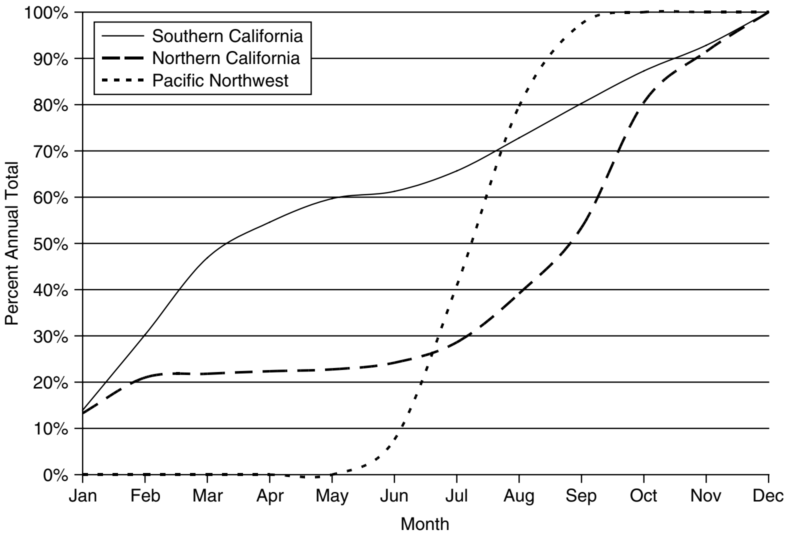

Figure 5.7 Cumulative monthly sardine landings by subarea, 1999–2002 average

Without an allocation framework in place it was conceivable that the southern Californian fishery could harvest the entire ABC before the northern Californian fishery began its season. At the time of the development of the FMP there was no Pacific Northwest fishery to consider in terms of an allocation framework. The existing allocation framework partitioned the ABC 66 per cent to the southern subarea and 33 per cent to the northern subarea on 1 January. On 1 October, the remaining ABC would be pooled and reallocated 50:50 to each subarea. The subarea line was at 35°40′N.

With expansion of the Pacific sardine fishery into the Pacific Northwest, the northern subarea allocation came to be shared by the northern Californian and Pacific Northwest fisheries. This raised concerns that the existing allocation framework would not provide an optimal harvest opportunity to each regional fishery. Under the adopted allocation framework (and given *status quo* harvest levels), there was reasonable fear that the northern area fisheries would attain their portion of the annual ABC prior to the scheduled 1 October reallocation, which would effectively cause premature closure of the Pacific Northwest fishery. Indeed, in 2002, the northern area allocation was reached, and the fishery closed in mid-September. Owing to concern over the economic and community impacts resulting from this closure, emergency action was taken to reallocate the remaining ABC before the designated date of 1 October. The express purpose of this emergency reallocation was to avoid unnecessary economic hardship. At the time, a sufficient amount of the Pacific sardine ABC remained to satisfy all users. Moreover, by the end of the year, the ABC had not been attained (more than 17 000 tonnes remained unharvested). Had the reallocation taken place earlier, avoiding the mid-September closure in the north, there would probably have been fuller utilization of the ABC, resulting in an increase in net benefits from the fishery. This was particularly noteworthy in that a principal goal of the CPS FMP is to ensure full utilization of the ABC. In recent years, however, as much as 59 000 tonnes of the ABC has remained unharvested at the end of the season. This situation has raised concerns that foregone harvest opportunities could be worsened by the current allocation formula, and could result in negative impacts for the coastwide fishery and a substantial loss of net benefits. The potential welfare gains would result not only from fuller utilization of the ABC, but also from the greater profitability of sardine harvested and processed in the Pacific Northwest fishery relative to the fisheries off California.

Based on its relative scale, and the particular market niche it supplies, the Pacific Northwest fishery can be characterized as a relatively low volume, but high value, enterprise. Because of their availability at the northern extent of the distribution, the Pacific Northwest fishery has been harvesting relatively large sardine,[15] which are highly sought for use as longline bait, particularly in Japan, where only the highest quality sardine are used as bait in their longline fisheries. Moreover, owing to the shortage of their own 'local' sardine resource, Japan has been especially interested in foreign sources of sardine for a number of uses. As the longline bait market matured, processors in the Pacific Northwest began turning their attention to market opportunities in Japan and elsewhere for sardine for human consumption. The amount destined for human consumption is expected to grow as additional food markets are developed and the longline bait market becomes saturated.

Market potential was not lost on the California fisheries, but owing to size and quality considerations, theirs has been more of a high-volume, low-value operation than that in the Pacific Northwest. The sardine landed in California are generally smaller than those landed in the Pacific Northwest.[16] Most of the recent landings in northern California have been exported to Japan for either human consumption or for small longline bait, with the balance going to Australia, either to tuna farms for feed or for sportfishing bait. Pacific sardine landings in southern California are generally smaller sardine than the landings in northern California. Lately, the bulk of southern California landings has been exported to Australian tuna farms, although this market appears to be weakening owing to a revival of the Australian sardine fishery. On the other hand, exports for human consumption and bait have been increasing, boosted by increased demand in Japan for all sizes of sardine, because of reduced production from the Japanese domestic fishery.

With the northern subarea allocation now shared by the northern California and the Pacific Northwest fisheries, another concern was that harvest opportunity for the northern Californian fishery could be preempted by the Pacific Northwest fishery (Figure 5.7). Although the development of a significant fishery in the Pacific Northwest has changed the harvesting dynamics coastwide, it would appear that, at the most recent ABC levels and utilization patterns, additional fishing opportunities could be provided to the northern Californian and Pacific Northwest fisheries without adverse impacts on the southern Californian fishery under a revised allocation framework.

To begin developing a durable allocation framework that would provide optimal harvesting opportunities when a Pacific Northwest fishery was present, and to avoid a repeat of the emergency situation that arose in the Pacific Northwest during 2002, the Council adopted a new, interim[17] allocation framework in April 2003. The interim framework promoted more efficient and fuller utilization of the annual Pacific sardine ABC, while accounting for the socioeconomic impact on any sector of the west coast Pacific sardine fishing industry, and on fishing communities. Prominent features of the interim allocation system were: (i) movement of the geographic boundary between the northern and southern subareas from 35°40′N (Point Piedras Blancas, California) to 39°N (Point Arena, California), which combined the Californian fisheries into the southern subarea; and (ii) movement of the reallocation date for the remaining Pacific sardine ABC from 1 October to 1 September.

It was generally agreed that the impacts of the interim allocation scheme used to partition the Pacific sardine harvest guideline would primarily be socioeconomic. However, the development of a long-term allocation framework would require that the biologically based implications of different allocation schemes be further evaluated to provide management guidance regarding how the operations of the regional fisheries might affect the dynamics of the Pacific sardine population as a whole. Therefore, in an effort to achieve an optimal domestic harvesting policy, a more comprehensive analysis of alternative allocation frameworks in terms of long-term socioeconomic, biological and ecological impacts is now underway.

POLICY CONSIDERATIONS

The US's harvest policy for Pacific sardine includes a well founded risk-averse and environmentally sensitive harvest control rule designed to prevent biological overfishing. The control rule incorporates a set aside (cut-off) to provide for the forage needs of predator species, and promotes recruitment success through a relationship between F_{MSY} for Pacific sardine and sea surface temperature. The harvest policy also includes a CPS finfish limited entry programme to prevent economic overfishing and to promote efficient harvesting.[18]

At this point, however, development of an optimal US harvest policy for Pacific sardine is incomplete. Lacking is a rational allocation of the estimated ABC among regional fisheries within the US, as well as among the US, Mexico and Canada fisheries. On the domestic front, the PFMC has been making progress towards establishing a long-term framework for allocating the US portion of the ABC between US fishery sectors.

From a strictly economic perspective, the coastwide Pacific sardine ABC would be allocated between the northern and southern fishery sectors in a manner that maximizes the net national benefits from harvesting and processing the ABC. This would be the resulting allocation if increments of the ABC were put up for competitive bid, so assuring that each increment of ABC went into its highest valued use, and that net national benefits from utilization of the resource were maximized. However, the Council operates within a multiple objective setting, and in this case the optimal harvest policy must also account for the distributional aspects and community impacts of any allocation action. Therefore, any rights-based solution (for example individual transferable quotas, ITQs) would require the Council initially to distribute individual user rights on the basis of each potential recipient's history of participation in the fishery. Under those conditions, the initial allocation of ITQs can be structured to minimize negative community impacts and even to enhance fishing community stability. Given the initial distribution of ITQ shares, the holders can then exercise their user rights, and the anticipated efficiency gains can be realized. This type of ITQ programme can be viewed as a conditional market solution offering significant efficiency gains, while lessening the distributional inequities often associated with a market solution.

Given the current geographic extent of the US Pacific sardine fisheries, the coastwide allocation framework should also:

1. assure fuller use of the ABC by eliminating, or at least greatly reducing, the risk of an early closure of the Pacific Northwest fishery, with minimal risk of early closure for the traditional Californian fisheries;
2. provide considerable gains in producer surplus in Pacific Northwest fisheries, which report strong markets, increasing demand, and higher product prices than in California. It is also expected to provide considerable increases in Pacific Northwest employment and income, while

resulting in either no risk or minimal risk of disruption to other fishery sectors;

3. recognize the historic dependence on the Pacific sardine resource of Californian fisheries and fishery communities (see below);
4. not significantly impact nor disrupt the limited entry fishery;
5. ensure stability in all fishery sectors at the peak of their respective seasons.

The question remains as to what happens if there is a change in environmental conditions that might cause the biomass to shrink and the distribution of sardine to contract southwards. It is likely that at least some of the larger sardine that have in recent years been centred in the Pacific Northwest will contract into the Californian fishing areas as the stock(s) retreat south. Industry members recall that the average size of sardine was much larger when the fishery resumed off California in the 1980s. Under such circumstances it is doubtful that the Pacific Northwest fishery would endure, because it is built on the availability of large sardine harvested and processed for highly discriminating export markets. The cost of maintaining those markets in the absence of readily available raw product would probably be prohibitive.

However, the economic fallout from a collapse of the modern-day Pacific Northwest sardine fishery is likely to be minimal. Industry recognized that the fishery was fraught with risk and uncertainty to begin with, with a fairly ephemeral resource base and specialized markets. Compounding the problem is the possible renewal of the Pacific sardine fishery off Japan. Accordingly, investment in the Pacific Northwest sardine fishery was considered a relatively short-term venture, with fishers and processors banking on not much more than a 15-year duration. Therefore, those fishers and processors in the Northwest that chose to specialize in Pacific sardine were fairly well positioned to recoup their investments, and were reasonably prepared for a well-timed exit.[19] Correspondingly, the established processors have other species/product lines to fall back on (for example groundfish and salmon) should sardine disappear to the south. Likewise, many of the vessels participating in the fishery were also permitted to participate in the Californian CPS fisheries, so they have some options there as well.

Even if the distribution were not to retreat south in the event of a shrinking biomass, there would still be a reduction in the ABC under the existing harvest control rule. At some point the ABC might be reduced to a level that would fully satisfy the needs of the Californian fisheries. If this was the case, the Californian fisheries would probably press hard for sole utilization of the ABC, based on their historical dependence on the species. Harvesters and processors in the Pacific Northwest do not have the long history with sardine that those in San Pedro and Monterey have. Even in the heyday of the Pacific sardine fishery, the Pacific Northwest harvest encompassed little more than a decade – beginning at the height of the fishery in 1935–1936 and peaking in 1937–1940. Therefore, a shortage of sardine in the south is likely to have greater unfavourable economic and community impacts than a shortage in the Northwest, especially if the Northwest can more readily shift into alternative fisheries. This argument will certainly not be lost in the Council arena, where community impacts carry considerable weight as part of multiple objective fisheries conservation and management. It could even happen that the ABC would be reduced to a level that would justify the re-establishment of the north-south boundary at 35°40′N, to prevent the southern Californian fishery from pre-empting the northern fishery.

It is also possible that, unrelated to a change in biomass, the harvest guideline could dramatically decrease in the short term if sea surface temperature continues to decline (Figures 5.3, 5.4). In that case, the same line of reasoning presented above could be invoked to justify the ABC going entirely to California.

A long-term framework resolution of the domestic fisheries allocation issue is a necessary, but not necessarily a final, step in achieving an optimal harvest policy for Pacific sardine. Beyond the domestic allocation issues are those relating to the transboundary nature of the resource and optimal allocation of the coastwide ABC on an international basis. Positive steps in this direction have occurred with scientific exchanges between scientists from the three countries. This has highlighted the need and garnered support among the scientific community for cooperative conservation and management at the international level. The realization of an international agreement, however, is more in the political realm, and therefore subject to prolonged and possibly arduous negotiations involving trade-offs between common and private interests among parties.

NOTES

1. Clark and Marr (1955) note that a school of sardine of a given diameter may be denser and contain more fish at lower water temperature than it does at higher temperature, suggesting that catch rates might increase with a decrease in water temperature.
2. Both the state of California and Federal fishery agencies became concerned about the use of sardine for reduction. California passed a number of laws between 1920 and 1941 to curtail the use of whole sardine for reduction; only plants that canned sardine for human consumption could legally reduce the species (Uber and MacCall, 1990). A loophole was discovered that allowed sardine to be reduced outside the state's three-mile jurisdiction, which led to the proliferation of offshore, floating reduction plants from the mid-1920s and into the late 1930s, when legislation closing the loophole, together with increased operating costs, effectively ended offshore reduction.

3. Normally, a reduction in biomass is associated with a decrease in catch per unit effort, *ceteris paribus*.
4. Given this phenomenon, Radovich (1973) hypothesized that the fishery experienced an increase in its catchability coefficient as the biomass decreased. MacCall (1976) tested this hypothesis empirically, and found that catchability increased with a reduction in size of the Pacific sardine population.
5. California sardine quotas were set using a formula identical to the preferred option in *Amendment 7 to the FMP for Northern Anchovy*, which was rejected by NMFS (Pacific Fishery Management Council, 1998).
6. Although most CPS species exhibit dramatic changes in abundance, even in the absence of a commercial fishery, they rarely fluctuate in the same direction simultaneously (that is when one species is low in biomass, this will be offset by another species being high in biomass). From an economic standpoint this collective trend in overall availability is apt to offset the volatility in many of the markets for CPS, because individual CPS are frequently substitutes for each other.
7. The term H is equivalent to the acceptable biological catch (ABC) from the fishery. ABC is a prudent harvest level based on an MSY control rule. ABC will be less than or equal to MSY.
8. Optimum yield (OY) is a decision mechanism for resolving multiple conservation, social and economic goals. Determination of OY involves consideration of ecological, economic and social factors, but biological factors and sustainability are most important (US Department of Commerce, 1996). The OY for Pacific sardine is the level of harvest, less than or equal to ABC, that achieves maximum benefits from the resource while adhering to ecosystem-based principles of resource utilization.
9. The distribution of west coast CPS and their principal spawning grounds are centred in the southern California Bight and off northern Baja California. During both the heyday of the sardine fishery (1915 through 1945) and recently, most of the west coast sardine harvests have been from waters south of 39°N (Point Arena, California). However, during periods of relatively warm water, sardine abundance escalates, and its distribution will extend north of 39°N. Under such circumstances, additional fishing opportunities in the northern area could be most effectively pursued under open access. When the water cools and the biomass contracts, sardine fishing opportunities are not expected to be available in northern areas, and an open access fishery there would be effectively 'turned off'. Economic efficiency is improved by capping or reducing the number of vessels participating in the customary fishery, while allowing for additional vessels when the need arises in the north.
10. DEA provided an output-orientated measure of each vessel's capacity utilization, its observed output per trip relative to the output per trip that would place it on the best practice frontier (Farrell, 1957; Färe *et al.*, 1989, 1994).
11. Because a vessel's physical capacity can range widely within length categories depending upon breadth and depth of the hull design, we calculated a vessel's gross tonnage as $GT = 2/3(\text{length} \times \text{breadth} \times \text{depth})/100$.
12. It is probably more reasonable to expect that peak-period conditions would apply to a few trips throughout the year, or be seasonable in nature, particularly given alternative fishing opportunities for the CPS finfish fleet. Hence, physical capacity would only be required for selected trips, or only during part of the year.
13. Oregon and Washington actively manage the Pacific NW fishery, in part because of the heightened potential for salmon by-catch. States can impose management measures on their residents as long as they are compliant with those contained in the federal FMP. Both Oregon and Washington have limited entry programmes for Pacific sardine and collect information about landings and the environmental effects of these fisheries under their respective developmental and emerging commercial fishery authorities.
14. The re-emergence of the NW sardine fishery was not unexpected given the climate-induced northward expansion of sardine biomass, enhanced by the warm water *El Niño*/Southern oscillation (ENSO) event of 1997–1998.

15. Tagging studies (Clark and Janssen, 1945) and data from the fisheries indicate that, when Pacific sardine abundance is high, relatively larger fish are found in the north.
16. The reduced size of individual sardine moving north to south may be related to density-dependent factors; there have been recent indications of changes in maturity rates (that is delayed maturity) in the southern fishery.
17. The interim measure was put in place for 2003, 2004, and conditionally for 2005.
18. The limited entry programme also provides for easier transition from one form of fishery management to another, in that the reduced number of vessels makes the fishery much more manageable.
19. At the time, there was a surplus of plant and equipment in the area suitable for sardine processing.

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6. Long-term harvest strategies for small pelagic fisheries under regime shifts: the South African fishery for pilchard and anchovy

José De Oliveira

INTRODUCTION

Populations of small pelagic fish species in Eastern Boundary Currents (for example Humboldt, Benguela, California and Canary) and other systems (for example Japan) are characterized by 'regime shifts', in which the average abundance of small pelagic species such as sardine and anchovy (both in absolute terms and relative to one another) undergo extensive changes, typically on a timescale of several decades (Lluch-Belda *et al.*, 1989; Lluch-Cota *et al.*, 1997). This interdecadal variability is caused by poorly understood environmental forcing and creates substantial problems for the fishing industry and for fisheries management, leading to changes in average annual catches of each stock that can exceed an order of magnitude. The annual fluctuations impact on fishing practices, marketing and management. However, the underlying long-term changes are difficult to detect in the short term, because they are masked by both interannual variability and uncertainty in estimates of abundance. Incorporating some knowledge of the current status of a regime shift or cycle into management systems for small pelagic fish may provide substantial benefits in terms of planning, yield and stability for the fishing industry and managers in the medium to long term.

Environmentally driven long-term changes in fish populations, which can play a major role in determining how such populations respond to fishing pressure, are rapidly being recognized as a critical problem in fisheries science (MacCall, 2002). This is because these long-term changes cause severe confusion in management systems when assessment scientists are unable to distinguish between impacts that result from the environment, and those that result from fishing (Walters and Parma, 1996). The 'environment versus fishing' debates that ensue from such confusion can lead to damaging delays in corrective action, particularly as there is little prospect that fisheries research will provide sufficiently fast answers to such debates (Walters and Collie, 1988). Under these circumstances, Walters and Parma (1996) suggest the development of management strategies that are robust to causes of fluctuations, are easily understood by all stakeholders, and are practical.

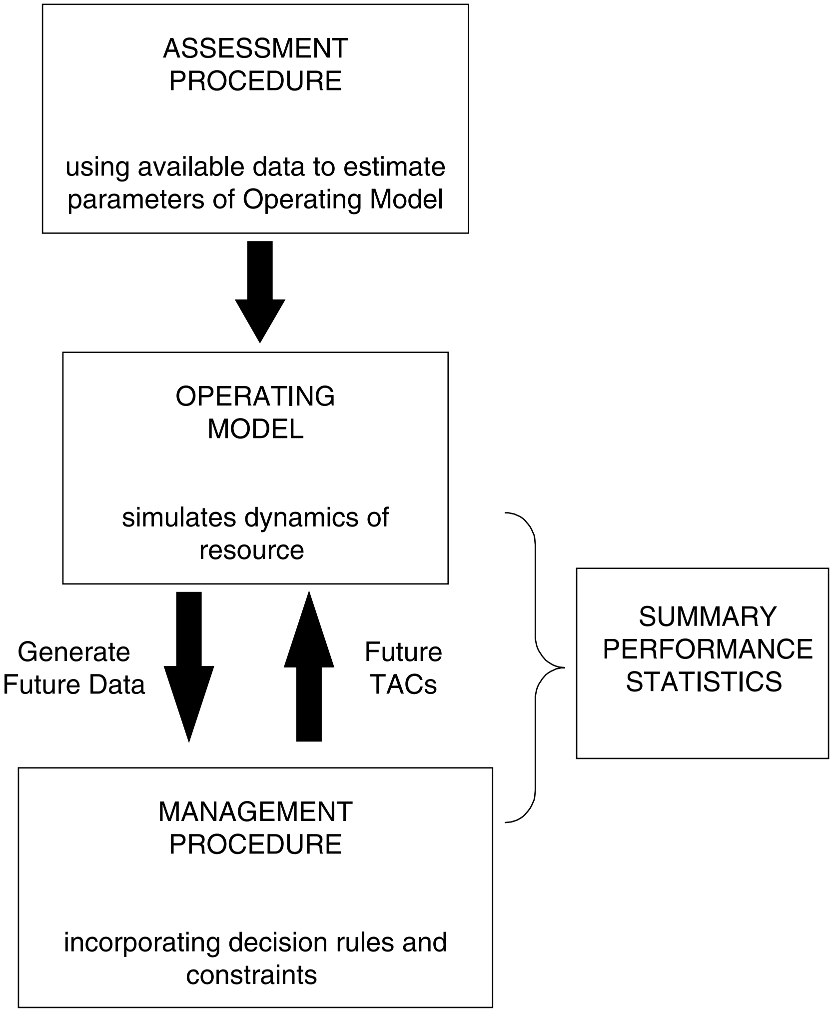
In the spirit of Walters and Parma (1996), this chapter aims to use the Management Procedure (MP) approach (Kirkwood, 1992, 1997; Butterworth *et al.*, 1997; Kell *et al.*, 1999; McAllister *et al.*, 1999; Smith *et al.*, 1999) to investigate different harvesting strategies in the South African fishery for pilchard (*Sardinops sagax*, also called sardine) and anchovy (*Engraulis encrasicolus*, formerly *E. capensis*), in order to identify improved general approaches to management in the face of both short- and long-term uncertainty.

METHODS

The MP approach consists of a simulation-testing framework whereby the performance of rules used to manage a fishery (for example formulae used to set Total Allowable Catches – TACs – noted as a 'Management Procedure' generating 'Future TACs' in Figure 6.1) is evaluated. The framework includes an 'Operating Model' that simulates the 'true' dynamics of the resources, and provides the MP with future data ('Generate Future Data') subject to the types of uncertainties encountered in practice. Several operating models may be considered, but these are generally conditioned on available data ('Assessment Procedure'). The performance of alternative MPs, given a particular operating model, is evaluated through the use of 'Summary Performance Statistics'. Butterworth *et al.* (1997) provide a more detailed description of MPs, whereas McAllister *et al.* (1999) place the MP approach in a more general context, and De Oliveira *et al.* (1998a) and Butterworth and Punt (1999) relate some experiences in the application of MPs.

Operating Models

Operating models for pilchard and anchovy are described in detail in De Oliveira (2003, but see also De Oliveira and Butterworth, 2005). They are essentially age-structured difference models with pulse fishing (Hilborn and Walters, 1992), and with recruitment drawn from a 'hockey-stick' stock-recruit relationship (defined by two straight lines, the one of positive slope through the origin intersecting the other, a horizontal line; Barrowman and Myers, 2000), incorporating recruitment serial



Note: TAC = total allowable catch.

Figure 6.1 A version of the MP testing framework as currently used for developing MPs to be applied in the South African fishery for pilchard and anchovy correlation. The operating models considered here differ from one another only by the parameters used to model regime cycles. In particular, regime cycles are modelled by calculating K_y^i (the adult biomass for which the no-exploitation replacement line intersects the stock-recruit curve) in the following manner:

$$K_y^i = a_y^i e^{\frac{1}{2}(\sigma_r^i)^2} \left[\sum_{n=1}^{n^i} \bar{w}_n^i e^{-M_{ju}^i-(n-1)M_{ad}^i} \right] \tag{6.1}$$

where $n^P = 5$, $n^A = 4$, and[1]

$$a_y^P = a^P e^{(\ln g_1)\sin(360(y-1998)/g_2)}$$

$$a_y^A = a^A e^{-(\ln g_1)\sin(360(y-1998)/g_2)} \tag{6.2}$$

where M_{ju}^i and M_{ad}^i are juvenile and adult natural mortality for species i (P for pilchard and A for anchovy), $\bar{w}_n^i$ are the population mean mass-at-age, the σ_r^i and a^i are from equation (6.3) below, g_1 is the amplitude factor for the regime cycle (for example $g_1 = 2$ means that a_y^i will fluctuate between $2a^i$ and $0.5a^i$), and g_2 is the period of the cycle.

Furthermore, future recruitment is generated from a hockey-stick stock-recruit curve as follows:

$$N_{y,0}^i = \begin{cases} a_y^i e^{\varepsilon_y^i \sigma_r^i}, & \text{if } B_{y,N}^i \geq b_y^i \\ \dfrac{a_y^i B_{y,N}^i}{b_y^i} e^{\varepsilon_y^i \sigma_r^i}, & \text{if } B_{y,N}^i < b_y^i \end{cases} \tag{6.3}$$

where

$$\varepsilon_y^i = s_{cor}^i \varepsilon_{y-1}^i + \sqrt{1-(s_{cor}^i)^2}\, \omega_y^i, \quad \omega_y^i \sim \mathrm{N}[0;\,1]$$

with σ_r^i reflecting variability about the stock-recruit curve, s_{cor}^i recruitment serial correlation, and N[0; 1] a normal distribution with mean 0 and variance 1. Furthermore, a_y^i and b_y^i are parameters of the hockey-stick model, with b_y^i calculated in the same way as a_y^i (equation 6.2), and $B_{y,N}^i$ is the 'true' adult biomass computed by the operating model.

The motivation for using sinusoids as the class of regime shifts considered in this chapter comes from studies on rates of deposition of fish scales, which show regime shifts to be roughly periodic for pilchard and anchovy in the California current system (Soutar and Isaacs, 1969; Baumgartner *et al.*, 1992). Furthermore, using spectral analysis and removing low frequency variability (periods >150 years) from the original series of pilchard and anchovy scale-deposition rates by applying a low-pass filter, Baumgartner *et al.* (1992) found a dominant 57-year peak in the resultant spectra for both species. This would suggest the use of a Fourier series, with a sinusoid as a dominant first-order term and a period of 50–60 years to model regime shifts. The analysis of Baumgartner *et al.* (1992) did not, however, consider periods shorter than 50 years owing to the sample resolution of their data (10 years). Because of the potentially more serious impact of cycles with shorter periods on short- to medium-term management, it was decided to consider sinusoids (termed cycles below) with periods (g_2) of 30 or 50 years in the analysis that follows.

The amplitude of the cycle is reflected by the amplitude factor g_1, and the following values were considered: $\{1; 1.2; 1.5; 2\}$. A value of $g_1 = 1$ means that there are no regime cycles (so g_2 need not be specified in this case). A value of $g_1 = 2$ reflects a doubling (at the peak of the cycle) and halving (at the trough) of the functions used to generate recruitment (equations 6.2 and 6.3). (Note the additional variation in recruitment arising from equation (6.3) over and above the regime cycle of equation (6.2). This means that, for $g_1 = 2$ (say), a 'good' recruitment event generated at the peak of the regime cycle could be much more than just double the very long-term average recruitment.)

Equation (6.2) is written so that the cycles commence in 1998, initially increasing for pilchard and decreasing for anchovy. The reason this is done is to capture the situation at the end of the 1990s, which suggests a shift towards pilchard dominance (De Oliveira and Butterworth, 2004).

SUMMARY PERFORMANCE STATISTICS

Summary performance statistics are used to evaluate a range of MP variants and to investigate their robustness to different underlying operating models. They are calculated from quantities of interest derived from 500 simulations of 100-year projections, with the 100-year trajectories differing from one another through stochastic variation (given by the error components of the operating model and the information passed to the management procedure – Figure 6.1). The summary statistics can be categorized under 'resource conservation' and 'economic performance'. (For ease of presentation, superscripts indicating P/A for pilchard/anchovy have been omitted below.)

Resource Conservation

Two summary statistics are considered here, namely:

- *risk* – the probability that adult biomass falls below 20 per cent of K_y (the average adult biomass in the absence of exploitation, which varies over time) at least once during the projection period; and
- *depl* – 'depletion', which measures the average adult biomass at the end of the projection period as a proportion of K_y at the time concerned (equation 6.1).

Economic Performance

Four summary statistics are considered here, namely:

- $\bar{C}$ the average of the annual total catch (C_y) over the projection period;
- V the mean annual change in C_y;
- *NPV* the Net Present Value of future profits, calculated as an average (see Appendix 6.1); and
- *Loss* the proportion of years that experience zero or negative profit.

Performance of an MP is generally considered to be best when *risk*, V and *Loss* are minimized, *depl* is kept as high as possible, and $\bar{C}$ and *NPV* are maximized.

MANAGEMENT PROCEDURES

The MPs considered in this chapter are based on the joint MP used for the South African fishery for pilchard and anchovy fishery from 1999, OMP99 (De Oliveira, 2003; De Oliveira and Butterworth, 2004), but with a few modifications.[2] A key feature of this joint MP is that it accounts for the operational interaction between the catches of the two species (essentially, juvenile pilchard are taken as a by-catch in the anchovy fishery) by setting a Total Allowable By-catch (TAB) of pilchard based on the size of the anchovy TAC and the pilchard: anchovy ratio in the commercial catches. One of the outcomes of OMP99 was the underutilization of anchovy if a viable directed pilchard fishery (comprising mainly adult fish of suitable canning size) was desired, a problem which was partly solved by the introduction of an additional sub-season for anchovy later in the year, by which time the by-catch of juvenile pilchard is deemed less problematic (De Oliveira and Butterworth, 2004 – the additional sub-season was not formally included in OMP99). OMP99 is described in Table 6.1, and the modified version used in this chapter is referred to throughout as BL (baseline). De Oliveira and Butterworth (2004) provide a brief description of the fishery and the population dynamics of the two resources.

If management in the presence of regime cycles is to succeed, information is required about the underlying position in the regime cycle at any given time. One can then decide how to manage the resources using this information. When investigating alternative MPs, this chapter has therefore considered two aspects, namely defining estimators that provide information about the regime cycle, and developing decision rules (and hence alternative MPs) that use this information.

Estimators

Two approaches were considered, denoted D1 and D2.

D1

This approach assumes that no direct information on the regime cycle is available and therefore uses an indirect method to obtain the necessary information. The approach simply calculates, in year y, an x-year running mean ($\mu_{y,x}$) and an overall mean (μ_y) of the survey estimates of pilchard adult biomass (Table 6.1) as follows:

$$\mu_{y,x} = \frac{1}{x} \sum_{j=y-(x-1)}^{y} B^{P}_{j,Nov}$$

$$\mu_y = \frac{1}{y-1983} \sum_{j=1984}^{y} B^{P}_{j,Nov} \tag{6.4}$$

(In equation (6.4), the actual survey estimates for the years 1984–1996 were used. Thereafter, simulated estimates were substituted.) The method then calculates the following:

$$d_{y,1} = \frac{\mu_{y,\,x}}{\mu_y} \tag{6.5}$$

Values of x considered are $\{2; 4; 6; 8; 10; 12; 14\}$. Figure 6.2 illustrates a typical simulation run where $\mu_{y,x}$, and μ_y are calculated for ‘true’ adult biomass values,[3] both with and without regime cycles for both pilchard and anchovy. The optimal choice for x differs, depending on the MP option used, as discussed later.

Table 6.1 *Management Procedure rules in the form of TAC/B equations and constraints for OMP99, together with definitions of all symbols*

Month	Parameter	TAC/B equations	TAC constraints
January	Pilchard-directed TAC	$TAC_y^P = \beta B_{y,Nov}^P$	$\max\{(1-c_{mxdn}^P)TAC_{y-1}^P;\ c_{mntac}^P\} \leq TAC_y^P \leq c_{mxtac}^P$
	Anchovy 1st TAC†	$TAC_y^{1,A} = \alpha\delta 300\left(0.7 + 0.3\dfrac{B_{y,Nov}^A}{\bar{B}_{Nov}^A}\right)$	$\max\{(1-c_{mxdn}^A)TAC_{y-1}^{2,A};\ c_{mntac}^A\} \leq TAC_y^{1,A} \leq c_{mxtac}^A$
	Pilchard 1st TAB	$TAB_y^{1,P} = \gamma TAC_y^{1,A} + TAB_{rh}^P$	
May/June	Anchovy 2nd TAC†	$TAC_y^{2,A} = \alpha 300\left(0.7\,\dfrac{N_{y,rec0}^A}{\bar{N}_{y,rec0}^A} + 0.3\,\dfrac{B_{y,Nov}^A}{\bar{B}_{Nov}^A}\right)$	$\max\{(1-c_{mxdn}^A)TAC_{y-1}^{2,A};\ TAC_y^{1,A};\ c_{mntac}^A\} \leq TAC_y^{2,A}$ $\leq \min\{c_{mxtac}^A;\ TAC_y^{1,A} + c_{mxinc}^A\}$
	Pilchard 2nd TAB	$TAB_y^{2,P} = \lambda TAC_y^{1,A} + r_y(TAC_y^{2,A} - TAC_y^{1,A}) + TAB_{rh}^P$ where $\lambda = \max\{\gamma, r_y\}$	

Definition of symbols

$B^i_{y,Nov}$	Adult biomass at the beginning of year y, from the simulated November survey in year $y-1$ for species i (1000 tonnes; $i=P$ for pilchard or A for anchovy)
$\bar{B}^A_{Nov}$	The mean of the actual $B^A_{y,Nov}$ estimates for anchovy for $y=1985, \ldots, 1998$ (945 000 tonnes), which unlike $\bar{N}^A_{y,rec0}$ is not updated annually
r_y	Simulated average of the juvenile pilchard to anchovy ratio in the commercial catches in May and in the recruit survey, in year y. (Although OMP99 did not restrict r_y, it is not allowed to exceed 0.5 in this chapter in order to avoid a situation where the pilchard by-catch with anchovy is much larger than the anchovy catch itself, particularly as a result of regime shifts.)
$N^A_{y,rec0}$	Simulated anchovy recruitment estimate from the midyear recruit survey, back-calculated to the start of the year by taking fishing and natural mortality into account (in billions of fish)
$\bar{N}^A_{y,rec0}$	Mean of the estimates of $N^A_{y,rec0}$, updated annually ($\bar{N}^A_{2001,rec0}=173$ billion fish)
TAB^P_{rh}	Fixed tonnage of adult pilchard by-catch set aside for the round-herring fishery each year (set at 10 000 tonnes)
α, β, γ	Control parameters that are fixed for the period over which a selected MP is implemented (initially set at 0.452, 0.1375 and 0.109)
δ	The ‘scale down’ factor used to set a lower anchovy initial TAC to provide a buffer against possible poor recruitment later in the season (0.85)
c^i_{mntac}	Minimum TAC to be set for species i (P: $\varepsilon \times 70\,000$ tonnes; A: $\alpha \times 200\,000$ tonnes, where $\varepsilon=\beta/0.1$ if $\beta<0.1$, and $\varepsilon=1$ otherwise)

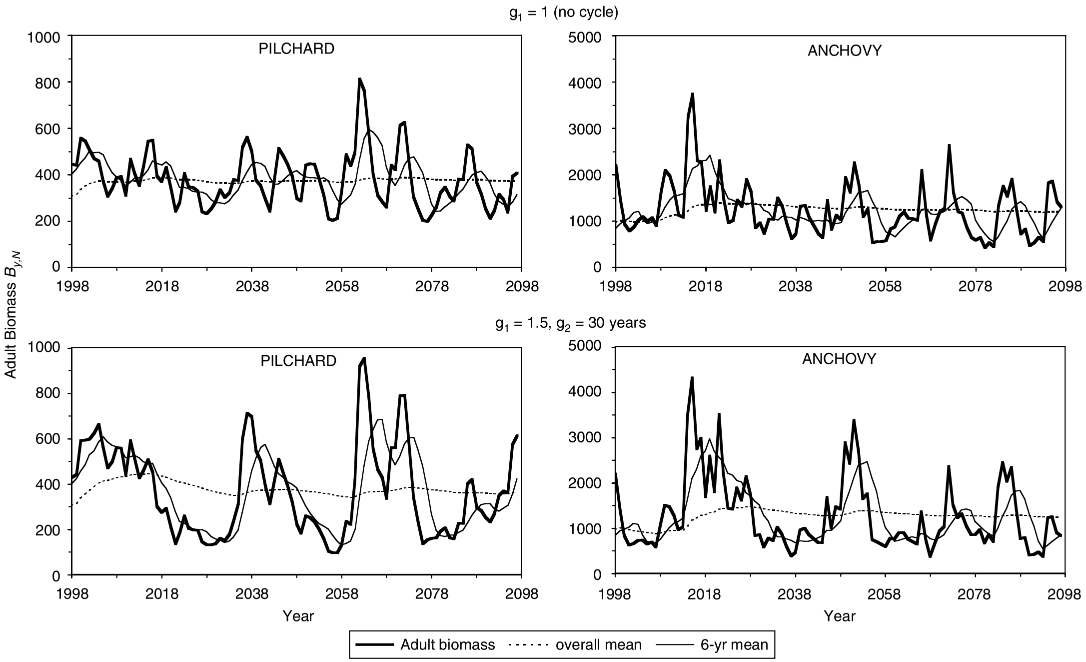
Table 6.1 *(continued)*

Definition of symbols	
c^{i}_{mxtac}	Maximum TAC to be set for species *i* (P: $\varepsilon \times 210\,000$ tonnes; A: $\alpha \times 600\,000$ tonnes, where ε is as above)
c^{i}_{mxdn}	Maximum proportion by which the TAC for species *i* can be reduced from one year to the next (P: 0.25; A: 0.4)
c^{A}_{mxinc}	Maximum by which the anchovy TAC is allowed to be increased within the year ($\alpha \times 150\,000$ tonnes)

Notes:
[†] The values 300, 0.7, and 0.3 come from previous OMP implementations, and were not newly specified for OMP99 (Butterworth *et al.*, 1993; De Oliveira, 1995). Furthermore, the recruit survey component in these equations is given more weight because of the importance of the incoming recruits to the year's catch.

'Exceptional Circumstances' provisions exist, so the TAC and associated constraint calculations are adjusted downwards if the November survey results for pilchard or anchovy (when calculating first-stage TACs), or if the projected survey result for anchovy (when calculating second-stage TACs for anchovy) are below certain threshold levels (150 000 tonnes for the pilchard survey result, and 400 000 tonnes for both the actual and projected anchovy survey results). De Oliveira (2003) provides more details of the meta-rules for 'Exceptional Circumstances', which have been omitted from the equations above for simplicity. Adjustments in TACs during the year, if applicable, may only be upwards, not downwards, because by the time the second TAC is set, the first TAC could have been fully caught.

Source: Based on De Oliveira (2003).



Note: Each of these trajectories is smoothed with an overall mean of all past values, updated annually, and with a 6-year running mean. The top panel shows the trajectories without any cycle ($g_1 = 1$), while the bottom panel shows the trajectories for a cycle of period 30 years and an amplitude factor of 1.5 ($g_1 = 1.5$, $g_2 = 30$ – see equation 6.2).

Figure 6.2 A typical example of simulated 100-year trajectories of 'true' adult biomass (in the operating model), which results from an application of BL, for pilchard and anchovy

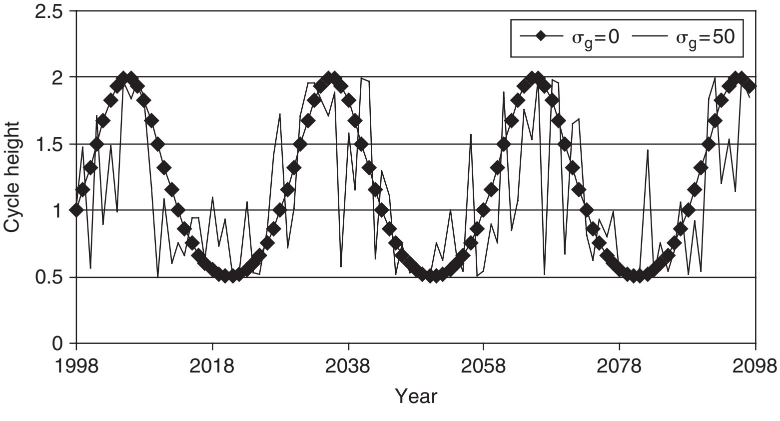
D2

This approach assumes that it is possible to obtain direct information on the regime cycle (although it does not specify how). It uses the actual position of the resource in the cycle (equation 6.2), but subject to a varying extent of measurement error. The method calculates the following:

$$d_{y,2} = e^{(\ln g_1)\sin(360(y-1988)/g_2+\sigma_g\eta)} \tag{6.6}$$

where η is drawn from N[0; 1], and σ_g reflects the maximal uncertainty as to where in the cycle one is at any time. Values of σ_g considered (in degrees) are {0; 10; 30; 50; 100}. A value of $\sigma_g = 0°$ implies that the position in the regime cycle is known exactly, while $\sigma_g = 10°$ implies roughly a 95 per cent certainty that one is within 20° of the actual position in the cycle. Figure 6.3 shows the effect of increasing σ_g (from 0° to 50°).

Both D1 and D2 consider only pilchard when calculating $d_{y,1}$ and $d_{y,2}$ respectively. The reason for this is that the pilchard TAB is dependent on the size of the anchovy TAC (Table 6.1). Because the two species follow cycles that are 180° out of phase (equation 6.2), poor pilchard recruitment coincides with high anchovy abundance and therefore increased by-catch pressure. This magnifies the vulnerability of the pilchard stock in the troughs of the pilchard cycle, making it more important to track the



Note: For this example, the corresponding cycle without error for anchovy (not shown) is identical to that for pilchard, but lags behind the pilchard cycle by 180°.

Figure 6.3 An illustration of a regime cycle for pilchard with an amplitude factor (g_1) of 2 and a period (g_2) of 30 years, with and without measurement error, σg (equation 6.6) pilchard cycle. The pilchard resource also consists of more age classes and is less variable than anchovy (De Oliveira, 2003), which means that pilchard will be less influenced by random fluctuations than anchovy.

Alternative MP Options

The current approach to managing pilchard and anchovy in South Africa accepts that a pilchard by-catch is inevitable, and that the purse-seine industry has not in the past been able to keep pilchard by-catch levels low (particularly juvenile pilchard occurring with anchovy) (Cochrane *et al.*, 1998). Pilchard by-catch is therefore regarded as a 'non-negotiable' component of BL, and is set in the MP to realistic levels by use of an estimate of the by-catch ratio for the year concerned, r_y, based on direct measurements of the juvenile pilchard to anchovy mix in both the recruit survey and commercial catches. Except for being restricted to a maximum of 0.5, this ratio is not 'controlled' in the MP. Pilchard by-catch can only be controlled by reducing anchovy catches (reducing α), reducing pilchard-directed catches (reducing β, which thereby allows for a greater anchovy catch), or reducing the initial by-catch ratio, γ, which is not based on any information on the likely mix of juvenile pilchard to anchovy (as this information is not available at the start of the year; Table 6.1).

The alternative MPs considered in this chapter centre around the three control parameters, α, β and γ, and the estimators of $d_{y,j}$ ($j = 1$ or 2), and are as follows:

M1

This MP reduces anchovy catches as $d_{y,j}$ falls below 1. This is an effective means of reducing pilchard by-catch, but leads to greater underutilization of anchovy than BL.

$$\alpha_y^* = \begin{cases} d_{y,j}\alpha, & d_{y,j} < 1,\ j = 1 \text{ or } 2 \\ \alpha, & d_{y,j} \geq 1,\ j = 1 \text{ or } 2 \end{cases} \tag{6.7}$$

α_y^* then replaces α in the equations in Table 6.1 (but not in the corresponding constraints).

M2

This MP reduces the initial by-catch ratio, γ, as $d_{y,j}$ falls below 1. This approach was considered a good potential approach because of the likelihood that mixing of juvenile pilchard and anchovy (and hence by-catch) becomes less of a problem with low pilchard and high anchovy abundance – there has been an increase in the by-catch ratio from 1994 when the two species had similar abundances, but low by-catch ratios prior to 1994 when anchovy was far more prevalent than pilchard (De Oliveira and Butterworth, 2004).

$$\gamma_y^* = \begin{cases} d_{y,j}\gamma,\ d_{y,j} < 1,\ j = 1 \text{ or } 2 \\ \gamma,\ d_{y,j} \geq 1,\ j = 1 \text{ or } 2 \end{cases} \tag{6.8}$$

γ_y^* then replaces γ in the equations in Table 6.1.

M3

This MP reduces the directed pilchard catch as $d_{y,j}$ falls below 1. Previously this has not been considered to be an effective means of managing by-catch, because a large amount of pilchard-directed catch would need to be sacrificed for a small amount of extra by-catch for the same risk (De Oliveira *et al.*, 1998b).

$$\beta_y^* = \begin{cases} d_{y,j}\beta,\ d_{y,j} < 1,\ j = 1 \text{ or } 2 \\ \beta,\ d_{y,j} \geq 1, j = 1 \text{ or } 2 \end{cases} \tag{6.9}$$

β_y^* then replaces β in the equations in Table 6.1 (but not in the corresponding constraints).

M1+2

A combination of M1 and M2 (that is, both anchovy catch and the initial by-catch ratio is reduced as $d_{y,j}$ falls below 1).

M1+2+3

A combination of M1, M2 and M3 (effectively reducing all three control parameters as $d_{y,j}$ falls below 1).

M4

As for M1, but α is also increased as $d_{y,j}$ increases above 1. This option attempts to 'alleviate' the underutilization of anchovy in M1.

$$\alpha_y^* = \begin{cases} d_{y,j}\alpha,\ d_{y,j} < 1 \\ \alpha'(d_{y,j} - 1) - \alpha(d_{y,j} - 2), 1 \leq d_{y,j} < 2 \\ \alpha',\ d_{y,j} \geq 2 \end{cases} \tag{6.10}$$

all for $j = 1$ or 2; α_y^* then replaces α in the equations in Table 6.1 (but not in the corresponding constraints). The values of α' considered are $\{\alpha, 0.6; 0.8; 1\}$. (Note that M1 is a special case of M4, with $\alpha' = \alpha$.)

PRELIMINARY INVESTIGATIONS

r_y and the Projection Period

When applied in 1999 to manage pilchard and anchovy, OMP99 had been selected on the basis of a 20-year projection period with unrestricted r_y. Before continuing, it is important to investigate the effect of restricting r_y to a maximum of 0.5 and lengthening the projection period from 20 to 100 years, these two changes being essentially the only ones made to OMP99 to derive BL in this chapter. Results are shown in Table 6.2. The increase in *risk* for both species when lengthening the projection period is expected, because each resource will be 'exposed to the elements' for a longer period of time, and will therefore have an increased chance of falling below 20 per cent of K at least once for the longer period, compared with the shorter one. The decrease in *risk* for pilchard when moving from an unrestricted to a restricted r_y (r_y does not affect anchovy) is also expected, because of the extra protection afforded to juvenile pilchard when restricting r_y. For the rest of this chapter, BL corresponds to the final row of Table 6.2.

Table 6.2 Summary performance statistics for OMP99, for a 20- and 100-year projection period, and for unrestricted and restricted r_y *(see Table 6.1)*

Changes to OMP99	Pilchard				Anchovy			
	$\bar{C}$ (1000 tonnes)	*risk*	*depl*	*V*	$\bar{C}$ (1000 tonnes)	*risk*	*depl*	*V*
20-year projection, unrestricted r_y	135.3	0.102	0.532	0.230	145.2	0.058	0.861	0.241
20-year projection, $r_y \leq 0.5$	134.8	0.086	0.545	0.229				
100-year projection, unrestricted r_y	129.5	0.460	0.497	0.230	145.9	0.354	0.847	0.229
100-year projection, $r_y \leq 0.5$	129.6	0.414	0.514	0.230				

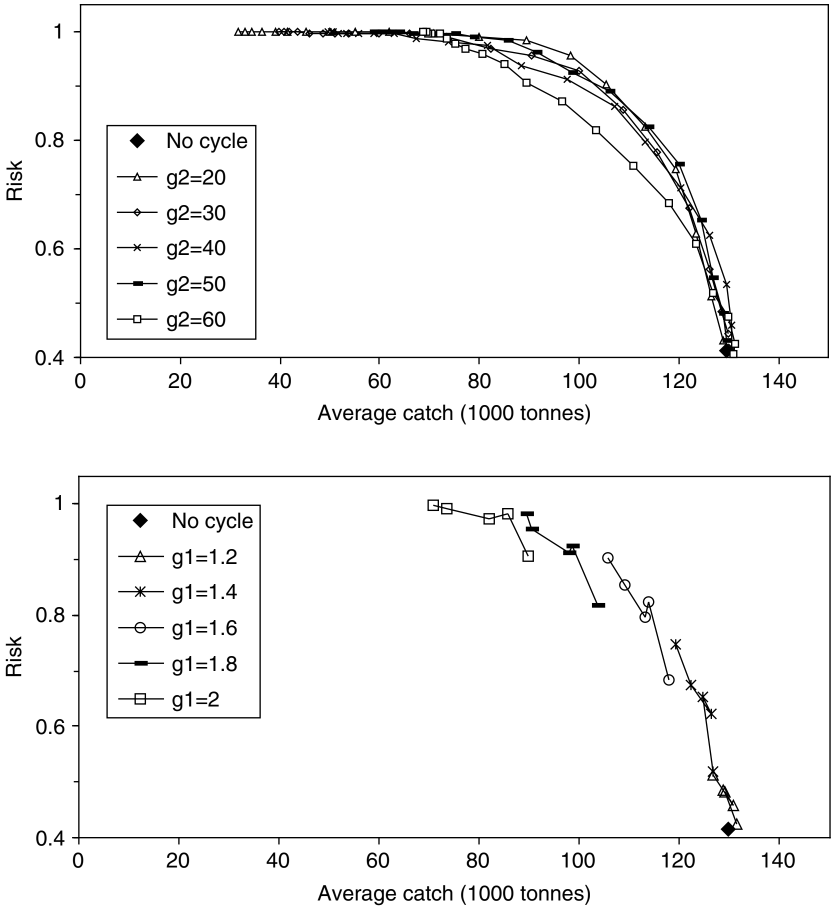
Note: Empty cells acquire the corresponding value in the first non-empty cell above them in the same column.

g_1 and g_2

Figures 6.4 and 6.5 show the performance of BL in terms of *risk* and the average annual catch $\bar{C}$ for a wide range of alternative operating models (that is, alternative regime-cycle scenarios defined by ranges of g_1 and g_2 values). In catch-*risk* plots of this type, an MP performs best when its performance statistics lie towards the bottom right corner of the plot. Performance deteriorates as the statistics move in a diagonal towards the top left corner of the plot as further operating models are used for testing. However, when the point corresponding to the performance statistic for the MP moves from the top right to the bottom left corner of the plot (or vice versa), overall performance is more difficult to judge because, for example, a decrease in *risk* (which is desirable) is accompanied by a decrease in $\bar{C}$ (which is not desirable). This basis for comparison holds both for plots showing the performance of a particular MP subject to alternative operating models (as in these Figures), and for alternative MPs subject to the same operating model (as in later Figures).

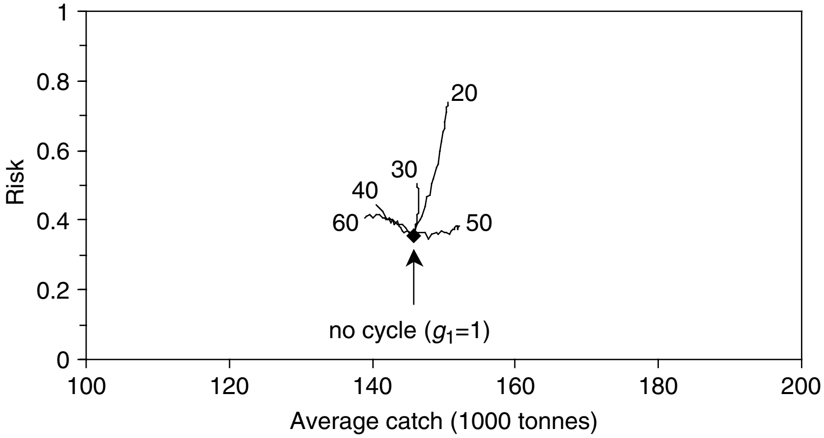
In the top panel of Figure 6.4, each curve corresponds to a particular value of g_2 for a range of g_1 values. The performance of BL for pilchard consistently deteriorates in terms of both *risk* and $\bar{C}$ with increasing g_1. In order to see the effect of changing g_2 on BL more clearly, the bottom panel of Figure 6.4 shows curves corresponding to particular values for g_1, for a range of g_2 values. There is a general pattern of deterioration as g_2 decreases, with $g_2 = 50$ being the exception. This probably results from the balance of peaks and troughs in the 100-year projection period defined by the g_2 value: for $g_2 = 20$ and 50, there are equal numbers of peaks and troughs for pilchard (equation 6.2), whereas there are more peaks than troughs when $g_2 = 40$ and 60 (and to a lesser extent when $g_2 = 30$). This effect could have been reduced by randomizing over the start of the cycle so as to average the effect out, whatever the value chosen for g_2. The argument, nevertheless, for starting the cycle in 1998 with pilchard increasing was to capture the situation at the end of the 1990s, which suggested a shift towards pilchard.

Figure 6.5 shows results for anchovy, which correspond to the top panel of Figure 6.4 for pilchard. In this case, the pattern is a general deterioration of *risk* for increasing g_1, but not necessarily of $\bar{C}$, which improves for $g_2 = 20$ and 50, but deteriorates for $g_2 = 40$ and 60. The reasons for this are likely to be similar to those given for pilchard in terms of the number of abundance peaks and troughs that arise as a consequence of the value selected for g_2.



Note: In the top panel, each curve corresponds to a particular g_2 value, the points within each curve corresponding to different g_1 values ranging from 1.1 (closest to the 'no cycle' option, for which $g_1 = 1$, at the bottom right of the plot) to 3 in steps of 0.1. In the bottom panel, curves are shown for particular g_1 values with points within each curve corresponding to different g_2 values ranging from 20 to 60 years, in steps of 10 years, with 60 being closest to the bottom-right corner for all curves. (Note that, for illustrative purposes, this Figure considers a wider range of g_1 and g_2 values than used elsewhere.)

Figure 6.4 Sensitivity of BL performance statistics for pilchard, when tested with alternative operating models defined by different $\{g_1; g_2\}$ *combinations*



Note: This plot corresponds to the top panel of Figure 6.4, with g_2 values shown in the plot (see caption to Figure 6.4), the g_1 values increasing away from the 'no cycle' point.

Figure 6.5 Sensitivity of BL performance statistics for anchovy

Choice of x for $\mu_{y,x}$

Tables 6.3 and 6.4 investigate the effect of using estimator D1 for MP options M1, M2 and M3 when the operating model has no regime cycle. The aim of this investigation is to determine which period x would be the most appropriate choice to specify the running mean $\mu_{y,x}$ for estimator D1 across all three MPs (and any combinations of these) for both pilchard and anchovy.

Option M1 aims to reduce pilchard by-catch when pilchard abundance is low, by reducing anchovy catches. Table 6.3 shows that pilchard performs better for smaller x values across all summary statistics, except (very slightly) for V. This is because smaller values of x result in D1 being able to respond almost immediately to changes in pilchard abundance. Consequently, occurrences of $d_{y,1}$ below 1 (equation 6.5) increase as x gets smaller. Anchovy catches are therefore reduced more often (equation 6.7), which is always advantageous for pilchard, because there will be a corresponding drop in by-catch. The lower by-catch leads to more efficient utilization of pilchard, and therefore increased directed pilchard catches. However, a higher frequency of reductions of anchovy catches implies that anchovy is further underutilized (*depl* closer to 1) compared with BL, and anchovy catches are slightly more variable (V is higher) for smaller x. Therefore, although a smaller value of x is advantageous for pilchard, this has to be weighed against lower and more variable catches for anchovy.

Table 6.3 Performance statistics for MP option M1 using estimator D1 for various periods x for the running means $\mu_{y,x}$ (equation 6.4) when the operating model has no regime cycle (g_1=1)

Estimator	Pilchard				Anchovy			
	$\bar{C}$ (1000 tonnes)	*risk*	*depl*	*V*	$\bar{C}$ (1000 tonnes)	*risk*	*depl*	*V*
BL	129.6	0.414	0.514	0.230	145.9	0.354	0.847	0.229
D1, $x=2$	133.0	0.318	0.558	0.231	136.2	0.336	0.860	0.239
D1, $x=4$	132.6	0.344	0.553	0.230	138.6	0.330	0.858	0.231
D1, $x=6$	132.1	0.370	0.547	0.230	139.9	0.340	0.857	0.229
D1, $x=8$	131.9	0.382	0.544	0.230	140.8	0.340	0.856	0.228
D1, $x=10$	131.5	0.390	0.540	0.230	141.3	0.346	0.855	0.227
D1, $x=12$	131.3	0.392	0.537	0.230	141.8	0.346	0.855	0.227
D1, $x=14$	131.0	0.394	0.534	0.230	142.1	0.344	0.854	0.227

Note: Results for BL (where estimator D1 is not used) are included for comparison.

Table 6.4 Performance statistics for MP options M2 and M3 using estimator D1 for various periods x for the running mean $\mu_{y,x}$ (equation 6.4), when the operating model has no regime cycle (g_1=1)

Estimator	M2				M3			
	$\bar{C}$ (1000 tonnes)	*risk*	*depl*	*V*	$\bar{C}$ (1000 tonnes)	*risk*	*depl*	*V*
BL	129.6	0.414	0.514	0.230	129.6	0.414	0.514	0.230
D1, $x=2$	134.0	0.358	0.554	0.229	128.9	0.410	0.516	0.227
D1, $x=4$	133.7	0.388	0.551	0.229	128.4	0.408	0.518	0.223
D1, $x=6$	133.5	0.392	0.550	0.229	128.5	0.404	0.518	0.224
D1, $x=8$	132.9	0.394	0.546	0.229	128.6	0.414	0.518	0.225
D1, $x=10$	132.1	0.398	0.539	0.229	128.7	0.412	0.517	0.226
D1, $x=12$	131.8	0.402	0.536	0.229	128.8	0.414	0.517	0.226
D1, $x=14$	131.5	0.404	0.534	0.229	128.9	0.414	0.517	0.227

Note: These options have no effect on anchovy, so the anchovy results are as for BL in Table 6.3 and are therefore not shown. The pilchard results for BL are included for comparison.

Options M2 and M3 avoid the problems of further underutilization and more variable catches of anchovy because they do not reduce anchovy as part of their management action (equations 6.8 and 6.9). Table 6.4 shows similar trends for M2 as for M1: for the former, reducing the initial by-catch ratio (γ in Table 6.1) more often than not leads to lower by-catch, which is always advantageous for pilchard (now all summary statistics including V improve for lower x compared to BL). This is not the case for M3, however, where $\mu_{y,6}$ appears to be the most effective estimator in terms of almost all the summary statistics. This is probably because $\mu_{y,6}$ is the best compromise between the extremes, $\mu_{y,2}$ and $\mu_{y,14}$. For example, $\mu_{y,2}$ responds too rapidly to changes in $B^P_{y,Nov}$, which is subject to measurement error, so it is more readily influenced by random effects, which tend to mask the actual biomass trend. On the other hand, $\mu_{y,14}$ is sluggish in its response, because of the substantial lag behind the actual trend (greater than that shown in Figure 6.2 for $\mu_{y,6}$), so any signal that it provides on trends is delayed and can be very weak. Also, in the operating model used, pilchard is assumed to consist of six age classes (De Oliveira, 2003), so it is not surprising that a six-year mean is a reasonably good indicator of biomass trend, albeit with a slight lag (Figure 6.2).

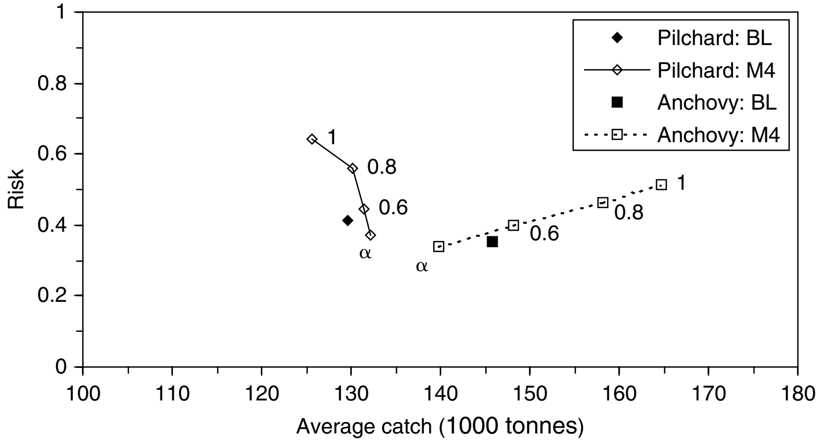
Tables 6.3 and 6.4 and the arguments presented above indicate that M1 and M2 simply demonstrate no more than that any reduction of by-catch (achieved by an $x > 0$) will be beneficial for pilchard. They are not really of much assistance in discriminating between the different options for x. The choice of $x = 6$ for all subsequent D1-based analyses in this chapter was therefore based upon the results for M3.

MP Option M4

Figure 6.6 shows results for M4, which attempts to overcome the underutilization of anchovy under M1 by increasing the exploitation of anchovy when pilchard abundance is high (equation 6.10). The four values of α' considered range from α to 1. The increase in $\bar{C}$ for anchovy is offset by an increase in *risk*, so there is no clear gain for anchovy. There is, however, a marked deterioration with increasing α' for both these summary statistics in the case of pilchard. This option is therefore not considered further in this chapter.

RESULTS

Results are shown in Figures 6.7–6.11. They compare the six MP options BL, M1, M2, M3, M1 + 2 and M1 + 2 + 3 using estimators D1 (with $\mu_{y,6}$) and D2 (with $\sigma_g = 0$, 10, 30, 50 and 100) for a variety of regime cycle



Note: For M4, the α′ values are shown on the plot.

Figure 6.6 Applying estimator D1 for M4 using a 6-year running mean, $\mu_{y,6}$, *for pilchard and anchovy with* $g_1 = 1$

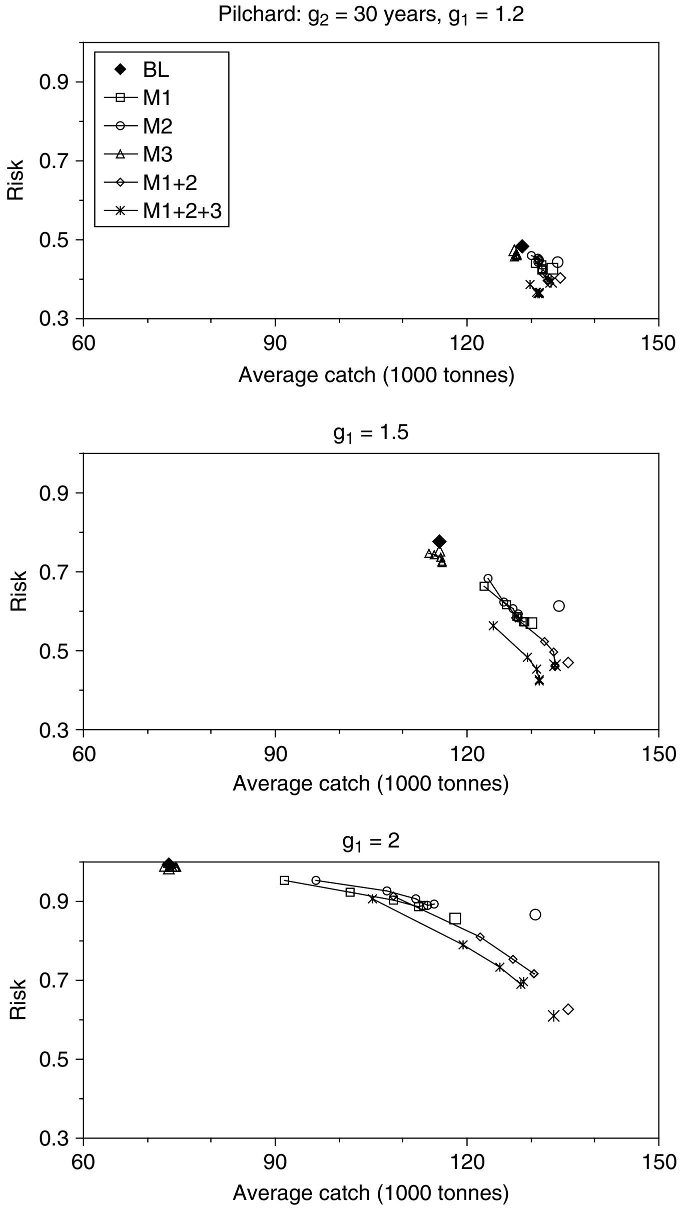
scenarios. Each plot shown (there are 12 plots for each Figure, six each under parts (a) and (b) of each) refers to a particular regime-cycle scenario (reflecting a particular operating model). Moreover, each plot compares a number of MPs, each of which has several sub-options corresponding to the use of estimator D1 or D2. The set of plots under (a) in each Figure correspond to regime cycles of period 30 years (g_2), and under (b) of 50 years.

Risk and Average Catch

To aid interpretation of Figure 6.7, results will be discussed first at the plot level (that is, within each plot), where the performance of individual MPs can be compared with one another, then across plots to consider the effect of increasing g_1, and finally plots in (a) will be compared with those in (b) to investigate the effect of changing g_2.

Comparing the Performances of Alternative MPs

To ease visual interpretation, only the plots in Figure 6.7a for $g_1 = 1.5$ are considered at first. Apart from M3, all MP options perform better than BL for pilchard. This is because, although M3 reduces pilchard-directed catches when pilchard abundance is low, it still fails to solve the by-catch problem, which all the other options attempt to minimize. When σ_g is



Pilchard: g_2 = 30 years, g_1 = 1.2

g_1 = 1.5

g_1 = 2

Note: See Figure 6.7b.

Figure 6.7a Performance of MP options in terms of the summary performance statistics risk and $\bar{C}$

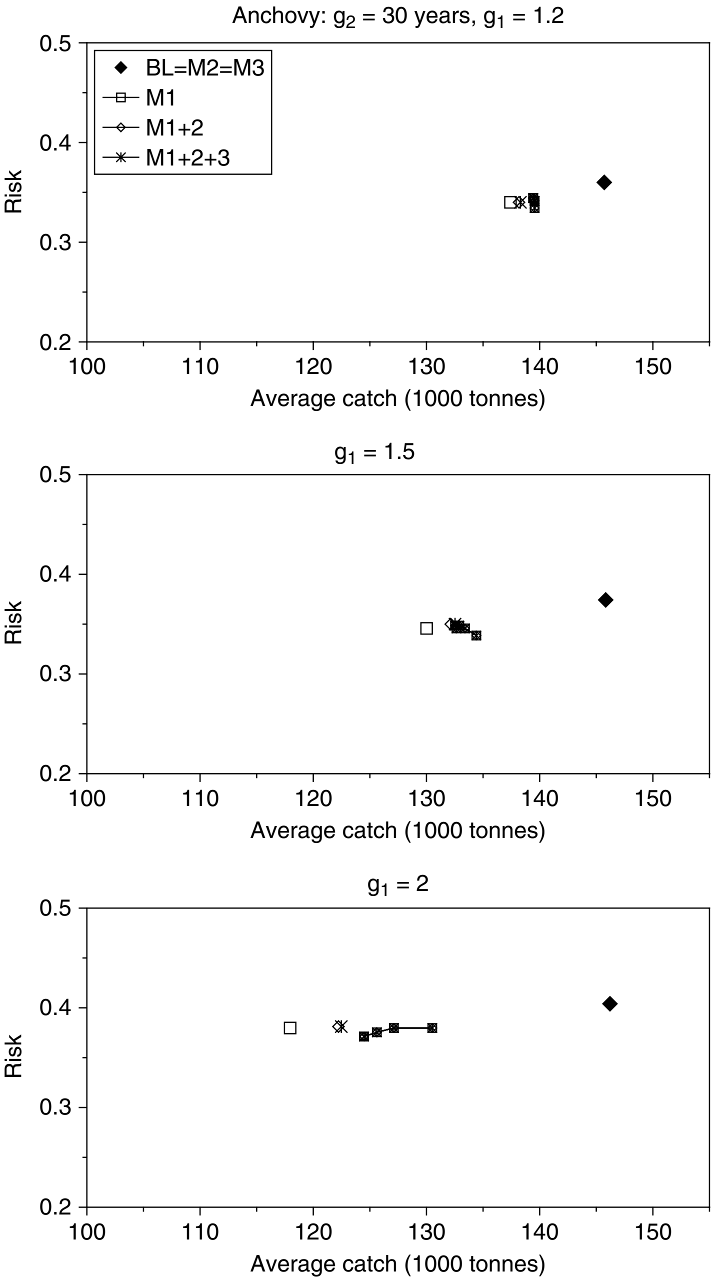
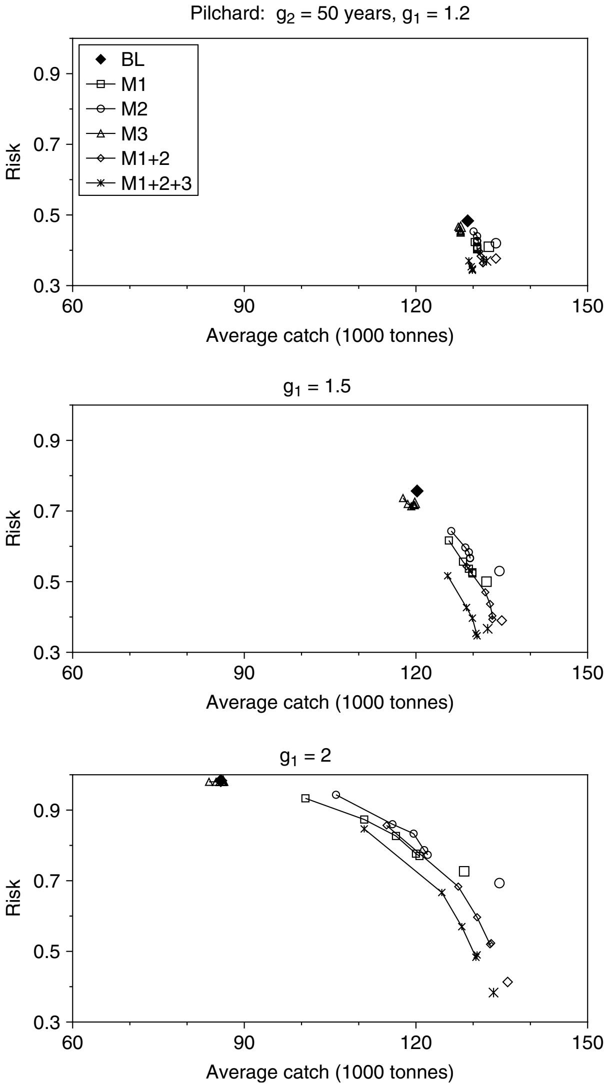


Figure 6.7a (continued)



Note: Points with the same symbol refer to the same MP option, where the larger symbols indicate use of the estimator D1 with $\mu_{y,6}$, and smaller symbols joined by a line refer to estimator D2 for the following σ_g values: 0°, 10°, 30°, 50° and 100° (for pilchard for each MP, $\sigma_g = 0°$ lies closest to the bottom right corner of the plot, with the symbol moving closer to the top left corner as σ_g increases; for anchovy, $\sigma_g = 0°$ has the lowest $\bar{C}$, with $\bar{C}$ increasing as σ_g increases).

Figure 6.7b Performance of MP options in terms of the summary performance statistics risk and $\bar{C}$

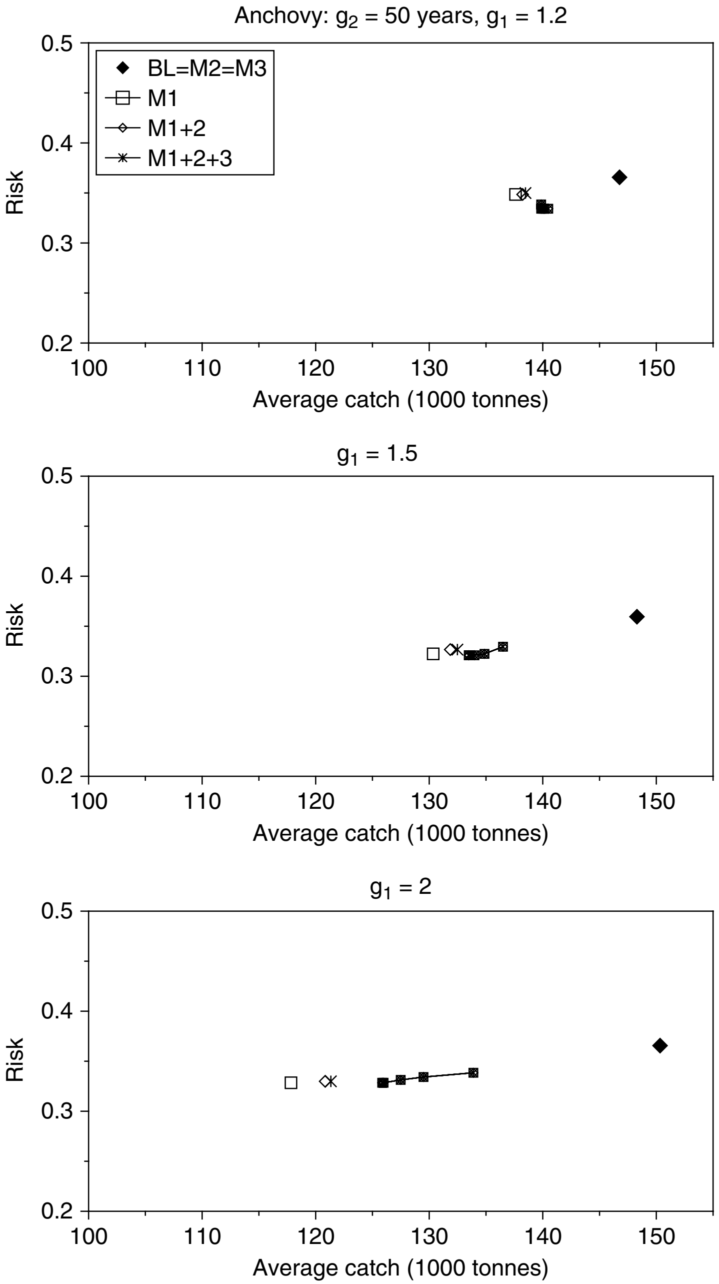


Figure 6.7b (continued)

varied for each option, $\sigma_g = 0$ at the one extreme performs best, while $\sigma_g = 100$ at the other extreme performs worst for pilchard. This is not surprising, because an increasing σ_g implies more uncertainty as to the underlying position in the regime cycle at any time.

Options M1 + 2 and M1 + 2 + 3 perform best overall for pilchard, although it is not clear when comparing these with each other which performs better overall (M1 + 2 performs better in terms of $\bar{C}$, but worse in terms of *risk*). The same argument applies when comparing M1 and M2 with one another. M1 and M2 do not perform as well as M1 + 2 and M1 + 2 + 3 for pilchard, however, particularly in terms of *risk* (although for estimator D1, $\bar{C}$ is slightly worse for M1 + 2 + 3 compared to M2). For anchovy, M2 and M3 (whose performances are identical to BL) perform better than M1, M1 + 2 and M1 + 2 + 3 in terms of $\bar{C}$. This is because they do not reduce anchovy catches (Table 6.4). All options apart from M2 and M3 perform worse for anchovy than BL in terms of $\bar{C}$, but only marginally better in terms of *risk*.

g_1 Increases

Figure 6.7a shows a marked deterioration, in terms of *risk*, in the performance of all MP options for both pilchard and anchovy as g_1 increases (confirming the results of Figures 6.4 and 6.5). It is also evident that the 'performance gap' between all options widens as g_1 increases. This means that the worst performers (BL and M3 for pilchard) deteriorate most as g_1 increases, while the deterioration for the best performers is not as marked.

For pilchard, performance in terms of $\bar{C}$ improves very slightly for estimator D1, but deteriorates markedly for estimator D2, when increasing g_1. When $g_1 = 2$, D1 also performs better than D2 (with $\sigma_g = 0$) in terms of both summary statistics for all MPs. This suggests that, with cycles of high amplitude, it may be better simply to use a running mean to track biomass trends rather than attempting some measure of the underlying position in the regime cycle. However, for smaller g_1 values, although D1 performs better than D2 in terms of $\bar{C}$ for all options (except M3), it performs worse than D2 in terms of *risk* when σ_g is small. This deterioration in *risk* for D1 compared with D2 (with small σ_g) would need to be judged against the feasibility and likely cost of obtaining relatively precise information about the underlying position in the regime cycle at any time, which D2 requires for small σ_g.

g_2 Changes

When comparing the plots in Figure 6.7a with those in 6.7b, there is a general shift of results towards the right and down for (b), indicating an overall improvement in performances when g_2 changes from 30 to 50 years. This is consistent with the results of Figures 6.4 and 6.5, which show a general improvement in performance for both pilchard and anchovy when $g_2 = 50$ rather than $g_2 = 30$.

NPV AND *LOSS*

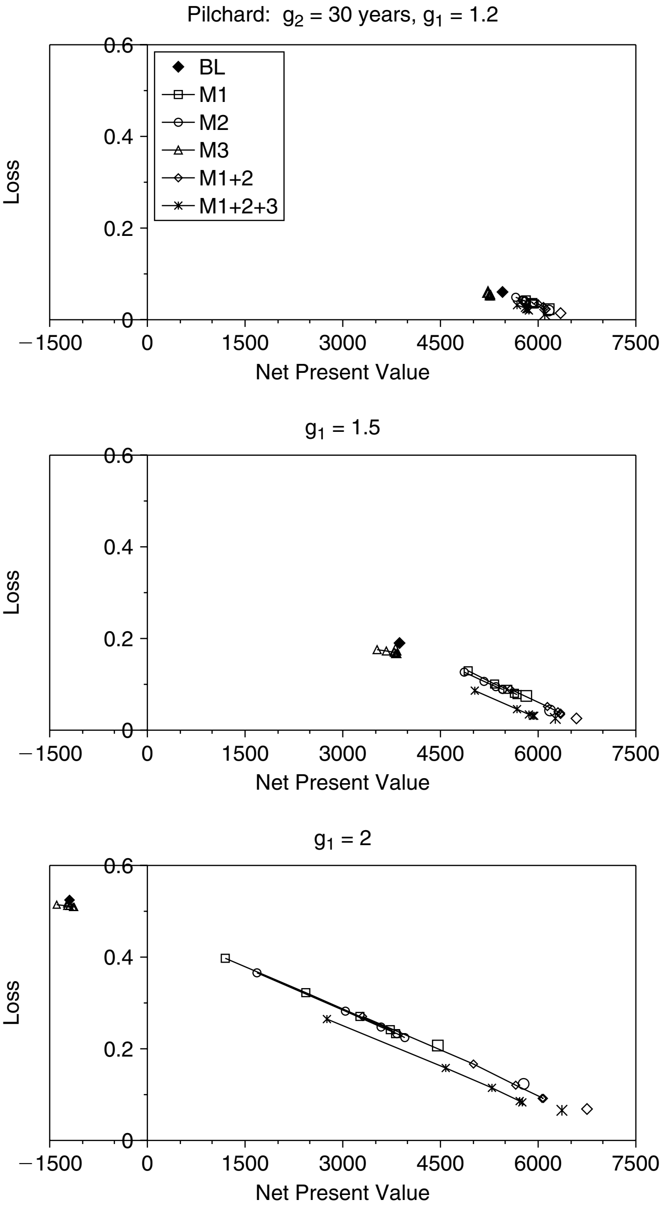
Figure 6.8 plots *Loss* against *NPV* to provide an economic equivalent to the catch-*risk* curves of Figure 6.7. The interpretation of 'better performance' is similar to that of Figure 6.7, namely MPs perform best when they lie towards the bottom-right corner of the plots. The general behaviour of MPs in Figure 6.8 is broadly similar to that in Figure 6.7 for pilchard, but quite different for anchovy. Although Figure 6.8 provides a broad summary of results, symbols are difficult to distinguish in some plots. Therefore, most of the discussion below will focus on separate plots for *NPV* (Figure 6.9) and *Loss* (Figure 6.10).

Pilchard

Figure 6.9 shows superior performance in terms of *NPV* for all MPs considered when estimator D1 is used rather than D2, whatever the value of g_1. This was also the case for the results in Figure 6.7 when considering *risk*, though there only for $g_1 = 2$. All MPs except M3 perform consistently better than BL, with BL and M3 showing negative *NPV*s for $g_1 = 2$. As before, MP performance deteriorates with increasing σ_g, this deterioration becoming worse as g_1 increases. Patterns shown in Figure 6.10 for *Loss* are generally the same (with higher *Loss* values indicating poorer performance in terms of this statistic). MP options M1 + 2 and M1 + 2 + 3 show the best performance overall in terms of both *NPV* and *Loss*. Comparing Figure 6.8a with Figure 6.8b, performance is generally better in the latter for all options, once again consistent with the results of Figure 6.4.

Anchovy

When considering anchovy in Figures 6.8–6.10, it should be noted that *NPV* and *Loss* are based not only on anchovy, but also pilchard by-catch (landed with anchovy – see Appendix 6.1). In Figures 6.9 and 6.10, there is a general deterioration for all MPs in terms of both *NPV* and *Loss*, as g_1 increases. M1 performs worst in terms of *NPV* (and *Loss* for $g_1 = 2$) for estimators D1 and D2 (with high σ_g), because anchovy catches are reduced when the pilchard cycle is in a trough, so reducing contributions to *NPV*.



Note: See Figure 6.8b.

Figure 6.8a Performance of MP options in terms of the summary performance statistics Loss *and* NPV

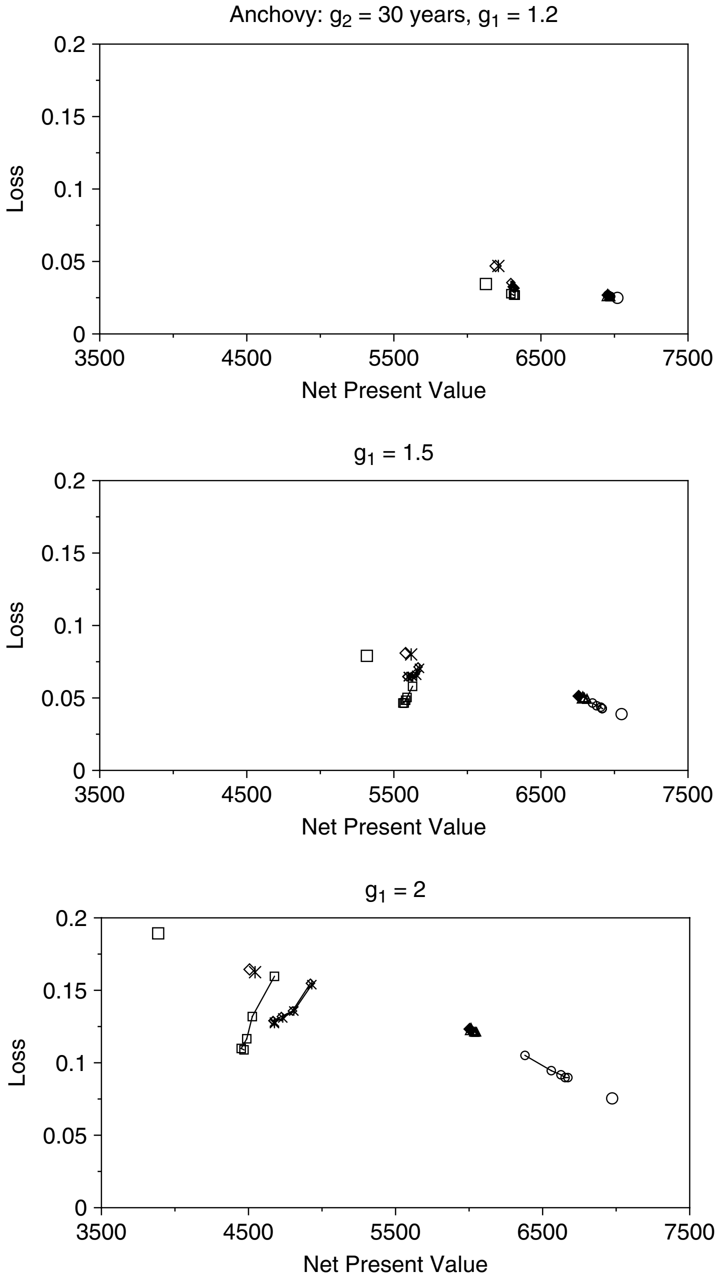
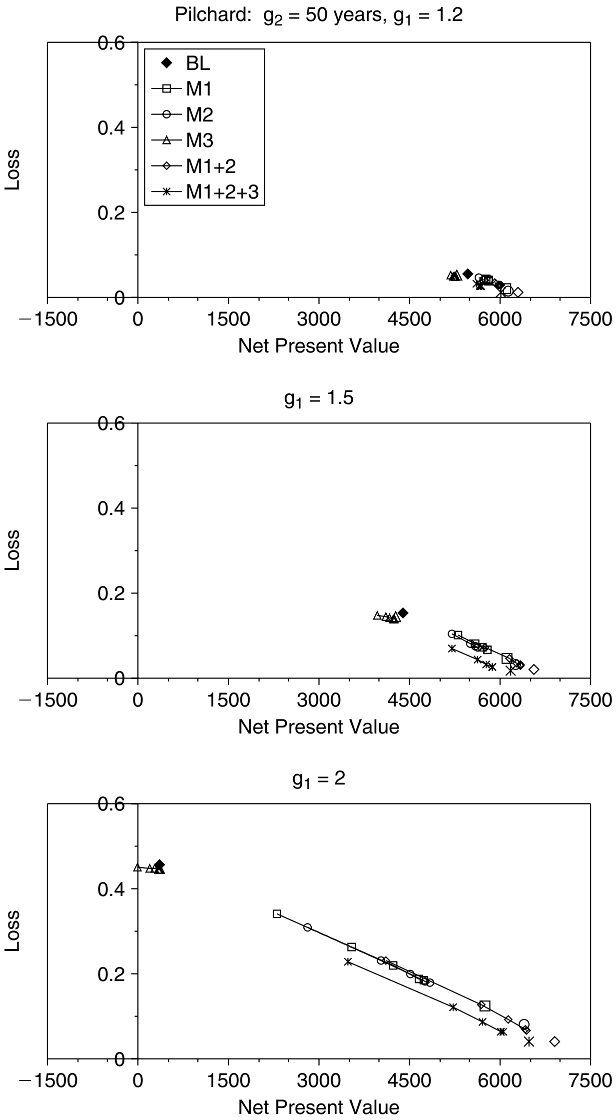


Figure 6.8a *(continued)*



Note: Description of the plots is as for Figure 6.7, except that for anchovy, *Loss* and *NPV* for M2 and M3 are not identical to BL, and these options are therefore now shown in the anchovy plots. Furthermore, in the anchovy plots, estimator D2 with $\sigma_g = 0°$ has the lowest *Loss* value, with *Loss* increasing as σ_g increases.

Figure 6.8b Performance of MP options in terms of the summary performance statistics Loss *and* NPV

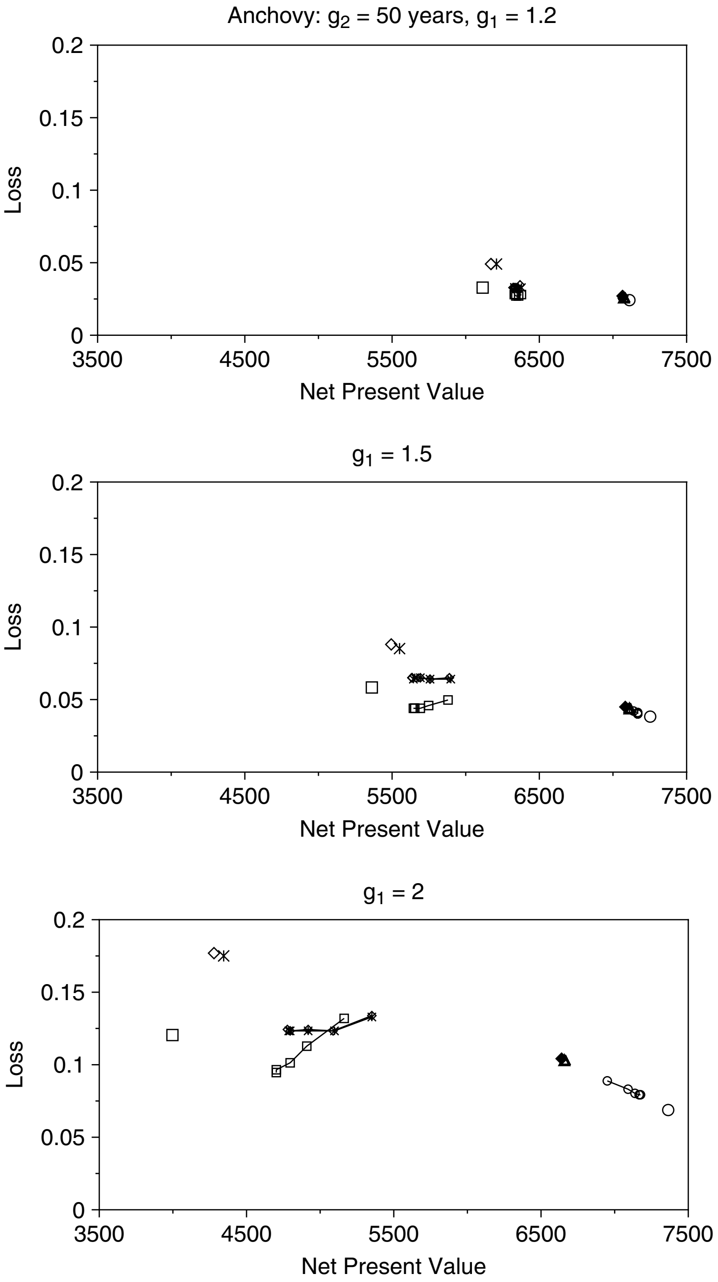
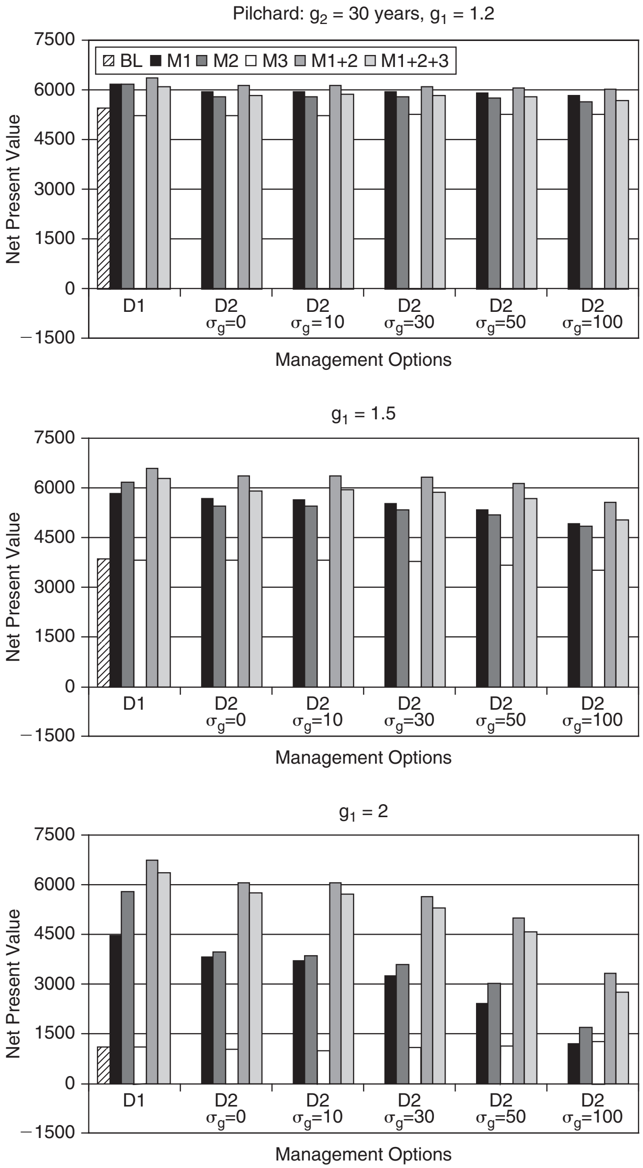


Figure 6.8b *(continued)*



Pilchard: g_2 = 30 years, g_1 = 1.2

g_1 = 1.5

g_1 = 2

Note: Differences between plots are as described for (a) in Figure 6.7.

Figure 6.9 Performance of MP options in terms of the summary performance statistic NPV

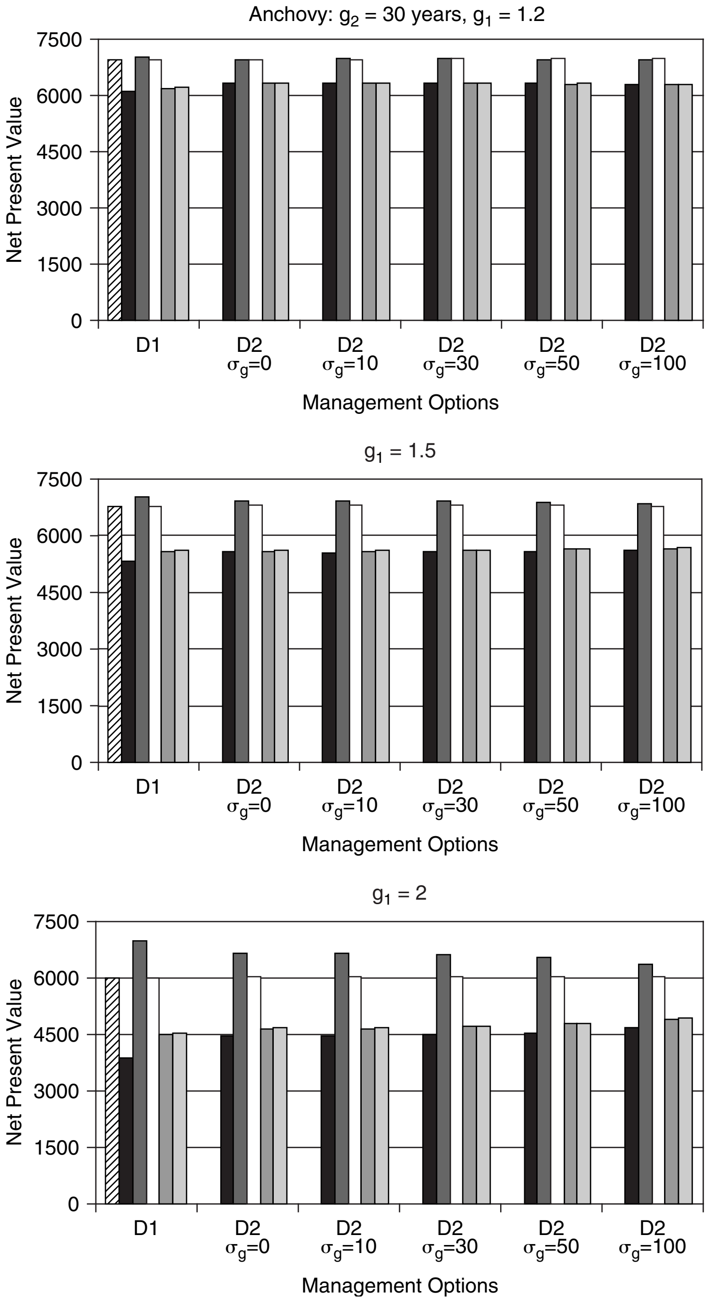
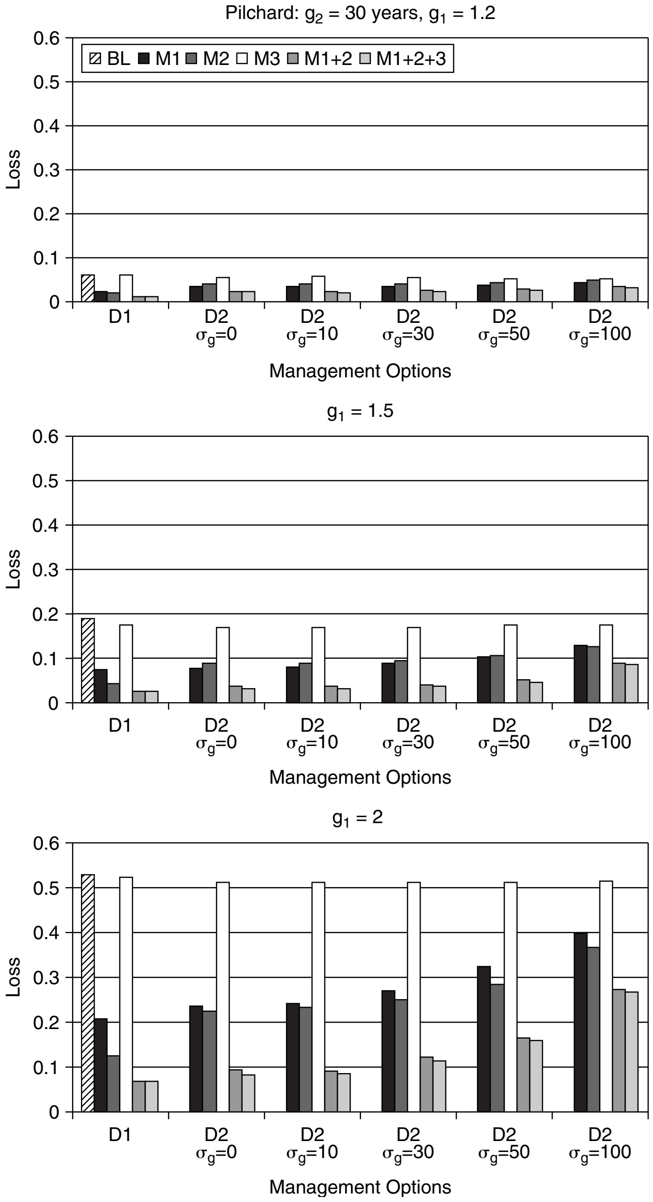


Figure 6.9 *(continued)*



Note: Differences between plots are as described for part (a) in Figure 6.7.

Figure 6.10 Performance of MP options in terms of the summary performance statistic Loss

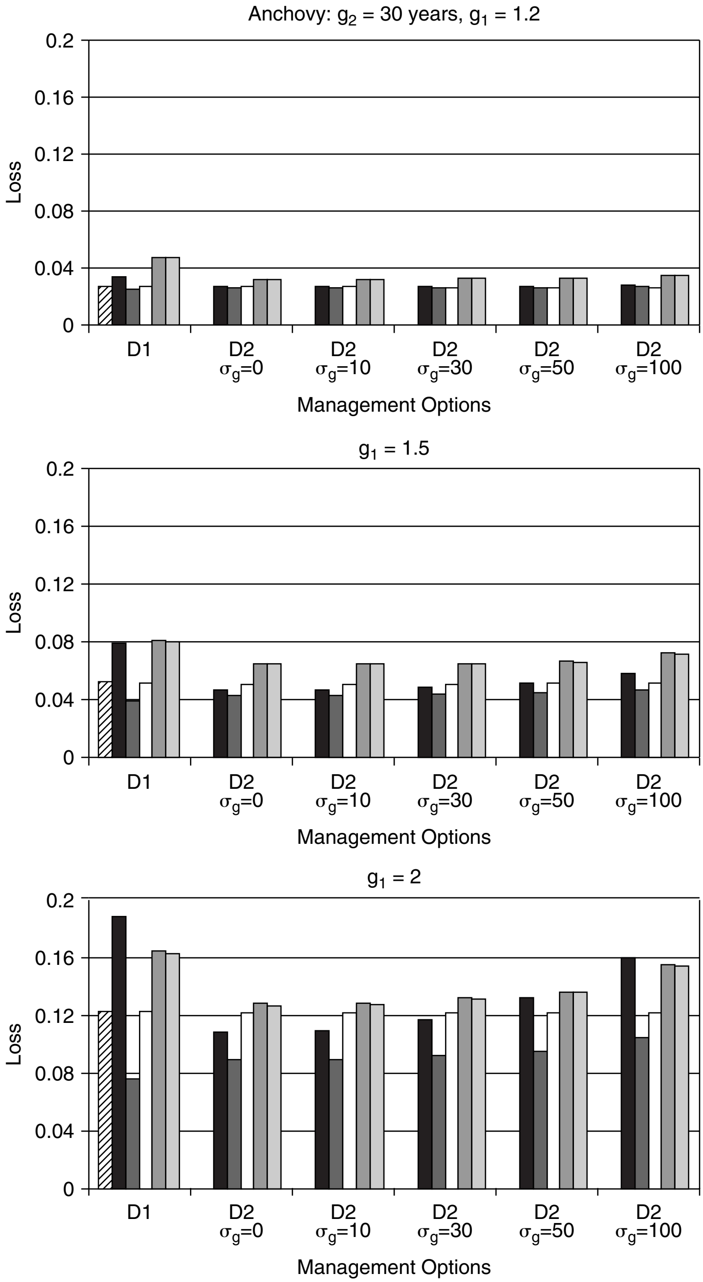


Figure 6.10 (continued)

Options M1 + 2 and M1 + 2 + 3 perform better than M1 in terms of *NPV* (and *Loss* for estimators D1 and D2, with high σ_g, when $g_1 = 2$). This is because, in addition to these options reducing anchovy catches, the M2 component of these options also reduces the initial by-catch ratio γ when pilchard is in a trough, so protecting the low pilchard resource from potentially damaging levels of initial pilchard TABs (pilchard troughs coincide with anchovy peaks under the operating models used, and therefore imply higher by-catch levels initially if γ is not reduced). Therefore, although M1 + 2 and M1 + 2 + 3 reduce pilchard by-catch while M1 does not, they lead (counter-intuitively) to higher *NPV* values (which result from pilchard by-catch and anchovy catches) than M1 because of the lower *risk* that arises as a consequence of the extra protection afforded to pilchard (see Figure 6.7). Thus, more pilchard (including by-catch) is available to be harvested.

In Figures 6.9 and 6.10, MP option M2 consistently performs the best in terms of both *NPV* and *Loss* for anchovy. This is because anchovy catches remain intact (they are not reduced for M2 and M3, unlike the other options). Both these anchovy catches and the additional pilchard by-catch that is gained (as a result of the extra protection accorded to the pilchard resource by not setting initial TABs that are too high) contribute towards increasing *NPV* and decreasing *Loss*. M3 is almost identical to BL for anchovy, because decreasing pilchard-directed catches when pilchard is in a trough has little impact on anchovy catches and pilchard by-catch (Figure 6.7).

Patterns are similar when g_2 is increased to 50 years (compare anchovy in Figures 6.8a and 6.8b), with a general improvement in performance across the board in terms of the two summary statistics shown, except for estimator D1 for MP options M1+2 and M1+2+3, for which both summary statistics deteriorate. This behaviour is counter-intuitive, because there is an improvement across the board (including for M1+2 and M1+2+3) for estimator D2. When comparing Figures 6.8a and 6.8b, the movement of points corresponding to estimator D1 in the opposite direction to those for D2 for these two MPs must therefore be a consequence of the estimator itself. This may imply that the efficiency of $\mu_{y,6}$ in following the biomass trend may change for different g_2 values (for example, it may become less efficient for smaller g_2 values), an aspect that is beyond the scope of this chapter.

Another interesting feature that arises from the comparison of Figures 6.8a and 6.8b for anchovy is the relatively large decrease in *Loss* for M1 using estimator D1, when compared with estimator D2 for the same MP, particularly for $g_1 = 2$ (one would expect the decrease to be to the level of D2 with $\sigma_g = 0$). The decrease itself is not surprising, because $g_2 = 30$ contains more troughs for anchovy than does $g_2 = 50$, and $\bar{C}$ is lower in the corresponding pilchard plots. However, the magnitude of the decrease may once again point to a change in efficiency of $\mu_{y,6}$ as g_2 is changed.

INTERANNUAL CATCH VARIABILITY

Figure 6.11 shows changes in V, the interannual catch variability, for all MP options with their associated D1/D2 estimator combinations. For pilchard, V is the interannual catch variability of the directed catch only.

Pilchard

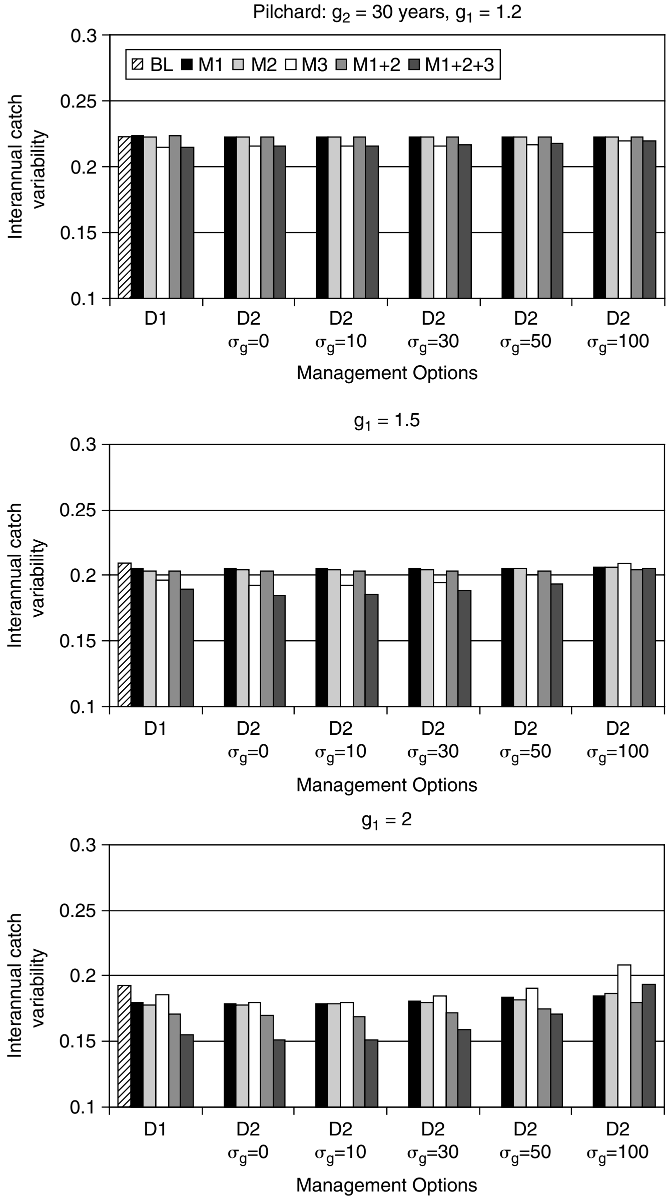
There is a general decrease in V for pilchard as g_1 increases, probably because the maximum and minimum TAC constraints (Table 6.1) come into play more frequently with increasing g_1, so limiting variations in catch. MP options M3 and M1 + 2 + 3 are the most sensitive to increases in σ_g, because M3 reduces the actual pilchard-directed catches themselves, and increases in σ_g lead to more variable adjustments to the directed catches (see Figure 6.3), and therefore higher V. All MP options provide improvements in V compared with BL, except for M3 under estimator D2 with $\sigma_g = 100$. Trends are similar when comparing Figures 6.11a and 6.11b, with (b) generally showing slightly lower V values.

Anchovy

For anchovy, the V values for M2 and M3 are insensitive to increases in g_1 and σ_g because the MP options do not reduce anchovy catches as part of the management action for pilchard troughs. The V values for M1, M1+2 and M1+2+3 are identical to one another for estimator D2 for the following reasons: the M2 and M3 components of these options do not affect anchovy catches directly; and estimator D2 tracks the actual position in the pilchard cycle and not the biomass, so is therefore not sensitive to the management actions affecting pilchard by-catch and directed catch in options M2 and M3.

This is not the case for estimator D1, however, because it tracks adult pilchard biomass, which is affected by reductions in both pilchard by-catch (M2) and directed catch (M3). V is generally lower for estimator D1 than for D2, particularly for option M1 and higher g_1 values. V increases slightly for estimator D2 when σ_g increases, for the same reason as with M3 and M1+2+3 for pilchard.

In Figure 6.11a for anchovy, when estimator D1 is used, the V value for M1 decreases relative to BL and the other MP options as g_1 increases. This



Note: Differences between plots, and parts (a) and (b) are described in Figure 6.7.

Figure 6.11a Performance of MP options in terms of summary performance statistic V

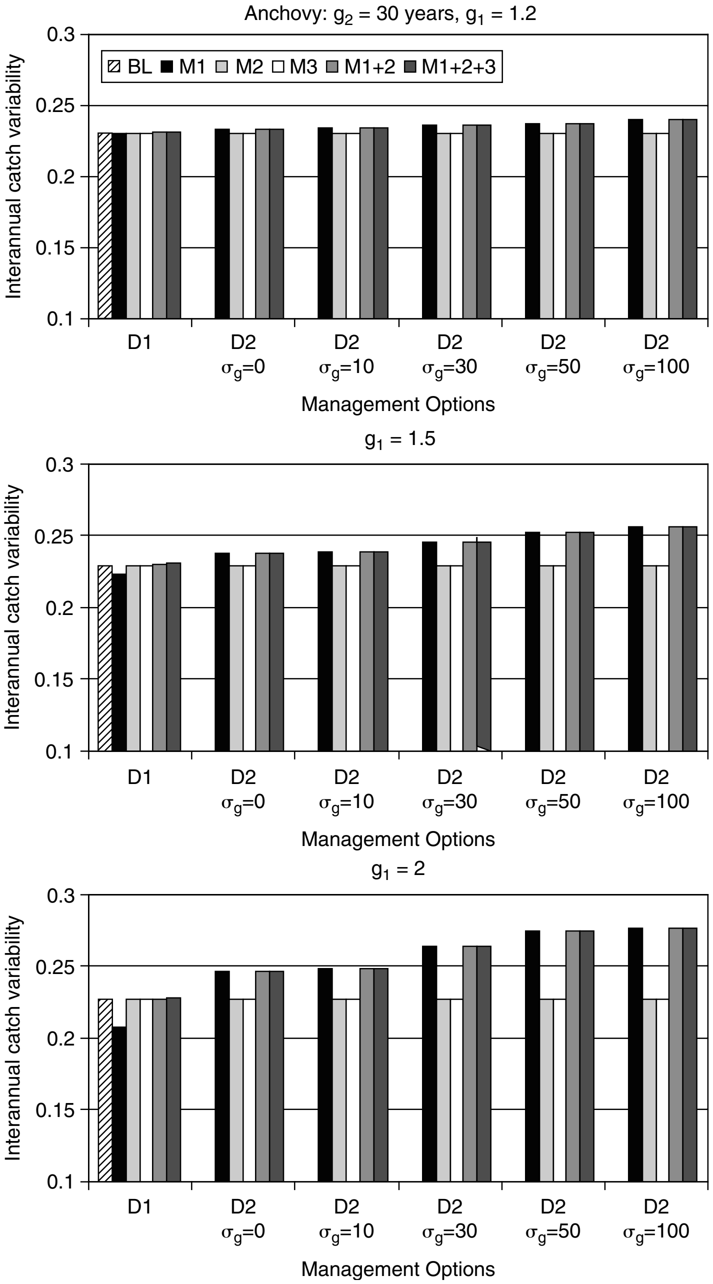
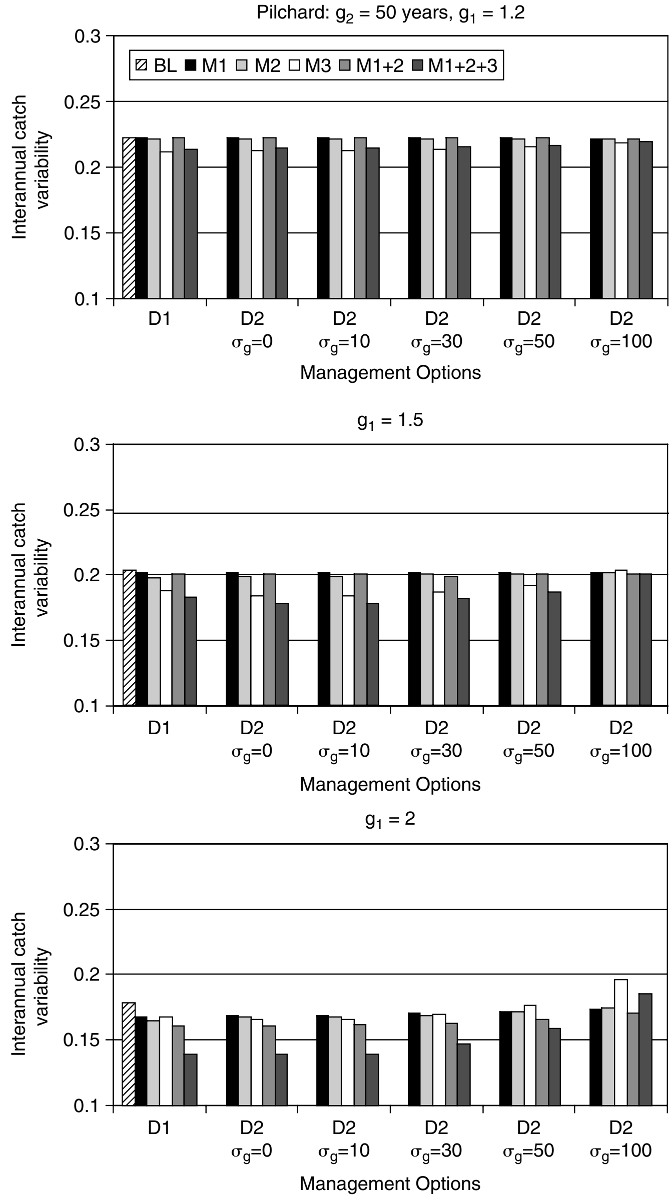


Figure 6.11a (continued)



Note: Differences between plots, and parts (a) and (b) are as described in Figure 6.7.

Figure 6.11b Performance of MP options in terms of the summary performance statistic V

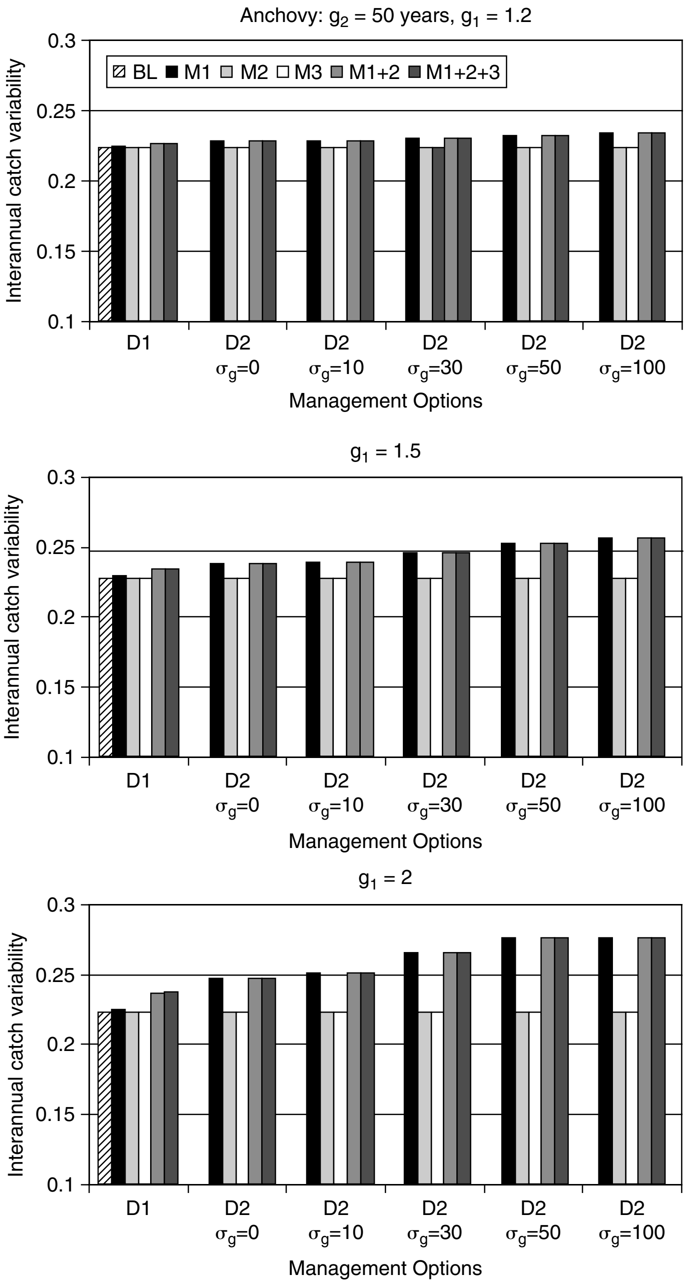


Figure 6.11b (continued)

pattern is not maintained, however, when D1 is used and g_2 changed from 30 to 50 years – in this case, *V* values for M1+2 and M1+2+3 increase relative to BL and the other MP options. There is no such change in pattern evident for D2 when changing g_2, which leads to the same conclusion as for Figures 6.8–6.10, namely that the optimal choice for the period x over which the running mean $\mu_{y,x}$ is taken may depend on the value of g_2.

REMOVING TAC CONSTRAINTS

All the results considered thus far are subject to the same TAC constraints as listed in Table 6.1, because although the control parameters are adjusted for the various MP options (equations 6.7–6.10), these adjusted parameters are used only in the equations in Table 6.1 (that is, not in the corresponding constraints in Table 6.1). Table 6.5 compares MP option M1+2 with an option that is identical except that there are no upper constraints on either pilchard or anchovy TAC (that is, the upper bounds of constraints in Table 6.1 for anchovy and pilchard, respectively, have been removed). This option

Table 6.5 MP option M1+2 compared with a related version, M1+2′

Management Option	Pilchard				Anchovy			
	$\bar{C}$ (1000 tonnes)	*risk*	*depl*	*V*	$\bar{C}$ (1000 tonnes)	*risk*	*depl*	*V*
No cycle ($g_1=1$)								
M1+2	133.6	0.358	0.560	0.230	140.5	0.342	0.856	0.231
M1+2′	141.1	0.358	0.565	0.281	118.7	0.238	0.877	0.231
$g_1=1.2$								
M1+2	134.6	0.402	0.593	0.224	138.1	0.340	0.826	0.231
M1+2′	143.4	0.402	0.582	0.280	118.8	0.252	0.848	0.232
$g_1=1.5$								
M1+2	135.8	0.470	0.640	0.203	132.2	0.350	0.793	0.230
M1+2′	149.6	0.470	0.608	0.280	115.3	0.278	0.814	0.233
$g_1=2$								
M1+2	135.8	0.628	0.713	0.171	122.2	0.382	0.755	0.227
M1+2′	161.9	0.628	0.640	0.289	110.2	0.306	0.773	0.233

Note: The two options are identical except that the latter has no upper bounds on either pilchard or anchovy TAC (that is, the upper bounds of the constraints in Table 6.1 for pilchard and anchovy, respectively, are omitted). Pilchard *risk* for M1+2′ is 'tuned' to the same value as for M1+2 by adjusting α in both the equations constraints in Table 6.1. The estimator used throughout is D1, with $g_1 > 1$ and $g_2 = 30$ years.

is labelled $M1+2'$, and a comparison is made for the scenarios with no regime cycle ($g_1 = 1$), and $g_1 = 1.2$, 1.5 and 2. For ease of comparison, $M1+2'$ is 'tuned' to the same pilchard *risk* level as $M1+2$ for each scenario considered (the note to Table 6.5 explains how this is achieved), and only estimator D1 is considered.

When comparing the two MP options, pilchard shows increasing gains in $\bar{C}$ for the same *risk* as g_1 increases, but at the cost of much higher V values and much lower anchovy $\bar{C}$ values. The trends in the V values for pilchard are consistent with an inference drawn from Figure 6.11, namely that a decreasing V with increasing g_1 is a result of the TAC constraints playing a more prominent role for $M1+2$ as g_1 increases. This is not the case for $M1+2'$, which removes the upper bounds on TAC. Whereas pilchard does show an improvement in terms of $\bar{C}$ for the same *risk*, there is no clear gain for anchovy, with lower $\bar{C}$ accompanied by lower *risk*.

DISCUSSION

If there were no operational interaction (that is, the pilchard by-catch) between the pilchard and anchovy fisheries, then given a certain set of TAC constraints, it appears as if there would be little (if any) gain in using management procedures markedly different from BL to take account of regime shifts. This is evident from the results for options M4 in Figure 6.6 and M3 in Figure 6.7. M4 tries to overcome underutilization of anchovy by increasing exploitation levels of anchovy when pilchard abundance allows, showing a bottom-left to top-right movement with increasing α' in the *risk* versus average catch plot, with the performance for BL lying very close to this trend-line. This implies no overall gain for anchovy, because gains in $\bar{C}$ are accompanied by increases in *risk* (although the objective of a less underutilized anchovy resource is achieved).

Moreover, M3 (see Figure 6.7) tries to improve upon the performance of BL by reducing exploitation of adult pilchard when its biomass is low, but with little success. Therefore, when changes are made to the exploitation of the adult biomass for each species, there is little gain for that species. This indicates that when each species is considered in isolation, by ignoring the operational interaction between them, BL already provides 'optimal' utilization. This is because BL uses a constant proportion strategy for both species (essentially tracking the status of the adult biomass of each species), with Exceptional Circumstances to buffer against 'freak' events when continued use of the normal MP decision rules might be detrimental to the resource (Table 6.1; De Oliveira, 2003). Therefore, the operational interaction between pilchard and anchovy aside, there appears to be little (if any) overall gain to be made by changing BL, even if one were given 'perfect' knowledge of the underlying position in the regime cycle at any time ($\sigma_g = 0$). However, the analyses in this chapter do not provide a comprehensive evaluation of this matter, which warrants further investigation, particularly in the light of the results of Table 6.5, which show possible gains when the TAC constraints of OMP99 are relaxed.

Nevertheless, gains are possible, particularly for pilchard, when focusing on the operational interactions between pilchard and anchovy. Reduction of juvenile pilchard by-catch (landed with anchovy) by adjusting the control parameters α and γ (options M1 and M2 and combinations thereof – see Table 6.1 and equations 6.7 and 6.8) offer the best performance for pilchard under regime cycles of varying amplitude for summary statistics $\bar{C}$, *risk*, *Loss* and *NPV*. Adding the adjustments of M3 to M1+2 (to form M1+2+3), leads to marginally lower *risk* and *Loss* (almost no improvement, in fact, for the latter) than M1+2, but for lower $\bar{C}$ and *NPV*, and therefore it shows no overall gain. (M3 reduces β when pilchard abundance is low, but does not focus on the actual operational interaction, that is by-catch – see Table 6.1 and equation 6.9.)

Effective management to take account of the operational interactions between pilchard and anchovy becomes more important as the amplitude of the regime cycles increases. This is shown in Figures 6.7 and 6.8 for pilchard by the increasing difference, as g_1 increases, in overall performance between M1 and M2 (and combinations thereof) on the one hand, and BL and M3 on the other. An interesting result is that, although *risk* and *Loss* deteriorate as g_1 increases, $\bar{C}$ and *NPV* are maintained at the same levels (and even increase) for M1+2 and M1+2+3 in combination with estimator D1, whereas *NPV* becomes negative and *Loss* deteriorates markedly for BL and M3. This indicates the importance of effective management under regime-cycle scenarios from an economic perspective. (It should be noted that the range $g_1 \leq 2$, as considered in this chapter, might not capture the full extent of possible variability of systems of small pelagic species. Further work may therefore need to consider larger g_1 values to deal with this concern.)

A surprising result is that, for pilchard, estimator D1 outperforms estimator D2 (with its additional information) in terms of *Loss*, *NPV* and $\bar{C}$ for MP options M1 and M2 and combinations thereof for all regime-cycle scenarios considered, and in terms of *risk* when σ_g is high, and when g_1 is high (then even for low σ_g). Furthermore, any gains in terms of *risk* for lower g_1 and σ_g values would need to be judged against the feasibility and likely cost of obtaining sufficiently precise information (a requirement of low σ_g for D2) on the underlying position in the cycle at any time. For anchovy, estimator D1 also outperforms D2 in terms of *Loss* and *NPV* for option M2 across all regime-cycle scenarios considered. D1 therefore appears to be a more effective basis on which to proceed than D2, probably because tracking the actual biomass with an appropriate running mean (which dilutes incorrect signals caused by measurement error) follows the 'genuine' highs and lows caused by both short- and long-term external factors (for example environmental forcing) more effectively than other attempts to follow the underlying position in the cycle, which do not take the shorter-term external factors into account.

If an option that performs better than BL for both pilchard and anchovy was being sought, then M2 seems the ideal candidate. Although it does not perform as well as M1+2 and M1+2+3 for pilchard, it performs best overall for anchovy, as is particularly noticeable in Figures 6.8–6.10 when considering *Loss* and *NPV*. M2 is effective for pilchard because of the extra protection afforded to juvenile pilchard as a result of setting lower initial by-catch ratios (γ is reduced) when pilchard abundance is low. Although the initial pilchard TAB will be lower, this option remains attractive because the revised pilchard TAB will still be realistic since it is based on the 'measured' ratios of juvenile pilchard to anchovy in the commercial catches and recruit survey (Table 6.1). M2 is also attractive for anchovy because it does not reduce anchovy catches, and therefore does not further underutilize anchovy. Moreover, the additional pilchard by-catch that becomes available as a result of the more efficient utilization of pilchard[4] contributes to a higher *NPV* and a lower *Loss* for anchovy. A concern for this option, however, is if the by-catch ratio remains high even when anchovy abundance is high and pilchard abundance low (contrary to experience so far – De Oliveira and Butterworth, 2004). Reducing γ would then amount to unrealistic expectations on the industry to keep initial pilchard by-catch ratios low, leading to illegal dumping of fish when these expectations cannot be met. Nevertheless, the circumstances of a high pilchard by-catch ratio given such abundances would seem unlikely.

Some aspects of the analyses in this chapter that warrant further investigation include the following:

- The efficiency of different choices for the period x of the running mean $\mu_{y,x}$ for estimator D1 over a range of periods (g_2 values) for regime cycles.
- The effect of changing the TAC constraints for both species (these remained unchanged for most of the analyses of this chapter).
- The consequences of not knowing the phase difference between pilchard and anchovy (set at 180° in the existing model), or of a phase difference that is variable.
- Considering results for a wider range of regime cycle amplitudes (that is $g_1 \geq 2$).

- Removing the end effects of different choices for the regime cycle period by randomizing over the start of the cycle to average out these effects, or projecting over much longer periods than 100 years to reduce the size of these effects.
- Considering a wider range of options for modelling regime shifts than sinusoid curves.
- Developing more dynamic and refined versions of the economic component, with different assumptions about the effort function (Appendix 6.1).
- Broadening the study to incorporate other upwelling systems (this chapter focuses on the South African Benguela system only).
- Different management strategies, such as constant catch and constant effort (only the constant proportion approach was considered in this chapter).

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APPENDIX 6.1: ECONOMIC SUB-MODEL USED TO CALCULATE *NPV* AND *LOSS*

The economic factors used to calculate *NPV* and *Loss* amount to a simple model, based on the formulations in BEAM 4 (Sparre and Willmann, 1993), which calculates fixed costs linked to the number of vessels in the purse-seine fleet for pilchard and anchovy, and variable costs linked to effort and landed catch. The purse-seine fleet may be divided into four categories, each with different requirements for the species targeted, the amount of fuel and ice used, the number of crew employed and the cost of vessels.

Category 1

Vessels in this category are shorter than 19 m and carry ice, concentrating on catching adult pilchard for bait.

Category 2

Vessels are in the length range 19–26 m and do not carry ice. They target anchovy for fishmeal. Any by-catch of pilchard with anchovy is also turned into fishmeal.

Category 3

Vessels are in the length range 19–26 m and have capacity to carry ice. They target anchovy for fishmeal (for which ice is not necessary) and adult pilchard for canning, using ice for the latter to maintain fish freshness. Any by-catch of pilchard with anchovy is converted to fishmeal.

Category 4

Vessels are 27 m or more long. They have refrigeration facilities on board, which require an initial amount of ice when used. They target anchovy for fishmeal (for which the refrigeration facility is not used) and adult pilchard for canning, with the latter needing to be cooled to maintain freshness. Any by-catch of pilchard with anchovy is turned into fishmeal.

The parameters used in the economic component and their actual values are given in Table A6.1.

Fixed Costs

It is assumed here that the depreciation costs of vessels in categories 3 and 4 are shared equally between the pilchard and anchovy fisheries, because vessels in these categories target both pilchard and anchovy.

$$\text{Pilchard:} \quad \varphi^{P}_{fix} = \tfrac{1}{2}\delta_{depr}(p_{ves,3}u_{ves,3} + p_{ves,4}u_{ves,4}) \; + \; \delta_{depr}p_{ves,1}u_{ves,1} \qquad (6.\text{A}1)$$

$$\text{Anchovy:} \quad \varphi^{A}_{fix} = \tfrac{1}{2}\delta_{depr}(p_{ves,3}u_{ves,3} + p_{ves,4}u_{ves,4}) \; + \; \delta_{depr}p_{ves,2}u_{ves,2} \qquad (6.\text{A}2)$$

Variable Costs

Ice (per trip-hour)

Ice is used only when targeting pilchard. It is loaded on to the vessel once, when it sets out to sea, so the average number of trip-hours per trip is required in this calculation.

$$\text{Pilchard:} \quad \varphi^{P}_{var,ice} = \sum_{j=1,3,4} (q^{P}_{j} u_{ice,j} / t^{P}_{j})\, p_{ice} \qquad (6.\text{A}3)$$

Table A6.1 *A description of the parameters used in deriving the economic summary performance statistics* Loss *and* NPV, *with actual values used*

Parameters	Description	Values			
By category		cat 1	cat 2	cat 3	cat 4
$p_{ves,j}$	Cost of a vessel in category *j* (thousand Rands per vessel)	2400	4500	4500	8000
$P^P_{crew,\,j}$	Earnings of crew in category *j* (thousand Rands per thousand tons landed) when targeting pilchard	25		10	10
$P^A_{crew,\,j}$	Earnings of crew in category *j* (thousand Rands per thousand tons landed) when targeting anchovy		8	8	8
$u_{ves,j}$	Number of vessels in category *j*	15	9	48	12
$u_{ice,j}$	Amount of ice used per trip in category *j* (tons per trip)	5		55	10
$u_{fuel,j}$, j=1,2,3 $u_{fuel,4a}$ $u_{fuel,4b}$	Amount of fuel used per trip-hour by vessels in category *j* (litres per trip-hour) – for category 4, 240 is used when targeting anchovy and an additional 55 (for refrigeration) when targeting pilchard	25	118	118	240 and 295 resp.
$u_{crew,j}$	Number of crew members per vessel in category *j*	8	9	9	10
t^P_j	Average number of trip-hours per trip by a vessel in category *j* when targeting pilchard	10		10	18
q^P_j	Proportion of the total catch of adult pilchard taken by category *j* vessels each year	0.08		0.43	0.49
q^A_j	Proportion of the total catch of anchovy taken by category *j* vessels each year		0.16	0.59	0.25

Parameters	Description	Values			
For all categories		cat 1	cat 2	cat 3	cat 4
δ_{depr}	Depreciation of capital (per year) – here, $1/\delta_{depr}$ gives the number of years to replace a vessel relative to when it was purchased	0.05			
p_{ice}	Cost of ice (thousand Rands per ton)	0.13			
p_{fuel}	Cost of fuel (thousand Rands per litre)	0.002			
p^P, p^A	Wholesale value of pilchard and anchovy landed (thousand Rands per thousand tons)	3100 and 600 respectively			
$C^P{}_{break,} C^A{}_{break}$	'Break-even' points for landed catch in terms of profit for pilchard and anchovy (in thousand tonnes) (Note, the break-even point for anchovy includes pilchard by-catch landed with anchovy.)	60 and 100 respectively			

Notes:
1. For the purpose of this chapter these values are only 'ball-park' figures. All parameters are year-invariant. Where appropriate, units are given in parentheses under 'Description'.
2. The average interbank exchange rate for 1999 was 1 USD = 6.12 Rands.

Source: These data were obtained in 1999 from the South African pelagic industry and Marine & Coastal Management (MCM) databases by Jan van der Westhuizen and Johan de Goede, both MCM, Cape Town, South Africa.

Fuel (per trip-hour)

Pilchard fishing requires extra fuel in category 4 because of the refrigeration capacity of vessels in this category.

$$\text{Pilchard:}\quad \varphi^{P}_{var,fuel} = [q^{P}_{1}u_{fuel,1} + q^{P}_{3}u_{fuel,3} + q^{P}_{4}(u_{fuel,4a} + u_{fuel,4b})]p_{fuel} \tag{6.A4}$$

$$\text{Anchovy:}\quad \varphi^{A}_{var,fuel} = [q^{A}_{2}u_{fuel,2} + q^{A}_{3}u_{fuel,3} + q^{A}_{4}u_{fuel,4a}]p_{fuel} \tag{6.A5}$$

Crew (per ton landed)

For this chapter, it is assumed that opportunity costs of labour are exactly the earnings of the crew manning the vessels, because we focus on the 'microcosm' of the fleet, without taking 'outside' effects into account. This makes the implicit assumption that the fleet and costs associated with it are not influenced by outside effects, which is unrealistic, but regarded as a first step in taking economic effects into account.

$$\text{Pilchard:}\quad \varphi^{P}_{var,crew} = \sum_{j=1,3,4} q^{P}_{j}u_{crew,j}p^{P}_{crew,j} \tag{6.A6}$$

$$\text{Anchovy:}\quad \varphi^{A}_{var,crew} = \sum_{j=2,3,4} q^{A}_{j}u_{crew,j}p^{A}_{crew,j} \tag{6.A7}$$

Profit

Profit is the difference between the price achieved for the landed catch, and the fixed and variable costs of the fleet. (Note that the operating model assumes that, unless the TAC/B exceeds the natural limitation of what can be caught, the whole TAC/B will be taken – that is, no under- or over-catching of the TAC/B. This assumption applied to OMP99 (De Oliveira, 2003); in contrast, OMP02 (De Oliveira and Butterworth, 2004) does not assume that the anchovy TAC and pilchard TAB is always taken.)

$$\text{Pilchard:}\quad \pi^{P}_{y} = p^{P}TAC^{P}_{y} - \varphi^{P}_{fix} - (\varphi^{P}_{var,ice} + \varphi^{P}_{var,fuel})E^{P}_{y} - \varphi^{P}_{var,crew}TAC^{P}_{y} \tag{6.A8}$$

$$\text{Anchovy:}\quad \pi^{A}_{y} = p^{A}(TAC^{2,A}_{y} + TAB^{2,P}_{y}) - \varphi^{A}_{fix} - \varphi^{A}_{var,fuel}E^{A}_{y} - \varphi^{A}_{var,crew}(TAC^{2,A}_{y} + TAB^{2,P}_{y}) \tag{6.A9}$$

where E^{i}_{y} is the effort (in trip-hours) used when targeting species i in year y, and the TAC/Bs are from the MP being tested.

Calculating Effort

Catch rate per unit effort (cpue) is not a good index of abundance for pilchard and anchovy because of the shoaling behaviour of these fish. This is because cpue may remain constant even when abundance drops, because of the effect of 'mining out' shoals. This feature of cpue implies that effort increases and decreases with catch. It is therefore assumed that effort is proportional to the TACs set for directed catches. It is further assumed that there is a level of catch for directed pilchard, and for anchovy that includes the pilchard by-catch landed with anchovy (C^P_{break} for pilchard and C^A_{break} for anchovy – see Table A6.1), below which that fishery will operate at a loss. By setting $TAC^P_y = C^P_{break}$, $E^P_y = E^P_{break}$ and $\pi^P_y = 0$ in equation 6.A8, and $TAC^{2,A}_y + TAB^{2,P}_y = C^A_{break}$, $E^A_y = E^A_{break}$ and $\pi^A_y = 0$ in equation 6.A9, these equations can be solved for E^P_{break} and E^A_{break} respectively. Effort in year y can then be calculated as follows:

$$\text{Pilchard:} \quad E^P_y = \frac{E^A_{break} TAC^P_y}{C^P_{break}} \tag{6.A10}$$

$$\text{Anchovy:} \quad E^A_y = \frac{E^A_{break}(TAC^{2,A}_y + TAB^{2,P}_y)}{C^A_{break}} \tag{6.A11}$$

Calculating *NPV* and *Loss*

The Net Present Value, NPV, for species i is calculated as follows:

$$NPV\,(\text{for species } i) = \frac{1}{100}\sum_{k=1}^{100}\left(\frac{1}{1+\delta_{disc}}\right)^k \pi^i_{1997+k} \tag{6.A12}$$

where δ_{disc} is the discount rate for the pelagic fishery. (Note that if more than one simulation of a 100-year trajectory is used, as is the case in this chapter, then NPV is also averaged over the number of simulations.) For this chapter, a value of $\delta_{disc} = 0$ has been used under the assumption that national objectives override those of individual companies, and that those objectives involve managing for the very long term. *Loss* is simply the number of years (out of the total number in the projection period, that is, 100 years) where the contribution to NPV for that year in equation 6.A12 is either zero or negative (*Loss* is also averaged over the number of simulations).

NOTES

1. Note for simplicity of explanation, the argument of sine is reflected in degrees rather than radians.
2. The projection used is 100 years, in contrast to the 20 used to evaluate OMP99, and a constraint has been placed on r_y (Table 6.1). This chapter is based on OMP99 because the work was completed before later versions (for example OMP02 in De Oliveira and Butterworth, 2004) had been developed and fully evaluated.
3. For the illustrative purposes of this plot, the operating model value $B_{y,N}$ replaces the corresponding MP value $B_{y,Nov}$ (that is, the simulated survey value passed to the MP) in equation (6.4).
4. The extra protection of juveniles offered by M2 leads to lower by-catch during years of low pilchard abundance with lower *risk* as a result, and higher catches in the long run.

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7. Declines in Namibia's pilchard catch: the reasons and consequences

Ussif Rashid Sumaila and Kevin Stephanus

INTRODUCTION

The objectives of this chapter are fourfold. First, we give an account of the collapses in the catches of Namibian pilchard over the last 40 to 50 years. Second, we present and discuss possible reasons for the declines. Third, we state some of the consequences of the collapses of pilchard in Namibia. Finally, we explore the question of whether the observed collapses in pilchard stocks can be considered 'economic-overfishing', that is, fishing that does not maximize the discounted economic rent from the fishery.

Pilchards live in temperate waters from southern Angola to KwaZulu-Natal in South Africa, generally within the 100 m isobaths and often close inshore. The degree of mixing of the southern (mainly off South Africa) and northern (north of Lüderitz, Namibia) populations of pilchard is unknown. However, given that the populations spawn in different, widely separated areas, and are separated by a large perennial area of cold, upwelled water off Lüderitz, mixing of the two populations is probably not significant for management purposes, except in anomalous years when the upwelling 'barrier' apparently breaks down. Zooplankton constitutes an important portion of the diet of juvenile pilchard in particular, though adult fish rely more on phytoplankton (Boyer and Hampton, 2001; Shannon *et al.*, 2004).

LANDINGS AND BIOMASS

Annual catches rose rapidly from about 200 000 tonnes in the 1950s to a maximum reported catch of 1.4 million tonnes in 1968. There was then a sharp decline to less than 300 000 tonnes in 1971, followed by a slight increase until 1977 and 1978, when catches collapsed again. Since then annual catches have rarely exceeded 50 000 tonnes, except in the early 1990s, when they rose briefly to 100 000 tonnes. From 1996, annual landings have varied between 0 and 25 000 tonnes (Figure 7.1).

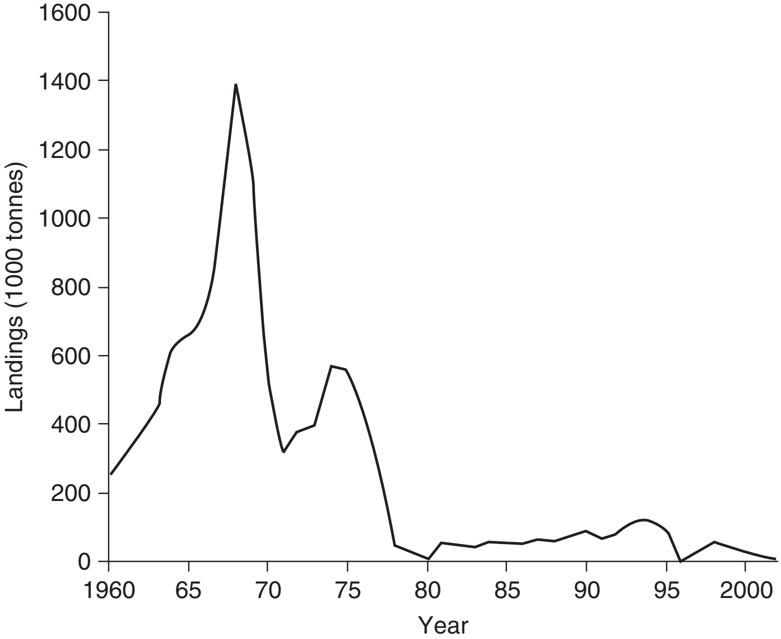

Figure 7.1 Pilchard landings from 1960 to 2002

Not surprisingly, pilchard biomass in Namibia followed similar trends as the landings (Figure 7.2). Estimates are that the stock collapsed from more than 11 million tonnes in the 1960s to less than 1 million tonnes by the mid 1970s (Butterworth, 1983). Acoustic survey estimates of pilchard on the Namibian/southern Angolan shelf since 1990 indicate that the adult stock is still very small, and was at an all-time low of just a few thousand tonnes in the summer of 1995/96 (Boyer and Hampton, 2001). Scientists at the Ministry of Fisheries and Marine Resources, Namibia (MFMR) estimated that, on the basis of surveys, the total pilchard biomass was about 180 000 tonnes in April 2000, 90 000 tonnes in March 2001 and just 40 000 tonnes in March 2002. Of the last figure, only 5000 tonnes were adult fish, the rest juveniles too young to spawn.

REASONS FOR THE DECLINE IN NAMIBIAN PILCHARD

According to Butterworth (1980), the reason for the collapses in 1971 and 1977/78 was overexploitation, coupled with considerable under-reporting

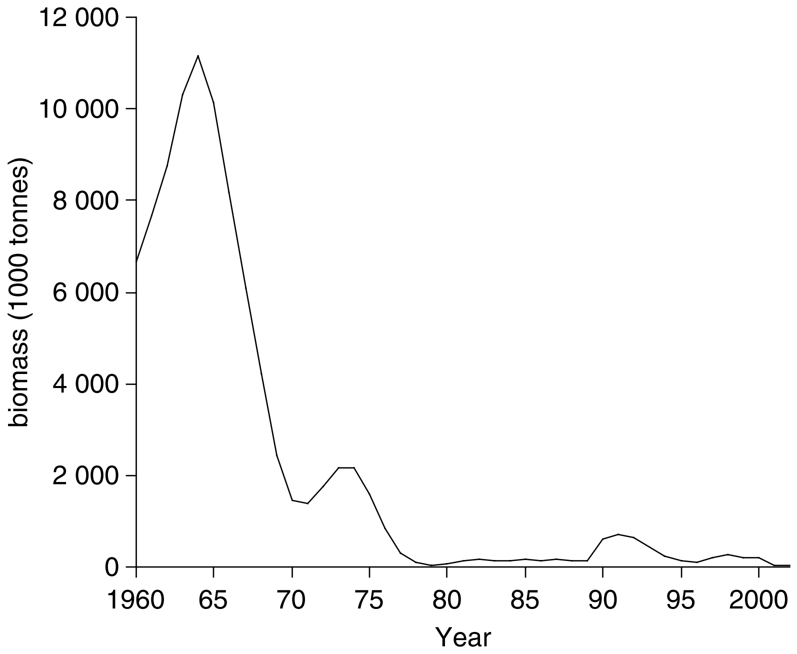

Figure 7.2 Pilchard biomass profile from 1960 to 2002

that resulted in the official catch of 1.4 million tonnes probably actually being in excess of 2 million tonnes. Further, dumping of fish unwanted by the canneries and fishmeal plants was rampant during that period. Boyer and Hampton (2001) recently supported Butterworth's views by stating that the collapse of Namibian pilchard was 'largely attributable to overfishing'. In particular, growth overfishing may have been a contributing factor, because a change from pilchard nets to anchovy nets, whose effective mesh size is smaller, was made in the 1960s (Boyer and Hampton, 2001).

Overfishing is not, however, the only reason advanced for the collapse of the pilchard stock off Namibia; adverse environmental conditions have also been implicated in the decline (Boyer *et al.*, 2001). The biomass of pilchard in both the northern and southern Benguela declined sharply following a system-wide Benguela *Niño* in 1963, which in the northern Benguela caused the fish to be concentrated close to Walvis Bay, where fishing pressure was high.

Further, the more recent decline in the biomass of Namibian pilchard, which started in 1993, was largely the result of the advection of low-oxygen water down from Angola in 1993 and 1994, aggravated by a major Benguela *Niño* in 1995. In March 1995, the entire shelf from Cabinda to central Namibia was covered by anomalously warm water of up to 8°C (Gammelsrod *et al.*, 1998). During this Benguela *Niño*, the stock shifted 4–5 degrees of latitude south, forced there by the advancing warm water, resulting in increased mortality and poor recruitment of pilchard and other species. However, despite these apparently adverse conditions, there was an increase in the availability of pilchard to the Walvis Bay fishing fleet, caused by a southward displacement of pilchard from northern Namibia and Angola, bringing them closer to the fish factories.

Collapse in the 1970s and late 1990s: Overfishing versus Adverse Environmental Conditions

Different weights have been given to overfishing and adverse environmental conditions as causes of the collapse, depending on when the biomass actually declined. The collapse of the 1970s is more often linked to overfishing (Butterworth, 1980, 1983; Boyer and Hampton, 2001), whereas that of the late 1990s is attributed more to adverse environmental conditions. The BENEFIT Stock Assessment Workshop report of November 2001, which focused on Namibian pilchard, intimates that abundance of pilchard had climbed to about 750 000 tonnes by 1992, but then declined in the mid-1990s, when there was a clear environmental anomaly off Namibia which also adversely impacted on other fish stocks, such as hake. There was also a major mortality of seals at the time, particularly pups. Even though the impact of poor recruitment (or poor environmental conditions) on the declines in catches has not been quantified, it is likely that negative environmental factors played a larger role in the 1990s than in the 1970s, when overharvesting was clearly the main cause (Doug Butterworth personal communication).

A review of the relevant literature reveals that concluding that overfishing was the main cause of the collapse in the 1970s makes economic sense because of the open access nature of the fishery then. Further, current literature does not adequately address how overfishing and adverse environmental conditions conspired together to deplete Namibian pilchard stocks. However, Cram (1981) did state that there was a protracted but less intense Benguela *Niño* between 1972 and 1974, the effect of which was probably aggravated by overfishing. Others argue that overfishing generally tends to cause collapses when the population goes through a phase of lower productivity (Manuel Barange personal communication).

THE CONSEQUENCES OF THE COLLAPSES

Biomass and Catch

The impacts of the collapses are vividly illustrated in Figures 7.1 and 7.2, with the ultimate consequence that a zero Total Allowable Catch (TAC) on pilchard was allocated for the 2002/03 fishing season! Catch patterns show a northward shift in the core distribution of the stock after the collapse in the 1970s, which may be due to the depletion of the southern spawning population and the cessation of associated migrations.

Historically, Namibian pilchard have spawned within 60 km of the coast off Walvis Bay and farther north, in the mixing of the zone south of the confluence of the Benguela and the Angola current systems. However, since the collapse of the stock in the 1970s, spawning in the south has diminished (Crawford *et al.*, 1987), and the migration of mature fish is believed to have decreased.

Fishers

Though there was a severe impact on employment in the canneries (the most labour-intensive sector), this was ameliorated to some extent by an ever-increasing fraction of the catch being canned: changes in the fleet included a move towards refrigeration, so that fish caught further from Walvis Bay were still suitable for canning.

Some labour was redeployed to other fishing sectors (for example trawling), which were going through a mini-boom at the time, but these sectors were not so labour-intensive. The vast majority of fishers were seasonal workers, and these people were simply dispensed with as workers. They had little choice but to return inland and north, where they tried to eke out an existence as subsistence farmers.

The Fleet

Most of the companies involved in the Namibian pilchard industry were multinational, so shifting vessels (and often crews) was quite straightforward. Some of the fleet moved north to northwest Africa in search of fish; others moved to the lucrative South American fishing grounds. Some of the fleet of small, predominantly wooden-hulled vessels were replaced by larger steel-hulled vessels that used refrigerated seawater to cool the catch and were thus capable of bringing the fish from northern Namibia to Walvis Bay still in a condition suitable for canning.

The number of vessels dropped from about 120 to 90 between 1969 and 1971, but some of this reduction may reflect the departure of the offshore factory ships (two former whaling factory vessels) with their attendant purse-seiners. Following the second collapse, there was a further reduction from 90 to 50 vessels over the period 1977–1980, although interestingly the total fleet hold capacity stayed fairly constant throughout the late 1960s to the early 1980s. Clearly, therefore, older, smaller vessels (which may have ended up being scrapped) were being replaced by newer, larger ones (see Figure 7.3b in Butterworth (1983) for the trends described herein).

The pelagic purse-seine fleet of Namibia declined from between 35–49 vessels in the 1980s and early 1990s to fewer than 15 vessels by 2000, concomitant with the decline in the TAC. Even with this decline, the remaining fleet is active only for a few days per year, catching only 10 tonnes per gross registered tonnage (GRT) each year compared with 90 tonnes per GRT in the 1970s, suggesting a huge overcapacity.

The Processing Sector

The consequence of the collapse on the processing sector, based mainly on anecdotes communicated by Doug Butterworth, is that a major portion of the land-based processing machinery worldwide moved from California (collapse in 1950s) to South Africa (collapse mid-1960s), to Namibia (collapse late 1960s), and then to South America (notably Chile) after that.

Two pilchard-canning factories, owned by Etosha Fishing and United Fishing Industries, operated in Namibia in 2004. Several options were considered by the Namibian government to minimize the potential impact of cannery operation variability. Although operations at the factories were brought to a standstill with the closure of the fishery, no full-time equivalent jobs were lost. Since much of the employment was of a seasonal nature at the factories, labour was just not used during the time of closure, and the authorities did not view this as loss of employment. However, no study has been carried out to understand how the seasonal workers managed during the time they were not working.

The factory owned by Etosha Fishing sometimes operated by re-canning pilchard from the previous year's catch, employing some of the seasonal workers for the purpose. There were some problems with the cans, and the company could not easily obtain certification and clearance from the South African Bureau of Standards, which regulates food (including fish product) quality. The other factory was closed down completely. However, very recently both factories have been operating because a TAC of 20 000 tonnes was allocated for the 2003/04 fishing season, which was increased to 25 000 tonnes for the 2004/05 season.

On the National Economy

At the time of the 1970s collapse, South Africa and the then South West Africa (now Namibia) were tightly intertwined, with the major fishing companies nearly all having a major South African investment, if not outright ownership. It is therefore impossible to talk about the impact of this collapse on the Namibian economy as such, because Namibia did not have a separate economy, being a contributor to the South African economy. Because of this situation, it may well be that the effects of the collapse were partly buffered by South Africa.

Overall, the other sectors cushioned the collapse of the pilchard fishery, while the decrease in the value of the Rand/Namibian dollar throughout the 1990s further lessened the blow, as income from other sectors orientated towards exports outside the Rand monetary area increased.

It appears that no serious impacts of this recent collapse on the Namibian economy were felt, because the early 1990s were when the more valuable local hake fishery was expanding and so probably compensating for the decline in pilchard catches.

ECONOMIC ANALYSIS OF THE HISTORIC LANDINGS PROFILE OF NAMIBIAN HAKE

Has Namibian pilchard been biologically and economically overfished? From Figures 7.1 and 7.2, it is clear that this stock has, during the period considered, declined drastically. However, it is not obvious that the pilchard stock has been overfished economically. To explore this issue, we address the question: would a sole-owner rational economic agent have chosen the observed catch pattern over a more sustainable catch profile? If the answer is yes, then the landings and biomass profile cannot be described as economic overfishing. To address this, we value the discounted landed value of the actual harvest profile and compare it with the discounted value of an assumed average sustainable annual landing of 400 000 tonnes. For pelagic stocks such as pilchard, landed values provide similar qualitative results as economic rent because the cost of fishing can realistically be assumed to be independent of stock size.

Price and landings data for this analysis were obtained from the MFMR. An average sustainable annual landing profile of 400 000 tonnes was analysed. This number is based on work carried out by Sumaila and Vasconcellos (2000) and Steinshamn *et al.*, (2004), along with the fact that the landings of pilchard did not start declining until they had peaked at well above 400 000 tonnes (Figure 7.1). It is important to stress that this

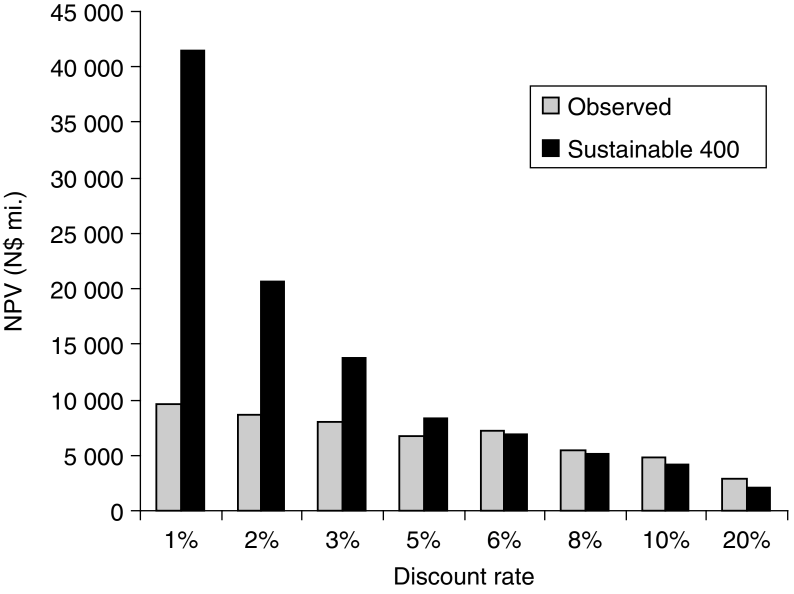

Figure 7.3 Total discounted present values from the observed and average sustainable catches of Namibian pilchard for the period 1960–2002

assumed catch is only an average because the stock of pilchard is subject to substantial fluctuations through time.

The result from the analysis is presented in Figure 7.3 for discount rates ranging from 1 to 20 per cent. The sole-owner rational economic agent will prefer the average sustainable catch profile to the observed catch path analysed only for discount rates that are less than or equal to 5 per cent, because the present values of landings are otherwise greater with the observed than with the average sustainable catch profile. Thus if the social discount rate in Namibia during this period was greater than 5 per cent, it cannot be said that pilchard has been subject to economic overfishing.

Hence, an economically rational sole owner of the pilchard resources would have achieved a higher present value of landings by pursuing the observed catch profile rather than an assumed average sustainable catch profile for discount rates greater than 5 per cent. This may be because fishers may have been able to anticipate and incorporate the expected adverse environmental effects on the present value of their landings, resulting in the observed landings profile. It would seem that biological (not economic) overfishing and adverse environmental conditions worked together to deplete pilchard stocks.

DISCUSSION AND CONCLUDING REMARKS

Our review of the literature reveals that the pilchard biomass collapses in 1971 and 1977/78 are largely attributable to overfishing. However, the collapse of the 1990s is blamed principally on the effect of environmental change. From the biomass and catch profiles, we see a steep decline in both the quantity of pilchard in the ocean, and the amount of catch taken over the period analysed. It can, therefore, be concluded that biological overfishing (due to the interplay of fishing and environmental changes) has occurred. On the other hand, economic overfishing cannot be claimed, because, for realistic levels of discount rates, the present value of landings is higher under the observed catch profile than when an average sustainable harvest regime is implemented.

A number of coping strategies were used to absorb the sharp declines in catches by both the pilchard harvesting and processing sectors in Namibia. Before Namibian independence, most of the vessels used to exploit pilchard were foreign, so they simply moved to other parts of the world (Northwest Africa, South America). In recent years, Namibian vessels were licensed to fish in Angola to reduce the pressure on the harvesting sector. In addition, horse mackerel TAC was allocated to the processing sector for fishmeal production. To improve economic returns, a larger proportion of pilchard landings are used for canning as opposed to fishmeal when total landings are low. In this way, the higher prices commanded by canned pilchard helped improve the total economic returns to the fishery. Furthermore, because canning is more labour-intensive, the diversion of the catch from fishmeal to canning also contributed to softening the impact of the declines. In respect of the consequences of the collapse of the pilchard population on workers in the pilchard fishery, it appears in general that seasonal workers from the north of Namibia probably paid the highest price for the collapse of pilchard.

ACKNOWLEDGEMENTS

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8. Climate change and small pelagic fisheries in developing Asia: the economic impact on fish producers and consumers

Roehl Briones, Len Garces and Mahfuzuddin Ahmed

INTRODUCTION

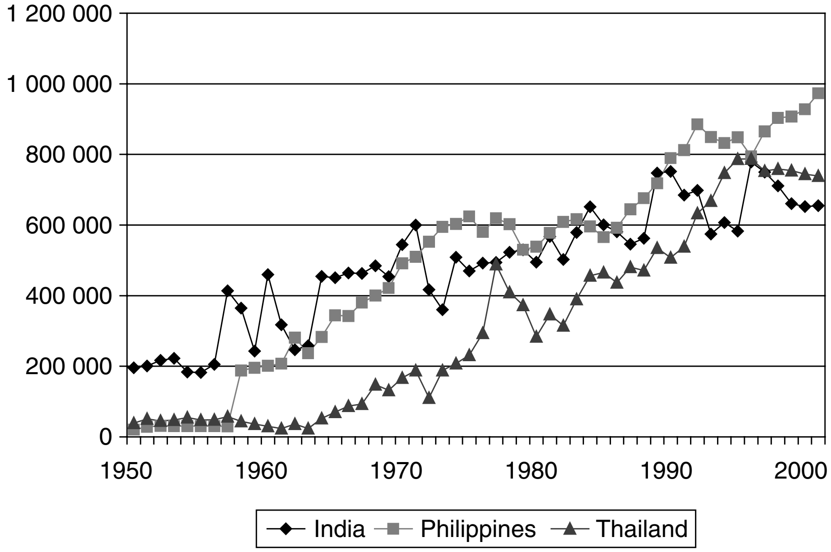
In developing Asia, the risk of climate change poses additional threats to the livelihood and food security of poor households dependent on small pelagic fisheries. The Food and Agriculture Organization (FAO) estimates that around 22 million people in Asia obtain their livelihood from fishing. The bulk of these are small scale or artisanal fishers in coastal areas, for whom small pelagics represent an important subset of their target species. On the consumption side, fish contributes more than one-fourth of animal protein intake in Asia (FAO, 2002); fish consumption is concentrated on low value species (Kent, 1998), which includes most of the small pelagics.

Overfishing and other human activities have already reduced small pelagic resources in Asia to a precarious state. Conceivably, climate change could tip some of the more endangered stocks over the edge. The first step towards gauging the magnitude of the threat to livelihoods and food security is to assess the economic consequences of a decline in fish supply. This study contributes to such an assessment by applying the tools of supply-demand and welfare analysis. We juxtapose three case studies, namely India, the Philippines, and Thailand, in which there is a high dependence of the poor on small pelagic fisheries, and for which systematic secondary information is available.[1]

DESCRIPTION OF SMALL PELAGIC FISHERIES

Catches of small pelagic resources in the Asia-Pacific region have been declining since the late 1980s. Catches fell by 6 per cent from 13.1 million tonnes in 1988 to 12.7 million tonnes in 1995 (Devaraj and Martosobroto, 1997). However, catches in some Asian countries actually increased, China being the leading contributor, followed by the Philippines, Indonesia and Thailand. The immediate cause was increased fishing effort as well as improved fishing efficiency, for example through the use of Fish Aggregating Devices (FADs), lights and navigational aids (such as the Global Positioning System, or GPS). A set of studies collected in Silvestre *et al.* (2003a) note, however, that the catch trends have masked the deterioration of the underlying resource base, especially for key species such as sardines, anchovies, scads and mackerels.

Coastal fishers in India, the Philippines and Thailand deploy a diverse variety of gear and vessel types. Artisanal fishers relying on the simplest gears and vessels are highly dependent on small pelagic fisheries. For example, in the Philippines, small-scale municipal fishers, who legally have exclusive access to inshore waters, largely target small pelagics using hook



Source: FAO Fishstat.

Figure 8.1 Annual catch of small pelagic fish in selected Asian countries, 1950–2000, in tonnes

Table 8.1 Inshore fish catch by major fish species and fishing gears, Philippines, 1995, in tonnes

Fish types	Hook and line	Ring-net	Fish corral	Gill net	Others
Squid	7407	223	1394	14857	19420
Fimbrated sardines	1640	1765	1672	33187	4999
Frigate tuna	20171	3385	1561	7462	9780
Anchovies	265	2576	3858	7072	28137
Indian sardines	2377	11	391	15901	13922
Yellowfin tuna	25505	78	58	1658	3004
Blue crabs	21	38	838	16695	10349
Indian mackerel	1919	1673	197	19939	3881
Big-eyed Scad	16743	614	388	4363	1895
Roundscad	4620	6219	83	6901	6132
Others	106834	23934	20616	129986	166755
% of total	23.9	5.2	4.0	32.9	34.2

Source: Trinidad (2003).

and line as well as gill nets (Table 8.1). In India, small-scale fishers using non-mechanized vessels are estimated to earn US$500–1 200 annually per household, placing them on average well under the poverty line (Dey, 2004).

Supongpan (1996) reports that sustainable levels of catch have long been reached or even exceeded for a number of major stocks in the Gulf of Thailand, namely: Indo-Pacific mackerel (since 1984), sardines (since 1988), anchovies (since 1990), small tunas (since 1988) and roundscad (since 1977). For India, the catches of pelagic fish have either been very close to or have exceeded the potential yield in the late 1990s, particularly for lesser sardines, whitebaits, carangids and mackerel (Vivekanandan *et al*., 2003).

Another indicator that points to excessive fishing pressure is the exploitation status, defined as the ratio of fishing mortality to total mortality. For maximum biological yield, population models indicate a range of 0.30– 0.50 as a norm for the exploitation status. As can be seen in Table 8.2, for most species on which the indicator is available, the norm is exceeded, in some instances by a large margin.

Status of Small Pelagic Stocks

As noted in the foregoing, catches are misleading indicators of fish abundance. A more appropriate indicator is cpue (catch per unit effort). Dalzell *et al*. (1987) has calculated cpue for the Philippines from 1955–1985, where

Table 8.2 Estimates of mean mortality parameters and exploitation ratio of some small pelagic species in South and Southeast Asia

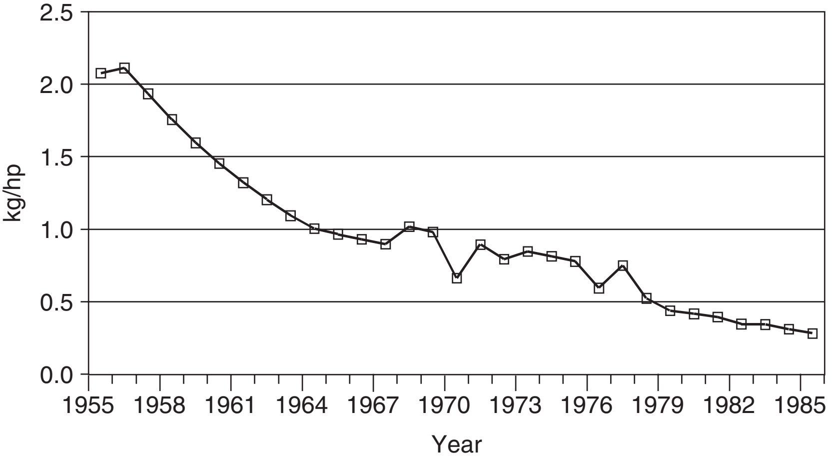
Family/Species	Natural mortality	Fishing mortality	Total mortality	Exploitation status
CARANGIDAE				
Alepes djedaba	1.77	5.43	7.20	0.78
Atropus atropus	1.51	5.14	6.65	0.77
Atule mate	1.64	2.90	4.54	0.64
Carangoides equula	1.25	1.53	2.78	0.56
Carangoides malabaricus	1.48	4.03	5.51	0.64
Carangoides ciliarius	1.80	2.04	3.84	0.52
Caranx hippos	1.18	4.05	5.23	0.76
Caranx malabaricus	1.61	3.38	4.99	0.57
Decapterus macrosoma	1.90	3.62	5.14	0.61
Decapterus maruadsi	4.00	2.40	3.34	0.61
Decapterus russelli	1.77	4.22	6.24	0.63
Megalaspis cordyla	2.16	3.55	3.91	0.53
Parastomateus niger	1.14	2.78	3.93	0.63
Selar crumenopthalmus	2.59	5.55	4.86	0.60
Selar mate	1.66	2.09	3.75	0.56
Selaroides leptolepis	5.21	3.81	6.49	0.59
ENGRAULIDAE				
Coilia dussumieri	1.77	2.30	3.98	0.56
Stolephorus commersonii	2.31	3.55	5.47	0.57
Stolephorus heteroloba	2.70	8.56	11.26	0.76
Stolephorus indicus	3.98	3.37	6.03	0.61
Stolephorus tri	2.55	7.25	9.80	0.72

Source: Silvestre *et al.* (2003b).

effort is measured in horsepower-equivalent units (Figure 8.2). Cpue was found to have plummeted from 2.10 t/hp in 1955 to just 0.29 t/hp in 1985 – an 85 per cent decline, usually attributed to overfishing. Trinidad (2003) estimates that fishing effort is 30 per cent above sustainable levels, resulting in economic loss from rent dissipation in the order of US$125 million per year.

Global Climate Change and Small Pelagic Stocks

A number of studies have documented the ecological impacts of global climate change on aquatic resources (Klyashtorin, 2001; Chavez *et al.*, 2003).



Source: Dalzell *et al.* (1987).

Figure 8.2 Catch per unit of effort for the Philippine inshore fishery, 1955–1985

Climate change can cause environmental adjustments such as: changes in precipitation patterns and in atmospheric carbon dioxide concentration; increase in temperature/chemical reaction rates affecting aquatic production; and changes in wind and ocean circulation patterns. Particularly sensitive to environmental fluctuation is plankton; the biomass of small pelagics, which are usually short-lived plankton feeders, is therefore strongly influenced by climate change.

In the Pacific, coupled changes in the atmosphere and the oceans occur irregularly every few years, causing the *El Niño*-Southern Oscillation or ENSO (Botsford *et al.*, 1997; Francis *et al.*, 1998). ENSO events along the coast of South and North America may have reduced survival and growth rates of some fishes such as salmon, mackerel and anchovetta. The reduction occurs through the decline in primary productivity near the equator, with adverse consequences for zooplankton productivity in the California Current (Francis *et al.*, 1998; Lu *et al.*, 1998; Tsai *et al.*, 1997; Chavez *et al.*, 2003). Climate change has been implicated in the observed fluctuations of small pelagic fish abundance; overfishing, by reducing fish stocks to a fragile state, has amplified the threat posed by climate change. Indeed it is likely that, through ecosystem and trophic interactions, overfishing may exacerbate the natural variability of population size in relation to climatic fluctuation (Pauly, 2003). Currently however, it is difficult to answer the question: how much of the catch fluctuation is actually due to climate change? The absence of environmental information, uncertainty of ecological interactions, and the unavailability of fishing effort data preclude a straightforward answer. However one can conduct a meaningful analysis on a different question: if climate change does introduce a decline in fishing productivity, what would be the implications for fish producers and consumers? This question is addressed below.

MODELLING THE IMPACT OF CLIMATE CHANGE

Modelling Approaches

Quantitative analysis of the impacts of climate change often takes the bioeconomic modelling approach. Bioeconomic models have been developed to varying degrees of complexity in terms of population dynamics and ecosystem interactions. A great merit of such approaches is that climate change scenarios can be incorporated by altering an appropriate set of ecosystem or population parameters.

Another approach is to represent climate change in terms of a supply shock, and work out its economic consequence using the microeconomic tools of supply and demand. A prominent example of this is the 'fisheries collapse' scenario analysed by the extended International Model for Policy Analysis of Commodities and Trade model or IMPACT (Delgado *et al.*, 2003). The main advantage of this approach is that it incorporates price response as well as intersectoral effects (when the supply-demand analysis encompasses an array of related markets). Such an analysis can temper large first-order effects of stock decline, which are subsequently mitigated by price adjustments in the face of diminishing supplies. On the other hand, price adjustments in one market can have repercussions, both favourable and unfavourable, on other markets whose demands and supplies are related to the market from which the shock originated. The trade-off though is that this type of analysis typically does not have the richness of detail that some bioeconomic models can attain in terms of capturing resource and ecosystem interactions.

Structure of the Fish Sector Model

The structure of the fish sector model used here closely resembles that of the AsiaFish model, which is a set of country-specific models for selected Asian countries (Bangladesh, China, India, Indonesia, Malaysia, the Philippines, Sri Lanka, Thailand and Vietnam). A detailed description of this model, including a list of structural equations, is found in Dey *et al.* (in press).

The model is composed of a producer core, a consumer, core and a trade core. The producer core is divided into two production categories, namely capture and culture. For some fish types the two categories are assumed to produce approximately homogeneous products; for others the distinction carries over in the fish type (for example crustacean cultured versus crustacean captured). The supply equations take the linear multiple-product form as derived from the normalized quadratic profit function. Proportional supply shifts are incorporated by the formalism of the 'effective price' (Alston, Norton and Pardey, 1995). The consumer core represents the household fish demand following a two-stage budgeting framework. The first stage determines expenditure on fish as determined by income, and prices of food and non-food items, under a double-logarithmic specification. The second stage determines fish expenditure shares under a quadratic Almost Ideal Demand System (AIDS). For the producer and consumer cores, the model structure is an application of the Martin and Alston (1994) procedure to the fish sector. Demand for non-food uses of fish is incorporated as a fixed ratio to food demand.

The trade core incorporates exports and imports by a differentiated-product formalization. Total demand is disaggregated into demand for domestically produced fish and demand for imported fish; imports and domestic production are treated as differentiated products. Likewise, total domestic supply is disaggregated into supply for domestic markets and export supply; exports and domestic production are also treated as differentiated products. This is essentially the Armington (1969) approach, though here a fixed-shares aggregation (for both demand and supply) simplifies the Constant Elasticity of Substitution (CES) formulation in the original. Both import and export prices are deemed fixed (the small open economy assumption). Equilibrium occurs when demand equals supply. The model allows non-identical fish classifications on the supply side and demand side; model closure would require the matching of these types, by a method discussed below. The model solution entails finding the domestic prices in each matched fish type that will simultaneously clear all markets. Impact analysis compares model solutions with and without a shift in the relevant supply function. From the changes in prices and quantities, combined with the functional form of the supply and demand equations, changes in producer and consumer surplus can be calculated as a measure of the welfare impact of a supply shock.

Numerical Implementation

Based on the foregoing structure, three independent country models are programmed using the Generalized Algebraic Modeling Solver (GAMS) software. The baseline data is taken from the FAO databases (Fishstat, FAOStat Fisheries – Primary, and FAOStat Fisheries – Processed). A common classification of fish types is implemented:

1. Supply fish types:
   - Carp-tilapias (CARTIL) – cichlids and cyprinids
   - High value freshwater and diadromous fish (HVFWD) – salmons, trouts, smelts; sturgeons and paddlefishes
   - Other freshwater and diadromous fish (OTHFWD)
   - High value pelagic marine fish (HVPEL) – tunas and bonitos
   - Low value pelagic marine fish (LVPEL) – all other pelagics
   - Demersals cultured (DEMCUL)
   - Demersals captured (DEMCAP)
   - Crustaceans cultured (CRUSTCUL)
   - Crustaceans captured (CRUSTCAP)
   - Molluscs (MOLL) – except cephalopods
   - Cephalopods (CEPH)
   - Other marine fish (OTHM)
2. Demand fish types:
   - Low value fish (LVF)
   - High value finfish (HVFNF)
   - Crustaceans (CRUST)
   - Molluscs (MOLL)
   - Cephalopods (CEPH)
   - Other marine fish (OTHM)

The model allows CARTIL, OTHFWD, HVFWD, HVPEL, LVPEL, MOLL and OTHM to be sourced from two categories (capture and culture). Matching of demand and supply is shown in Table 8.3. LVF, HVFNF, and CRUST are demand 'composites' to be disaggregated to the corresponding fish types in supply. Model closure is stated in terms of the supply fish types.

For the base year 2001, data on quantities and prices are assumed to be at supply-demand equilibrium. Fixed shares (for the trade core, for the matching of the demand composites, and for the non-food fish demand) are calibrated from the baseline data. Processed fish is converted back to fresh fish weight and is reintegrated into the source fish type. Price response parameters are calibrated using prior estimates of demand and supply elasticities derived in part from the elasticity estimation exercise for the AsiaFish model, and in part by a review of the literature. Intercept

Table 8.3 Matching of demand fish types with supply fish types in the fish sector model

Demand	Supply
LVF	CARTIL OTHFWD LVPEL
HVFNF	HVFWD HVPEL DEMCUL DEMCAP
CRUST	CRUSTCUL CRUSTCAP
MOLL	MOLL
CEPH	CEPH
OTHM	OTHM

parameters can then be calibrated from the baseline data. Baseline information on supply and demand in the three countries is presented in Tables 8.4 and 8.5 respectively.

The scenarios for simulation are stated in terms of negative supply shocks to Low value pelagics, which are run at the five, ten and twenty per cent levels. These levels are deemed to be illustrative of the range of severity associated with the production impact of climate change. Changes in welfare are approximated by a Taylor expansion of the supply function (for the producer surplus component) and a similar expansion of the quadratic AIDS (for the consumer surplus component).

SIMULATION RESULTS

The simulation results are presented by country in the form of three tabulations: the first for supply impact, the second for consumption impact and the third for welfare impact.

India

The supply shock results in nearly the same proportional adjustment in the equilibrium quantities of the directly affected fish type (Table 8.6). That is,

Table 8.4 Baseline data on supply and producer price for India, the Philippines, and Thailand, in 2001

	India	Philippines	Thailand
Quantity supplied (t)			
CARTIL	2 112 909	160 757	250 426
OTHFWD	934 482	254 787	243 435
DEMCUL	0	203	1 467
DEMCAP	953 224	311 498	384 815
LVPEL	788 250	1 324 684	870 943
HVPEL	350	23 191	8 465
CRUSTCUL	127 160	47 021	292 077
CRUSTCAP	371 667	85 061	139 356
MOLL	2 722	102 145	224 222
CEPH	114 681	56 008	170 945
OTHM	559 785	13 588	934 909
Producer price (US$/kg)			
CARTIL	0.86	2.69	0.87
OTHFWD	0.86	2.69	0.87
DEMCUL	–	3.25	4.45
DEMCAP	1.56	3.25	4.45
LVPEL	1.99	2.65	0.78
HVPEL	0.85	1.94	1.70
CRUSTCUL	3.91	2.65	0.78
CRUSTCAP	3.91	2.65	0.78
MOLL	0.31	0.39	0.27
CEPH	1.99	3.00	1.00
OTHM	5.65	13.00	6.00

Source: Authors' calculation using basic FAO data.

LVPEL declines in quantity by nearly 5, 10, and 20 per cent even after incorporating price adjustments. Other fish types (except for OTHM) undergo mild increases in production. Meanwhile the price increases for LVPEL are somewhat higher than the percentage shock. Other prices (again save for OTHM) also rise, though slightly.

The fact that price increases by a greater proportion than the production decline suggests that the second-order effects (induced by market adjustments) from climate change may end up amplifying the original adverse shock, as fishing effort rises. This point is often missed in standard bio-economic analysis, which assumes a fixed price. At this point though, the possibility of increased effort as a consequence of market adjustment

Table 8.5 Baseline data on demand and consumer price for India, the Philippines and Thailand in 2001

	India	Philippines	Thailand
Quantity demanded (mt)			
LVF	3 873 703	1 717 644	1 387 582
HVFNF	951 936	334 596	294 820
CRUST	345 284	87 746	133 099
MOLL	0	96 781	194 837
CEPH	55 408	54 695	131 975
OTHM	327 649	8 320	1 271 883
Consumer price (USD/kg)			
LVF	1.10	2.58	0.85
HVFNF	1.56	3.16	3.94
CRUST	3.91	2.64	1.64
MOLL	–	0.40	0.25
CEPH	1.99	2.58	0.99
OTHM	5.59	9.56	4.56
Share in total fish expenditure (%)			
LVF	45.7	74.5	32.4
HVFNF	15.2	17.2	45.8
CRUST	16.1	3.9	8.7
MOLL	0.0	0.7	1.9
CEPH	1.3	2.4	5.2
OTHM	21.8	1.3	6.0

Source: Authors' calculation using basic FAO data.

remains an intriguing conjecture, as fishing effort and stock dynamics are not part of the model. On the demand side (Table 8.7), consumption tends to fall, with the greatest decline in Low value fish (the demand composite that includes LVPEL). Most of the consumer prices tend to rise. Changes on the demand side are moderate; even a 20 per cent shock is projected to raise LVF prices by only 4.5 per cent.

The welfare impacts are shown in Table 8.8. Under all scenarios the simulations indicate modest gains in other sectors, for example freshwater culture and capture, as well as marine culture. These are however dwarfed in absolute terms by the decline in net producer income in the marine sector. This in turn is much smaller than the loss to consumers from higher prices and the corresponding lower consumption – the size of the consumer surplus loss is over 80 per cent of the total economic loss in all shock scenarios. The total loss to society under a minor supply shock is

Table 8.6 Changes in quantity supplied and producer price under alternative supply shocks in India (as a percentage)

Shock	Quantity supplied			Producer price		
	5%	10%	20%	5%	10%	20%
CARTIL	0.03	0.06	0.13	0.07	0.13	0.26
OTHFWD	0.03	0.05	0.09	0.08	0.15	0.29
Dem_cap	0.0	0.0	0.0	0.0	0.0	0.0
LVPEL	−4.95	−9.91	−19.82	5.45	11.5	25.93
HVPEL	0.04	0.1	0.2	0.14	0.32	0.68
CRUST_cul	0.09	0.18	0.36	0.19	0.36	0.72
CRUST_cap	0.07	0.13	0.26	0.22	0.43	0.86
CEPH	0.25	0.47	0.93	0.83	1.57	3.11
OTHM	−0.19	−0.37	−0.74	−0.62	−1.23	−2.47

Source: Authors' calculations.

Table 8.7 Changes in quantity demanded and consumer price under alternative supply shocks in India (as a percentage)

Shock	Quantity demanded			Consumer price		
	5%	10%	20%	5%	10%	20%
LVF	−0.98	−1.97	−3.94	1.1	2.21	4.5
HVFNF	−0.16	−0.16	−0.16	0.34	0.57	1.03
CRUST	0.01	0.02	0.03	0.21	0.42	0.82
CEPH	−0.17	−0.33	−0.64	0.82	1.55	3.06
OTHM	0.07	0.14	0.27	−0.61	−1.21	−2.44

Source: Authors' calculations.

US$53 million, which escalates to over US$210 million for the large shock scenario.

It is noteworthy that a back-of-the-envelope calculation of welfare loss (price × supply shock, using Table 8.4 and 8.5 data) yields a figure of over US$300 million, which seems to overestimate the loss. Furthermore, the population of affected consumers and producers is quite large – >300 million fish consumers and >5 million fishers. On average the impact is minimal. However it is possible that certain vulnerable sub-sectors may be disproportionately affected, that is the poorest fishers using non-mechanized boats and using either gill net or hook and line.

Table 8.8 Changes in welfare by sector and supply shock in India (in US$ 1000)

Shock	5%	10%	20%
Change in producer surplus:			
Freshwater culture	1 261	2 362	4 626
Freshwater capture	649	1 216	2 381
Marine – cultured only	927	1 798	3 569
Marine – captured only	−11 736	−23 476	−46 687
Change in consumer surplus	−44 201	−86 844	−174 018
Change in economic surplus	−53 099	−104 944	−210 129

Source: Authors' calculations.

Table 8.9 Changes in quantity supplied and producer price under alternative supply shocks, Philippines (in per cent)

Shock	Quantity supplied			Producer price		
	5%	10%	20%	5%	10%	20%
CARTIL	−0.59	−1.19	−2.47	−1.28	−2.61	−5.41
OTHFWD	−0.61	−1.23	−2.55	−1.26	−2.57	−5.33
DEMCUL	−0.34	−0.69	−1.45	−0.68	−1.39	−2.90
DEMCAP	−0.23	−0.48	−1.00	−0.78	−1.60	−3.34
LVPEL	−5.34	−10.66	−21.2	3.99	8.40	18.74
HVPEL	−0.25	−0.51	−1.05	−0.82	−1.68	−3.52
CRUSTCUL	−0.31	−0.61	−1.17	−0.34	−0.67	−1.30
CRUSTCAP	−0.17	−0.33	−0.63	−0.55	−1.09	−2.11
MOLL	1.61	3.34	7.15	4.44	9.17	19.66
CEPH	0.03	0.08	0.24	0.10	0.26	0.81
OTHM	0.44	0.88	1.80	1.45	2.93	5.97

Source: Authors' calculations.

The Philippines

As with the case of India, supply shocks lead to proportionately similar adjustments in the equilibrium quantity of LVPEL (Table 8.9); likewise price increases by proportionately more than the supply shock (except for the 5 per cent case). The major contrast with India is that most of the other sectors also suffer output contractions; these contracting fish types also

Table 8.10 Changes in quantity demanded and consumer prices under alternative supply shocks in the Philippines (as a percentage)

Shock	Quantity demanded			Consumer price		
	5%	10%	20%	5%	10%	20%
LVF	−4.24	−8.47	−16.86	2.52	5.2	11.12
HVFNF	−0.23	−0.48	−1.00	−0.78	−1.6	−3.34
CRUST	−0.06	−0.11	−0.22	−0.47	−0.94	−1.81
MOLL	1.38	2.85	6.07	4.39	9.08	19.43
CEPH	0.03	0.07	0.23	0.07	0.19	0.6
OTHM	0.04	0.09	0.20	1.01	2.04	4.12

Source: Authors' calculations.

Table 8.11 Changes in welfare by sector and supply shock in the Philippines (in US$ 1000)

Shock	5%	10%	20%
Change in producer surplus:			
Freshwater culture	−12 083	−24 490	−50 368
Freshwater capture	−2 082	−4 225	−8 713
Cultured marine	595	1 243	2 723
Captured marine	−38 434	−77 598	−158 175
Marine – cultured only	−428	−844	−1 632
Marine – captured only	−9 379	−18 966	−38 752
Change in consumer surplus	−102 349	−206 690	−422 406
Change in economic surplus	−164 158	−331 569	−677 323

Source: Authors' calculations.

suffer from lower prices. This suggests that complementarity effects outweigh substitution effects. On the other hand the shock does seem to cause net substitution effects from MOLL, CEPH and OTHM fish.

On the demand side (Table 8.10), the simulations suggest dramatic decreases in quantity demanded as well as increases in consumer price for LVF, which is probably because LVPEL weighs heavily in the fish consumption basket. The responses in consumption and prices of the other fish types suggest strong market interactions between small pelagics and the other fish sub-sectors.

Welfare impact is shown in Table 8.11. (Note that the last rows of the table reflect a distinction between marine fish types that are both captured and cultured, that is 'cultured marine' and 'captured marine', from marine fish types that are exclusively cultured or exclusively captured, that is 'marine – cultured only' and 'marine – captured only'). Given the importance of LVPEL in the Philippines, it is not surprising that the negative impacts are sizeable – almost US$700 million in the worst case scenario. This time a back-of-the-envelope figure (a little over US$700 million) is within range of the estimate. However the simulation allows a disaggregation of this loss: Table 8.11 shows that producer income from other production categories also suffers, consistent with projected contractions both in price and quantity for some other fish types. While the decline in producer surplus is largest in the captured marine category (which includes LVPEL), the total decline in producer surplus of the other categories is also serious (about sixty per cent of the size of the impact in the captured marine category). This suggests that the negative production impact of the supply shock has been distributed widely throughout the aquaculture and fisheries sector through the market mechanism. Owing to the large production impact, the share of the decline in consumer surplus in total economic loss (some 60 per cent) is lower than that of India.

Thailand

In the case of Thailand, note first that the LVPEL price is quite low in this country; meanwhile the share of HVF consumption (as well as production) is rather high – higher than that of Low value fish (Table 8.5). This is consistent with the relatively higher standard of living in Thailand. These and similar considerations may yield some counterintuitive results within a multi-market analysis.

Consider the supply impacts (Table 8.12). For LVPEL, the proportional decreases in quantity are reasonable enough, but the proportional price increases have simply been exaggerated by the low baseline value. As in the Philippines, many of the fish types undergo both an output and price contraction. Meanwhile quantity demanded declines not only for LVF, but also for three out of the five other demand fish types (Table 8.13). Attenuated price increases characterize LVF, though the other types undergo price reductions (save for OTHM).

The most unusual results are obtained for the welfare calculation (Table 8.14). The producer surplus in marine capture (which contains LVPEL) receives a greater producer surplus; this is because the price increases more than compensate for increase in unit costs. While rare, this occurrence is not unheard of in welfare analysis.

The producer surplus impacts for the other supply categories are comparatively minor. Meanwhile the consumer price changes imply a decline

Table 8.12 *Changes in quantity supplied and producer price under alternative supply shocks in Thailand (as a percentage)*

Shock	Quantity supplied			Producer price		
	5%	10%	20%	5%	10%	20%
CARTIL	−0.47	−0.94	−1.87	−1.11	−2.21	−4.39
OTHFWD	−0.45	−0.91	−1.81	−1.13	−2.26	−4.48
DEMCUL	−0.08	−0.17	−0.34	−0.17	−0.34	−0.69
DEMCAP	−0.06	−0.12	−0.25	−0.20	−0.41	−0.83
LVPEL	−3.61	−7.11	−13.74	10.41	23.00	57.60
HVPEL	−0.10	−0.21	−0.43	−0.35	−0.70	−1.42
CRUSTCUL	−0.50	−0.99	−1.95	−1.00	−1.99	−3.91
CRUSTCAP	−0.42	−0.82	−1.61	−1.38	−2.75	−5.38
MOLL	−0.34	−0.69	−1.36	−0.69	−1.38	−2.72
CEPH	−0.15	−0.30	−0.60	−0.50	−1.00	−2.00
OTHM	0.51	1.02	2.04	1.69	3.40	6.80

Source: Authors' calculations.

Table 8.13 *Changes in quantity demanded and consumer prices under alternative supply shocks in Thailand (as a percentage)*

Shock	Quantity demanded			Consumer price		
	5%	10%	20%	5%	10%	20%
LVF	−2.45	−4.84	−9.36	1.21	2.42	4.80
HVFNF	−0.02	−0.04	−0.07	−0.17	−0.35	−0.72
CRUST	0.08	0.16	0.33	−0.79	−1.58	−3.13
MOLL	−0.27	−0.54	−1.07	−0.57	−1.14	−2.26
CEPH	−0.09	−0.18	−0.36	−0.26	−0.52	−1.04
OTHM	0.95	1.91	3.83	1.21	2.41	4.78

Source: Authors' calculations.

in consumer surplus for all the scenarios. The net effect of course is a net loss to society; the loss turns out to be minimal for the low-shock scenario, but can exceed US$300 million in the worst-case scenario. In this case the back-of-the-envelope calculation (below US$140 million) is an underestimate.

Table 8.14 Changes in welfare by sector and supply shock in Thailand (in US$ 1000)

Shock	5%	10%	20%
Change in producer surplus:			
Freshwater culture	−2 758	−5 499	−10 860
Freshwater capture	−2 049	−4 090	−8 094
Cultured marine	−271	−539	−1 062
Captured marine	−147	−293	−577
Marine – cultured only	−2 284	−4 532	−8 868
Marine – captured only	117 574	243 083	524 155
Change in consumer surplus	−111 215	−284 379	−805 229
Change in economic surplus	−1 149	−56 249	−310 534

Source: Authors' calculations.

CONCLUSION AND DIRECTIONS FOR RESEARCH

Climate change poses an external threat to the productivity of small pelagic fisheries. In developing Asia, the threat is more alarming given the dependence of poor households on small pelagic fish stocks, as well as the fragile state to which these stocks have been reduced by decades of overfishing. Our study assesses the potential economic implications of this threat using supply and demand analysis. In contrast to traditional bioeconomic approaches, in which product price is typically kept fixed, this analytical approach assigns a central role to price adjustment within a multi-market setting. The drawback though is the absence of an in-depth treatment of stock dynamics and fishing effort. As such, the simulation scenarios pertain to exogenously specified supply shocks.

As expected, a negative supply shock to small pelagics raises its price and reduces its quantity. However, there are also significant repercussions for prices and quantities in the other fish markets. The size of pecuniary externalities may depend on the initial size and importance of the small pelagic sector. Economic losses from a supply shock can be significant. The bulk of the loss is borne by consumers. However, losses in producer income need not be confined to just fishers of small pelagics, but may be absorbed throughout the fisheries and aquaculture sector.

All this is conjectured upon the actual impact of climate change on productivity. A more comprehensive study would integrate socioeconomic impact analysis with a resource and environmental assessment, including information on fishing effort, catches and biophysical parameters. Ideally, this assessment can support the formulation of strategies and options for managing and maintaining the resilience of small pelagic fisheries in the face of external shocks.

Further research requires an extension of the analysis in two major directions. First, the analyis of supply and demand could be considerably expanded. Baseline information could be collated from national sources using official fish type classifications. The set of countries could be enlarged, to include those that catch a large quantity of small pelagics, for example China and Peru. With more countries, the small open economy assumption becomes untenable; hence the model should be developed to incorporate endogenous world prices. Second, the supply-demand model must be integrated within a wider bioeconomic system. Stock dynamics, extraction costs, environmental fluctuations and other fixtures of the economics of renewable resources should be merged with the standard tools of microeconomics. This would clearly entail a tight interdisciplinary collaboration within a unified, quantitative framework.

Recently many biologists have called for a broadening of management approaches, away from single species assessments towards ecosystem approaches that address the complexities of biotic and abiotic linkages. Analogously, we argue that bioeconomic analysis should no longer treat economic decisions and outcomes for a particular fish type in isolation. Rather, each fish type must be embedded within a wider economic system, in which the interactions between parts are mediated by market prices. While an 'economic system approach' may appear intractable, the challenges are no different from those confronted by the ecosystem approach. Research must be directed towards integrating the entire range of systems linkages, whether in nature or in society, to understand properly the human-fish-climate interface.

APPENDIX 8.1

The species cited in this chapter are listed in Table A8.1. Note that this is by no means a comprehensive list of small pelagic species in Asia. Along the southwestern coast of India, the pelagic resources are exploited by purse-seines operated from mechanized craft (11–14 m long), drift gill nets, ring-seines and hooks and lines (Vivekanandan *et al.*, 2003). In the Gulf of Thailand, pelagic fish are caught by stake traps, purse-seiners (Chinese, Thai, luring and anchovy), encircling gill nets and drift nets. Important pelagic fish are mackerels (*Rastrelliger* spp.), scads (*Decapterus* spp.), sardines (*Sardinella* spp.), anchovies (*Encrasicholina* spp. and *Stolephorus* spp.), king mackerel (*Scomberomorus* spp.), and tuna (*Thunnus* spp. and *Euthynnus* spp.).

Table A8.1 Major small pelagic resources in South and Southeast Asia

Common Name	Family	Representative Species
Round scads	Carangidae	Decapterus macrosoma, D. ruselli, D. maruadsi
Big-eye scads	Carangidae	Selar spp., S. crumenopthalmus
Scads	Carangidae	Atule spp.
Torpedo scads	Carangidae	Megalaspis cordyla
Sardines	Clupeidae	Amblygaster sirm, Sardinella fimbriata, S. longiceps
Round herrings	Clupeidae	Dussumieria acuta
Gizzard shads	Clupeidae	Anodontostoma chacunda
Fusiliers	Caesionidae	Caesio erythrogaster, C. tile, C. diagramma
Anchovies	Engaulidae	Stolephorus spp., Engraulis japonicus
Flying fish	Exocoetidae	Cheilopogon cyanopterus C. nigricans Oxyporhampus convexus Hirundichthys spp.
Half beaks	Hemiramphidae	Hemiramphus far
Bombay duck	Synodontidae	Harpadon nehereus
Silversides	Atherinidae	Atherina spp.
Seerfishes	Scombridae	Scomberomorous spp.
Mackerels	Scombridae	Rastrelliger kanagurta, R. brachysoma, Scomber japonicus

Source: Adapted from Devaraj and Vivekanandan (1997); Trinidad *et al.* (1993).

NOTE

1. The information is collected mainly from two recently concluded projects of the WorldFish Center, namely the 'Sustainable Management of Coastal Fish Stocks in Asia' (Asian Development Bank RETA 5766) and 'Strategies and Options for Increasing and Sustaining Fisheries and Aquaculture Production to Benefit Poor Households in Asia' (ADB RETA 5945).

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9. Bi-national management of a transboundary marine fishery: modelling the destabilizing impacts of erratic climatic shifts

Robert McKelvey, Peter Golubtsov, Kathleen Miller and Greg Cripe

INTRODUCTION: COMPETITIVE HARVESTING IN A FLUCTUATING ENVIRONMENT

The economic and biological implications of competitive harvest of a reproducing biological stock are well-known in the fisheries economics literature. The common-property externality – the fact that no competitor bears the full economic costs of his harvesting – leads to ‘tragedy of the commons’ effects of over-harvesting: both stock depletion and the dissipation of economic rents.

Analyses of circumstances where there are a great many independently acting fishermen go back to the beginnings of modern fisheries economics. In that circumstance, no individual fisherman can significantly impact the stock by a unilateral decision to forgo harvesting. Stylized static formulations were adequate to capture this phenomenon (Gordon, 1954).

The more complex situation, where the harvesters are grouped as members of several internally managed and disciplined national fleets, was first analysed in the early 1970s, using more sophisticated dynamic bioeconomic models, (Clark, 1980; Levhari and Mirman, 1980). Initially it was assumed that the harvests occurred at a common location. Subsequent model variants allowed for harvesting at different places and at different times, and sometimes using different gear types. Most of these models are deterministic, though a few are stochastic.

Models of this sort have proved to be useful in guiding practical management of fisheries. All tell similar stories, centring on the destructive effects of open access. Other factors can be important too, interacting with common property externalities. For example, a substantial literature has developed around the role of overcapitalization and of capital immalleability.

Recently there has been increased awareness of the extent to which oceanic climatic shifts may also be implicated, affecting the oceanic environment, with often dramatic impacts on the productivity and migratory behaviour of economically important fish stocks (McFarlane *et al.*, 2000; Hare and Mantua, 2000; Stenseth *et al.*, 2002; Miller and Munro, 2004). Biologically-important climatic phenomena include *El Niño* events, the Pacific Decadal Oscillation, and the North Atlantic Oscillation. Furthermore, human-induced global climate change will lead to long-term changes in ocean temperature and circulation patterns that cannot yet be foreseen (Barnett *et al.,* 2001).

Empirical evidence of such phenomena is strong (Bakun, 1996, 1998). Stock fluctuations in the Northern Pacific have been studied intensively. Paleo-ecological research demonstrates that fish populations, including Alaskan salmon and California sardine and anchovy stocks have experienced significant long-term fluctuations in abundance for at least the past 2000 years (Baumgartner *et al.,* 1992; Finney *et al.,* 2002). More recently, a North Pacific climatic regime shift in 1977 to a pattern of persistent warm water conditions along the North American west coast affected the productivity and distribution of a wide range of marine organisms (Hare and Mantua, 2000; McFarlane *et al.*, 2000; Hollowed *et al.*, 2001). Even when the effects of strong *El Niños* in 1958–59 and 1983–84 are specifically excluded, Roemmich and McGowan (1995) document an 80 per cent decline in zooplankton biomass off the coast of southern California between the periods 1951–57 and 1987–93. They link the decline to a 1.5°C warming of sea surface temperatures, increased stratification, and less effective upwelling. The Pacific Decadal Oscillation (PDO) Index is one measure of these changing oceanic conditions. When the PDO is in its coastal warm (positive) phase, sea surface temperatures along the west coast of North America are unusually warm, westerly wind stress is stronger than normal, and there is a large area of unusually cool sea surface temperatures in the western and central North Pacific (Mantua *et al.*, 1997). The PDO was in the warm phase during the early 20th century. There was a shift to the coastal cool pattern from the mid-1940s through 1976, then an abrupt return to the warm PDO pattern in 1977, lasting through at least the mid-1990s (Francis *et al.*, 1998).

A strong case has been made for the involvement of the 1977 shift in the PDO in destabilization of bi-national management of the North American Pacific salmon fisheries (McKelvey *et al.*, 2003). Specifically, the shift contributed to significant increases in the productivity of salmon stocks in Southeast Alaska, and plummeting survival rates for stocks spawning in Southern British Columbia and the US West Coast states, which upset the expected distribution of fishery benefits under the terms of the 1985 Pacific

Salmon Treaty. The frustrated expectations, in turn, engendered a protracted dispute.

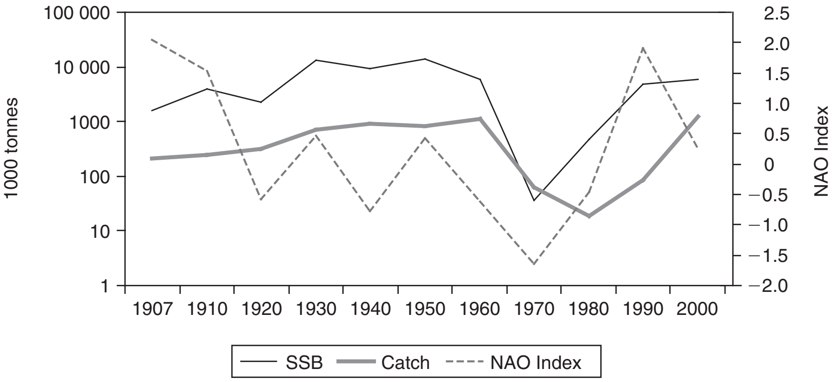
On the other side of the globe, in the Atlantic, the North Atlantic Oscillation (NAO) has been well-documented as a significant source of climatic variability over Europe, Northeastern Canada, Western Asia and North Africa (Van Loon and Rogers, 1978; Hurrell and Dickson, 2003). It is also a major determinant of changes in oceanographic conditions, with impacts on ecosystems and fishery resources in the North Atlantic and adjacent northern seas (Parsons and Lear, 2001; Alheit and Hagen, 1997, 2001; Ottersen and Stenseth, 2001).

Ignorance of the effects of fluctuations in the NAO, combined with excessive harvesting pressure, may account for the otherwise mysterious decline of cod populations in the Grand Banks and the Northwestern Atlantic. In addition, changes in oceanographic conditions associated with variations in the NAO have affected both recruitment success and the migratory behaviour of Norwegian spring spawning herring. During the 1960s and 1970s cold conditions in the Norwegian Sea associated with the negative phase of the NAO led to a series of poor recruitment years, but rapid improvements in fishing technology initially allowed harvests to remain strong. The combined effects of intensive harvesting and poor recruitment caused the stock to collapse. The stock remained very small throughout the 1970s and only began to recover significantly during the late 1980s, after several years of severely restricted fisheries coupled with a sustained shift to a positive NAO pattern (Bjørndal *et al.*, 1998) (Figure 9.1). This is described further in Chapter 2.

The extreme susceptibility to over-harvesting of tightly-schooled small pelagic fishes, especially anchovies and sardines, is a classic story in the fisheries literature. Evidence is now emerging of an important role in the phenomenon of stock migration induced by oceanic temperature shifts (Baumgartner, personal communication).

The modelling reported on in this chapter represents an extension of the classic bioeconomic harvest game analysis, not only incorporating the climatic shift phenomena but more importantly attempting to capture the implications of uncertainty and poor predictability of the onset of such climatic shifts. Hence this modelling focuses on the implications of imperfect information concerning the timing and intensity of relevant environmental shifts.

A particular feature of our study is to examine the role of transparency of information in a harvesting game. In cooperative resource management it is well understood that transparency of information is a positive asset. As we shall see here, in a competitive harvesting game transparency may actually be corrosively destructive.



Source: Adapted, using data from National Center for Atmospheric Research (2005), from Bjørndal (forthcoming), as based on data from Toresen and Østvedt (2000).

Figure 9.1 Norwegian Spring-spawning herring – Spawning Stock Biomass, catch at 10 year intervals and NAO Index averaged over previous 10 years

THE COMPETITIVE SPLIT-STREAM HARVEST GAME

The Split Stream Harvest Model is a discrete-time dynamic model of a competitive marine fishery, on a single non-overlapping-generation fish stock. In this chapter we will examine the evolution of the fishery under the influence of a stochastic oceanic environment which is imperfectly observed by the competing fleet managers.

The dynamic model incorporates successive life cycles of spawning, growth and harvest. A single such cycle may be represented schematically as follows:

$$\begin{tikzcd}
& R_\alpha = \theta_\alpha R \to S_\alpha \arrow[dr] & & \bullet \\
\to R \arrow[ur] \arrow[dr] & & S = S_\alpha + S_\beta \to R^+ = F(S, b) \arrow[ur] \arrow[dr] & \\
& R_\beta = \theta_\beta R \to S_\beta \arrow[ur] & & \bullet
\end{tikzcd}$$

The cycle begins at the harvest phase. Here R is the recruitment biomass of mature harvestable stock. This recruited biomass splits, with sub-stocks migrating along two separate streams, and being subject along the way to harvest by competing national fleets. For a symmetric presentation, let

v denote either α or β. The split-fractions θ_v, with $0 \leq \theta_v \leq 1$ and $\theta_\alpha + \theta_\beta = 1$, determine the v-sub-stream recruitment biomass $R_v = \theta_v R$, which is accessible to harvest by the v fleet.

The harvests then proceed, with the harvested sub-stream biomasses denoted H_v. The corresponding sub-stream escapement is

$$S_v = R_v - H_v.$$

Following the harvest, the sub-stream escapements merge, to form the total escapement biomass

$$S = S_\alpha + S_\beta.$$

This escapement forms the brood stock for the subsequent generation. Following the spawn, and over a period of time, this new generation's biomass grows, while experiencing some mortality, maturing into the new-generation recruitment stock

$$R^+ = F(S,b),$$

measured at the beginning of the new harvesting season. Here b is a growth parameter. Depending on b the stock-recruitment function F may be compensatory, depensatory or even may display critical depensation (see Figure 9.2). With R^+, the second-generation life cycle begins, and this cyclic process repeats through subsequent generations out to the horizon $T \leq \infty$.

Each fleet chooses its harvest policy $H_v(t)$, at each harvest season $t \leq T$, to optimize its expected payoff

$$\Pi_v = E\sum_{t=1}^{T}\delta^{t-1}\int_{S_v}^{R_v}[p_v - c_v/x]dx,$$

while taking into account the current state of the fish stock and the anticipated policy of its competitor. In the simulations we take the horizon T sufficiently large that the harvest policy stabilizes, that is, the dynamic system attains stationarity.

Typically we shall take the environmental parameters of this dynamic system, $\{\theta_v(t)\}$ and $\{b_v(t)\}$, to be stochastic random processes which are observed imperfectly by the fleet managers. Our main focus in this analysis will be on the profound, often surprising, implications of stochasticity, and (especially) of incomplete information, on the outcome of the game.

The mathematical analysis of the dynamic split stream game can be found in McKelvey and Golubtsov (2003) and in McKelvey and Cripe (submitted).

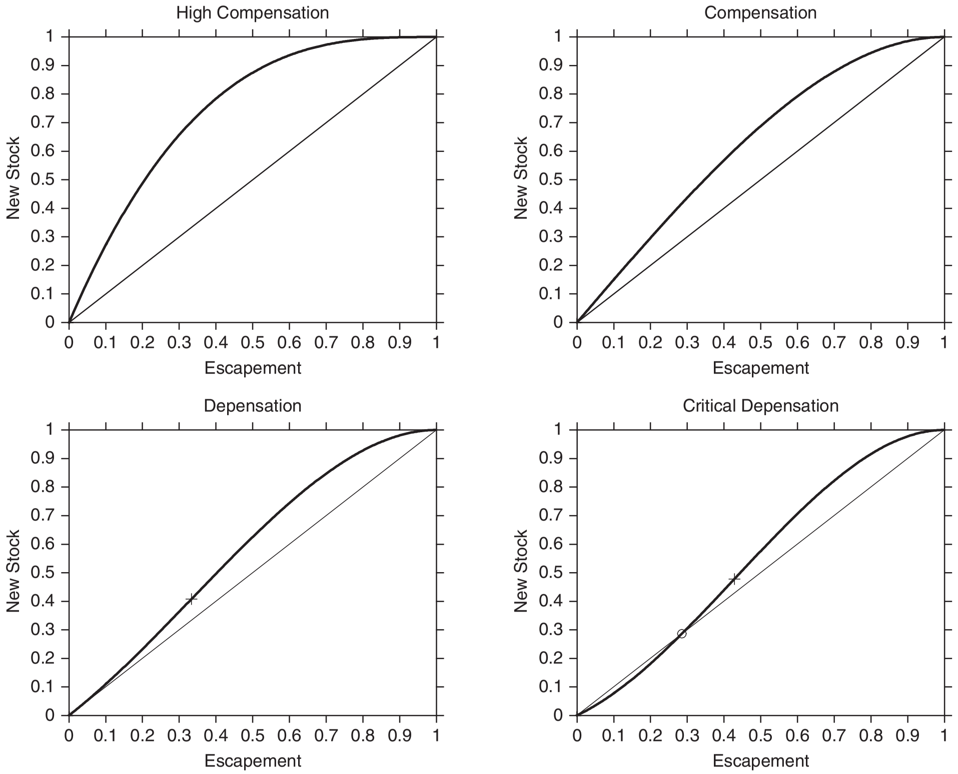


Figure 9.2 *Growth functions*

THE INFORMATION STRUCTURE OF THE GAME

The examples described in this chapter all have a relatively simple information structure. In several other articles we have examined other, more elaborate and perhaps more realistic, information states. Some of these we describe, very briefly, in our final section. However the simpler patterns of behaviour in the simulations that we describe here are quite representative of those observed in the broader class of incomplete-information competitive harvesting games. They illustrate very well the basic implications of information asymmetries in renewable-resource harvesting models.

In this chapter the splitting-parameter sequence $\{\theta_v(t)\}$ forms a two-valued independent identically distributed (iid) Markov chain, with specified values

$$\theta_v(t) = \theta_1^v \text{ or } \theta_2^v, \text{ with } \theta_i^\alpha + \theta_i^\beta = 1 \quad \text{for } i = 1 \text{ or } 2.$$

Furthermore the probability distribution of θ_v:

$$q_i \triangleq \text{prob}[\theta_v(t) = \theta_i^v],$$

is common knowledge held by both fleet managers.

Each fleet manager must adopt a harvest policy for his fleet. At the time when he must make his current harvest decision, he knows the current total recruitment, but cannot directly observe the current value of θ_v, or equivalently of the sub-stream recruitment R_v. Instead he estimates θ_v indirectly, by direct observation of a measurement variable ϑ_v. The random variable ϑ_v takes on the same two realized values as does θ_v, but with different frequencies. Specifically it registers the true-value correctly with the known frequency, with $\text{prob}[\vartheta_v = \theta_v] \geq 1/2$. We define measurement precision r as

$$r = 2 \cdot \text{prob}[\vartheta_v = \theta_v] - 1.$$

Thus $r = 1$ means that ϑ_v measures θ_v with complete accuracy, and $r = 0$ means that the measuring instrument cannot distinguish at all between the realizations of θ_v. In that case knowledge of θ_v is limited to knowing its *a priori* probability distribution.

Similar characterizations might be made concerning the stock growth parameter b. However in most of the examples discussed in this article, b is deterministic. In the exceptional example where b is random, it is iid, and only its probability distribution is known to the fleet managers. We return to a discussion of relevant generalizations of this simplified stochastic and information game structure in a later section.

Finally, we examine potential cooperative solutions of the game. We assume that utility transfers through monetary side payments are feasible, making possible the redistribution of the overall benefits of cooperation to conform to the relative competitive strengths of the fleets. For simplicity we assume that the negotiations over cooperative harvest policies, and the distribution of the competitive surplus, will lead to the classical Nash bargaining solution of our imperfect-information harvesting game.

MODEL OUTPUT: SOME EXAMPLES

There follow several examples of model simulation. No attempt has been made to set parameter values to mimic any particular fishery. Rather, our goal here is to explore parameter space, to illustrate the kinds of phenomena that are possible in principle, and to develop intuition concerning the circumstances and mechanisms for their occurrence.

Each fleet's choice of current harvest landings represents a trade-off at the margin between the expected net value of its immediate landings and the expected contribution to future harvest returns of its sub-stream escapement. Note that, in a competitive fishery, each fleet's calculation of future returns ignores the positive external effect of its current escapement on the future harvest returns to the other. Thus, from a societal perspective, competitive management will induce over-harvesting as compared to cooperative management.

Our central goal will be to examine how the quality of the oceanic environmental information possessed by the individual fleets will influence their trade-off of present against future returns, and the consequent biological and economic implications for the outcome of the harvesting game.

Interplay Between the c_v/p_v Ratio and the Degree of Depensation

Our first simulations examine the role of information on the stochastic switching process on the interplay between the harvest return-rate $p_v - c_v/x$ and the (compensatory or depensatory) stock-recruitment pattern.

Strongly compensatory recruitment (Figure 9.3a)

We begin with an example which illustrates what might be thought of as the 'normal' pattern for the harvesting game. This is the case of high compensation stock-recruitment, as shown in Figure 9.3a. Here c_v is fixed and results are compared for a range of levels of price p_v. Furthermore, the

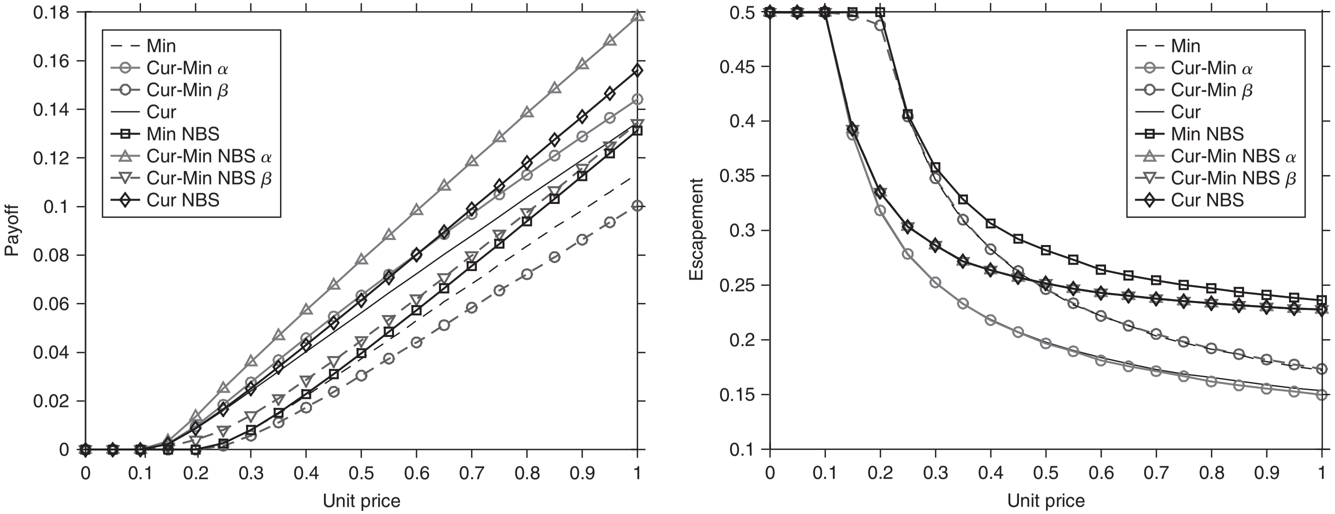


Figure 9.3a *High compensation*

Figure 9.3 *Interplay of landings price and level of compensation*

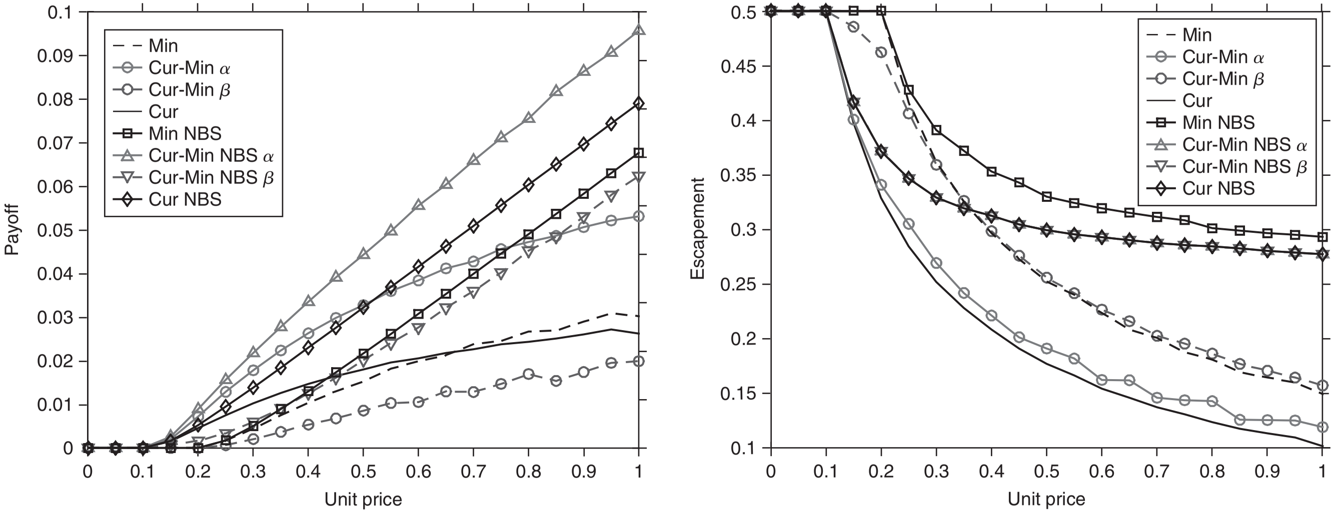


Figure 9.3b Compensation

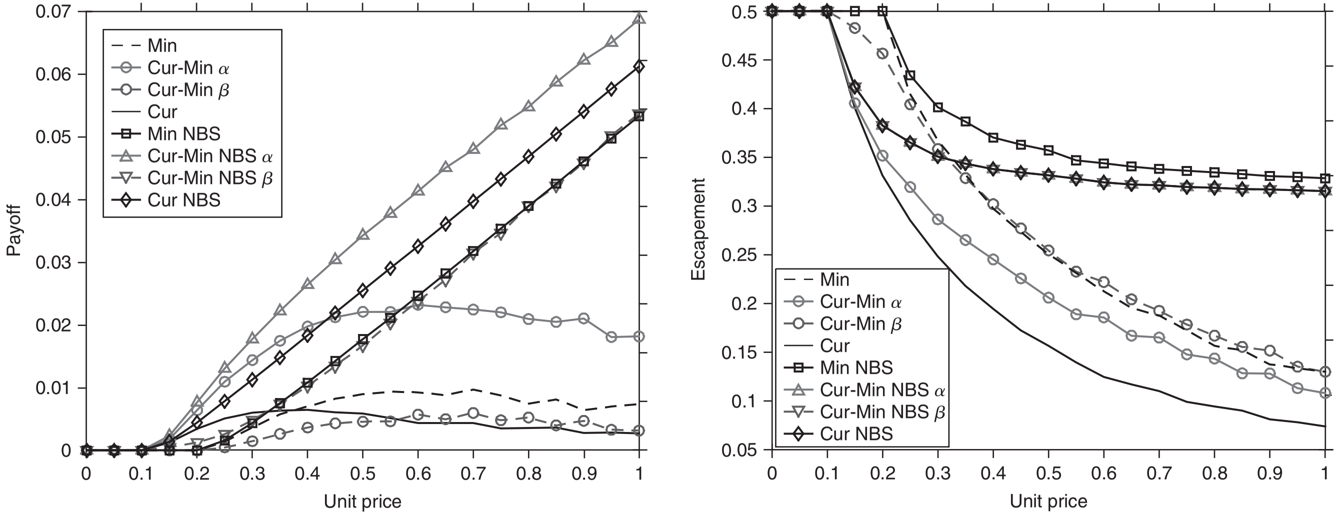


Figure 9.3c *Depensation*

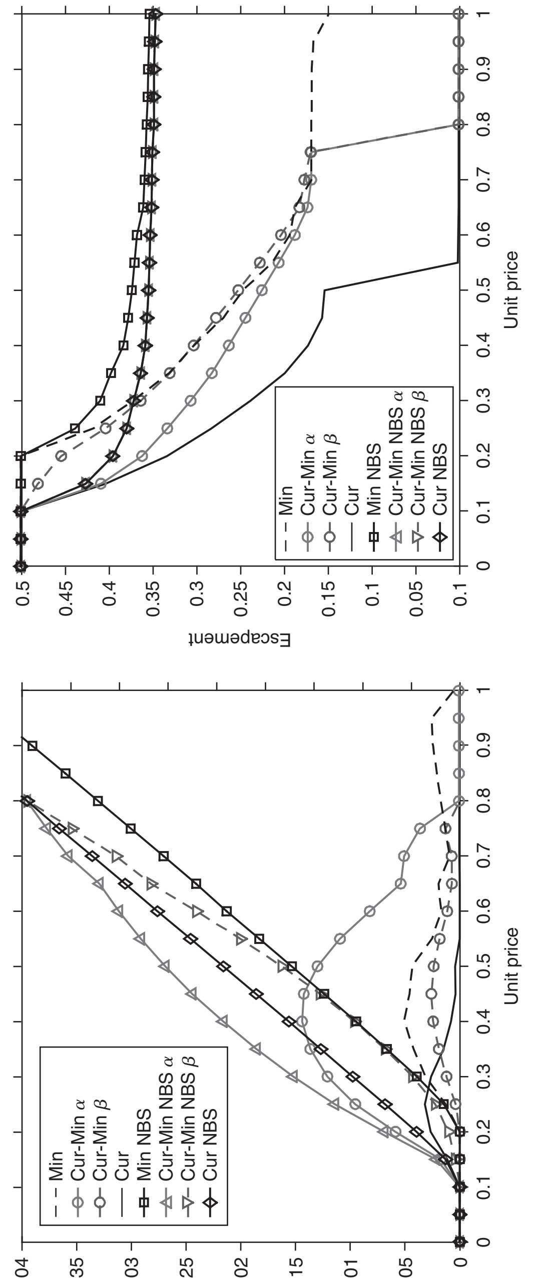


Figure 9.3d *Critical depensation*

stock-recruitment split, while stochastic, is symmetric between streams: that is, the streams are equally likely to draw the high fraction of recruitment, and thus equally also the low fraction.

One can get some insight into this stochastic split-stream situation from the familiar results for a fishery which is deterministic and involves simultaneous harvest at a single common site. It is well known that in this case cooperative harvest is optimized by setting an optimal target harvest escapement. Then the long-run return from the fishery will grow with price, while the recruitment stock will fall.

The situation is famously different under competitive harvest. The extreme case then is that of open access – a very large fleet of small independent and non-cooperative vessels. In the open-access fishery, escapements drop to the break-even bionomic level and stocks are depleted, well below the optimal cooperative level. This is attributed to the fact that an individual vessel's current harvest pattern will have vanishingly small impact on future recruitments. Hence he has no incentive to curb his myopically profitable harvest.

If there are only two, or a few, independently operated fleets, the results will be intermediate between these extremes. Each fleet will capture a portion of the future recruitments that result from his current harvest restraint, so each fleet will moderate its harvests to optimize the trade-off between current and future returns.

This picture is complicated in our model by the assumptions of split-stream harvesting sites, plus stochastic variability in the portion of recruitment that is allocated to each site. There are two major effects: first, a fleet's optimal harvest level, as it would have been had all recruitment been to a common site, may be unavailable when only a sub-stream portion of the total recruitment is accessible to the fleet. Second, uncertainty about how much recruitment will actually materialize at that site introduces uncertainty about the payoff to a specified harvest level, since harvest costs are high when the total sub-stream recruitment is low.

The issue here is how these added complications alter the trade-offs between present and future harvest payoffs. To answer this question we have recourse to simulation. The simulations in Figure 9.3a show payoffs resulting over a range of values of p_v for fixed $c_v = 0.1$. Four of the curves in the figure represent competitive situations, differing in the knowledge level of the fleets. The other four, (marked NBS) represent the corresponding cooperative Nash bargaining solutions. In all cases the long-run payoffs Π_v grow, more or less linearly, as landing price rises. This is perhaps to be expected, since with high compensation, stocks rebound quickly from a heavy harvest and then yield a good return from a harvest drawdown kept away from the break-even stock level (which falls as p grows).

It also seems reasonable that, among the two symmetric-information competitive curves, the highest returns occur (labelled on the graph as 'cur', for current) when both fleets know with precision the current realization of θ_α and $\theta_\beta = 1 - \theta_\alpha$. As in a deterministic model, each fleet knows accurately the immediate return from its current harvest. By contrast, when the fleets possess only 'min' knowledge, even this current status is unknown.

The simulations show that, in this high-compensation case, providing both fleets with equal additional knowledge has a positive value, even when they compete. Thus the payoffs are always lower, as shown by the line labelled 'Min' on the graph, where the fleets know only the steady-state probability distribution of θ_v than when both have full current knowledge.

The other two curves represent competitive payoffs to fleets α and β, respectively, in the asymmetric situation ('cur-min') where the α-fleet knows current values of θ_v but the β-fleet has only minimal knowledge. In that case, the α-fleet does better than when both fleets have current knowledge and the β-fleet does worse than when both have minimal knowledge. Plainly, it is not advantageous for the α-fleet to reveal its private information to the β-fleet.

The Transferable-Utility Nash Bargaining Solutions shown in the figure rely on the two fleets sharing their information, adopting a common objective which will maximize the total return to the fleet, and then transferring utility to share the total return in a way that respects the relative bargaining strengths of the players. Note that the NBS always improves the situation of both fleets relative to competition.

Critical depensation (Figure 9.3d)

Consider next a deterministic stock-recruitment function that exhibits critical depensation, so that $R^+ = F(S) > S$ over a range $0 \leq S \leq S_{crit}$ on the growth curve. Indeed, once having entered this region of critical depensation the stock cannot recover and, even in the absence of harvest, will ultimately become extinct. This hazard remains unchanged, when there is a stochastic split-stream.

Thus it is highly desirable to avoid over-fishing that will draw down the stock into a depensatory region. Note that intensive harvesting is dangerous as soon as total escapement S falls into the depensatory region. Hence each fleet, in choosing its harvest level, must judge its competitor's escapement as well as its own. On the other hand, so long as the sub-stream stock level exceeds c/p, a fleet's sub-stream harvesting will be immediately profitable, even if it helps lead to the subsequent escapement falling into the depensatory region.

Figure 9.3d shows the result of competitive harvesting when the stock-recruitment function shows critical depensation. Here, as landings prices rise, payoffs initially grow, but then drop and finally become zero, as the stock is driven to extinction. The immediate harvest return grows with p, but is not fully offset by increases in future returns (since these are captured in part by the competing fleet). And with p sufficiently large it will become optimal to drive the stock down into the depensatory region, leading to extinction. What is most notable here is that this effect is strongest when the fleets are most knowledgeable! – that is, the worst case is 'cur', when both fleets always know the true current realization of θ_v. With only minimal knowledge the fleets cannot respond with precision to high prices, since the two values of θ_v are equally likely. On the other hand, if only one fleet knows θ_v with precision, it will have a major advantage over its competitor, though ultimately the total escapement will fall into the critical region.

Figures 9.3b and 9.3c demonstrate the transition from strong compensation to critical depensation. Note that in the intermediate cases the stock is drawn down close to the break-even harvest level (which drops as p increases), but this does not lead to actual extinction. Notably, these dramatic features do not extend to the corresponding cooperative solutions.

Note that the case simulated is one in which the stock-split oscillates wildly between sub-streams, with 90 per cent of total recruited stock always in one or the other sub-stream. In a current information cooperative situation, this permits the fleets to split the harvest optimally, taking maximum advantage of the lower cost of harvest in that sub-stream where the stock is large. With minimal information, these adjustments cannot be made with such complete precision.

Measurement Precision

In Figure 9.4 we look in another way at this case of strong fluctuations in θ_v (alternating between $\theta_v = 0.1$ and 0.9). Now we fix on an intermediate c/p ratio of 0.2 (which corresponds to $p = 0.5$ in Figure 9.3a), and vary the information levels continuously between the extremes of 'cur' and 'min'. In particular the information level tagged as 'meas' coincides with 'cur' at the right side of the diagram (where measurement precision = 1), and coincides with 'min' at the left side (where measurement precision = 0).

Figure 9.4b (where growth is critically depensatory) illustrate that, with symmetric knowledge, both players will be better off when information levels are low than when they are high. Even with compensatory growth (Figure 9.4a), both symmetrically-endowed fleets will be somewhat better off at moderate than at high levels of measurement precision. Intuitively, common-property competition is destructive, especially under conditions of depensatory growth. It becomes more destructive when all players are provided with precise information to sharpen their competition.

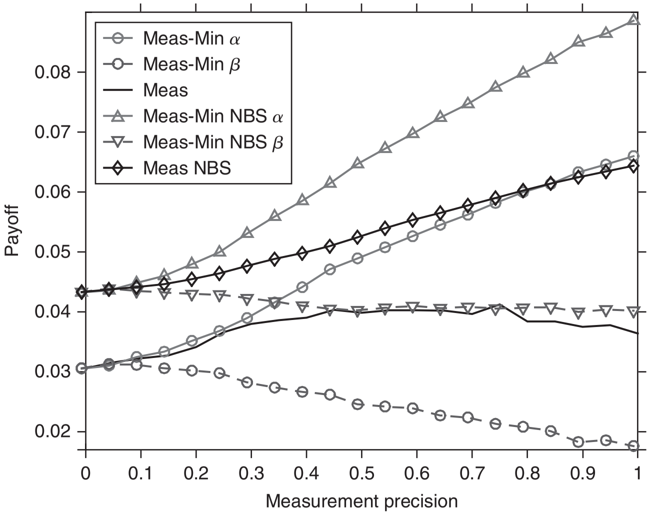


Figure 9.4a *Compensation*

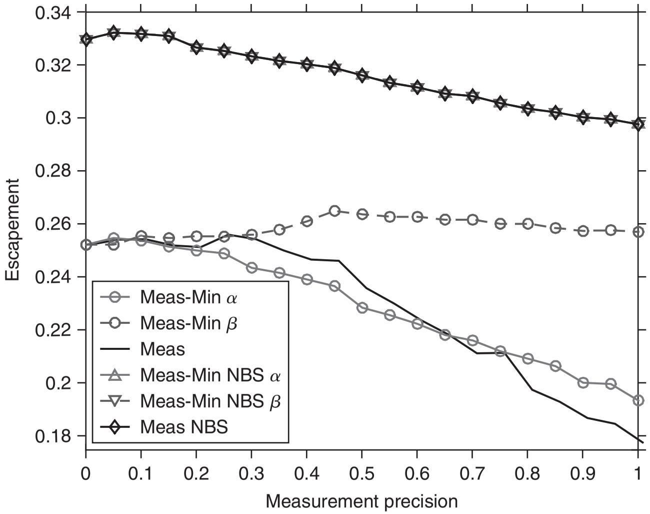


Figure 9.4 *Harvesting with information of varying quality*

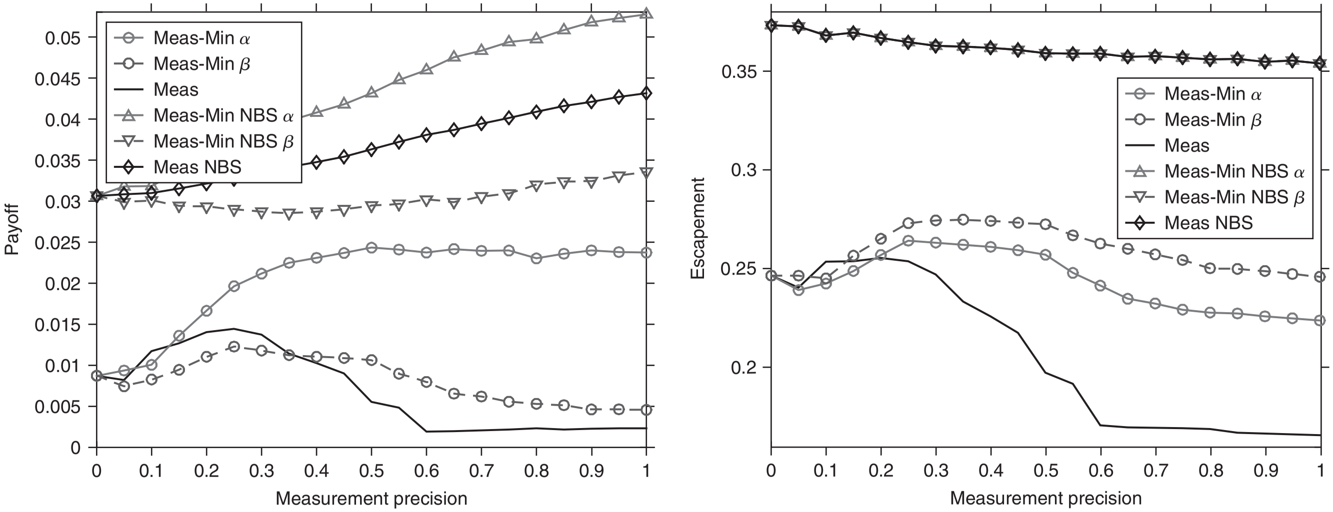


Figure 9.4b *Critical depensation*

Of course this is no longer true when the players agree to a cooperative bargaining resolution of the game. In that case, making information transparent insures a better outcome for both, while acknowledging the stronger bargaining position of the fleet which is asked to reveal its private information.

Asymmetry in the Oceanic Environment

Next we drop the assumption that the fleets' situations are symmetric, other than in their information status. We retain the *c*/*p* ratio of 0.5, but now vary the mean value of θ_α between the extremes of 0.8 (strongly favouring the α-fleet) and 0.2 (strongly favouring the β-fleet). The standard deviation of θ_v is taken to be 0.2, so that when mean $[\theta_v] = 0.8$ then θ_v takes on values 0.6 and 1.0 with equal likelihood. Thus the range of fluctuations of θ_v is now much smaller than in the previous simulations.

Nevertheless the simulations here show features quite similar to those seen before. For example, with competition, when the environment favours a particular fleet then symmetrical minimal knowledge produces a better outcome for that player than symmetrical current knowledge, but this is reversed when the environment favours its competitor. In the 'cur-min' case the payoff to the α-fleet is always better than either 'min' or 'cur', and the payoff to the β-fleet is usually worse.

The Pareto Boundary

Until now we have displayed the Nash Bargaining Solution with little comment. With transparent utility, as we have been assuming, and with cooperation entailing making any private information public, the Pareto boundary on which NBS is based will be symmetric whenever the positions of the fleets are symmetric except for asymmetries in information. In these cases, the NBS fleet payoff proportions closely reflect those of the corresponding competitive payoffs (that is those composing the 'threat point'). This applies in particular for most of our simulated cases, for example those in Figure 9.4, involving measurement precision. The corresponding Pareto diagram is displayed in Figure 9.6.

However when there are fleet asymmetries other than in information state, the Pareto Boundary itself will be asymmetric. This is true when mean$[\theta_v]$ is asymmetric between $v = \alpha$ and β, as in Figure 9.5. Examples of the determination of the NBS in this situation are shown in Figure 9.7.

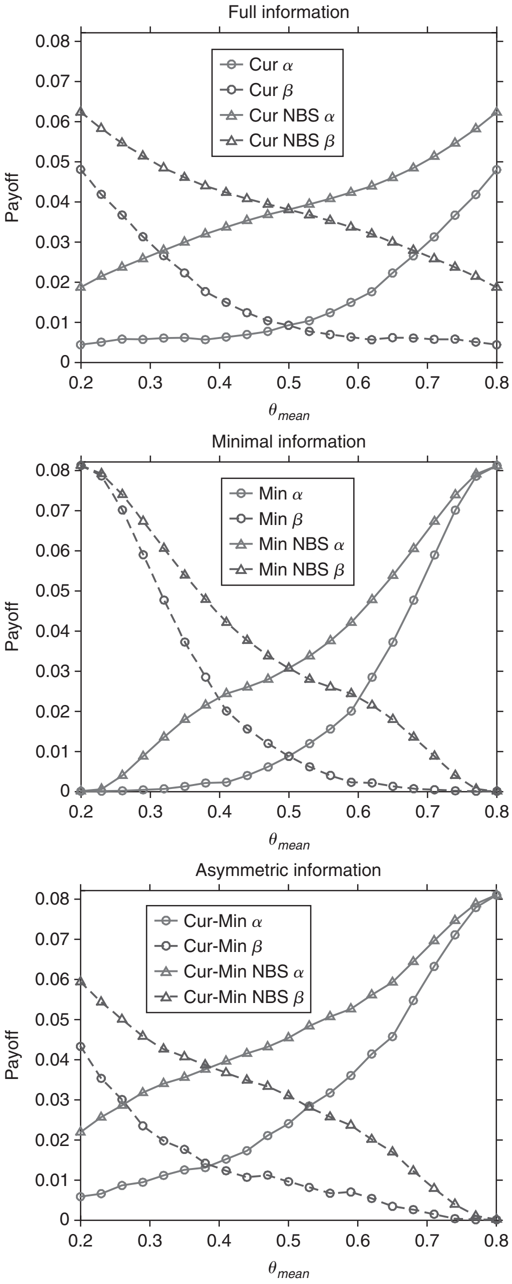


Figure 9.5 Variable environmental asymmetry (mean of θ*)*

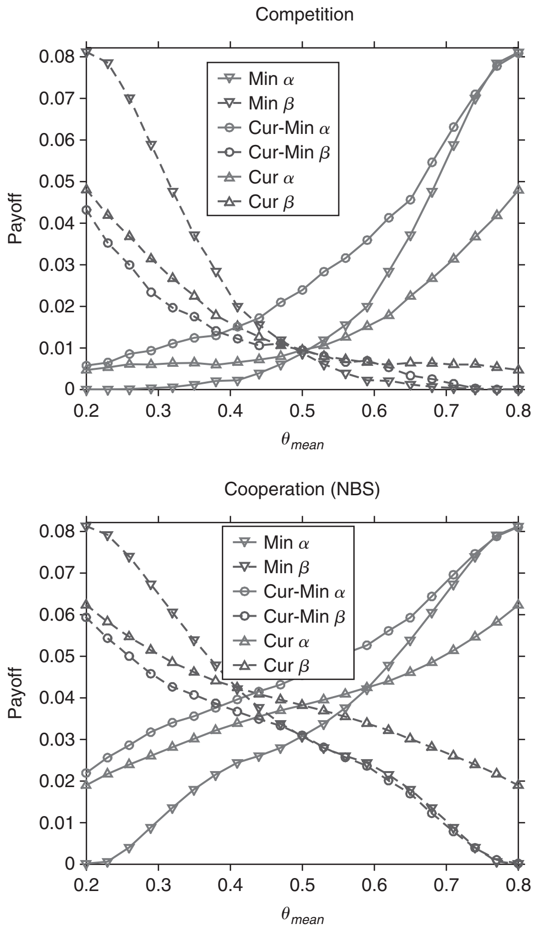


Figure 9.5 (continued)

FURTHER ADVANCES AND FUTURE GOALS

The split-stream model has been developed further in several significant directions. Much of this work is completed, and will be available in 2005.

First is the joint work of McKelvey and Golubtsov (forthcoming) which, while retaining the restricted information structures illustrated by simulations described in the present chapter, has been implemented in a computer model of considerable generality as regards biological growth assumptions and economic fleet objective functions, going well beyond the examples shown here.

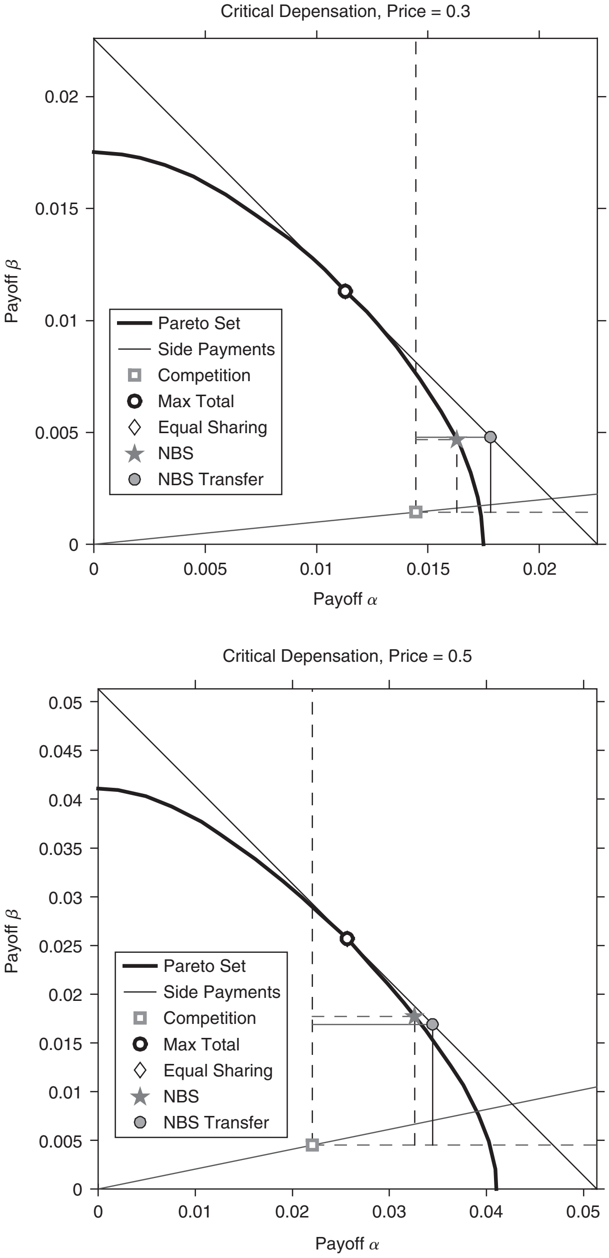
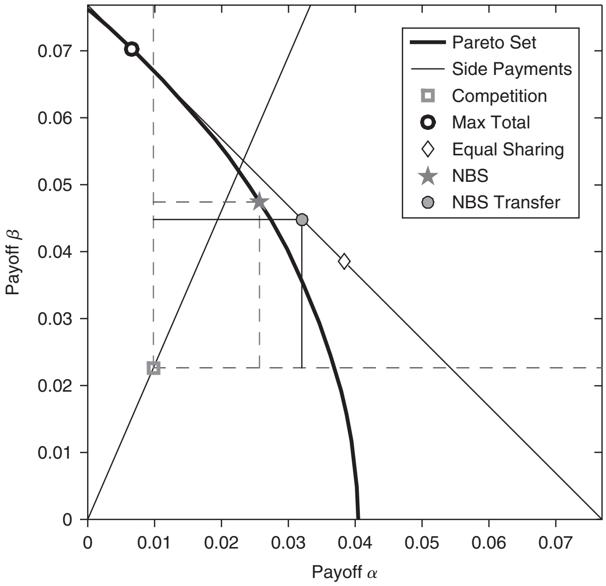


Figure 9.6 Pareto set and NBS for various price values (critical depensation)

Critical Depensation, $\theta_{mean} = 0.3$



Critical Depensation, $\theta_{mean} = 0.7$

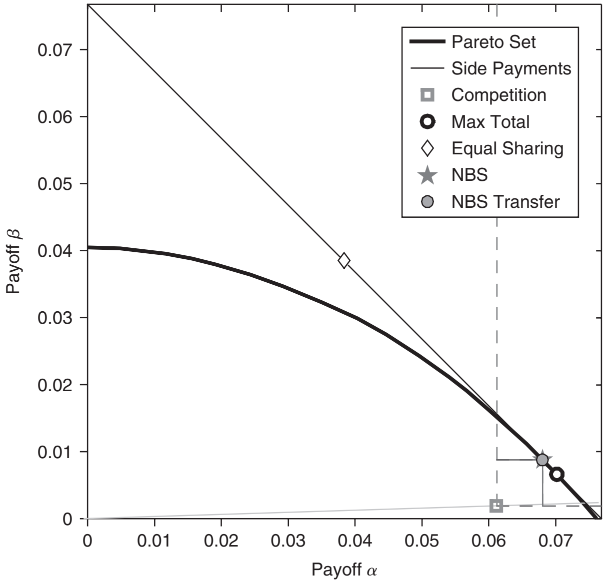


Figure 9.7 Pareto set and NBS for various θ_{mean} *values*

Second is the work of McKelvey, Miller and Golubtsov (2003), applying the McKelvey-Golubtsov model to the long-running disputes between Canada and the United States over joint management of the North American Pacific Salmon Fishery.

Perhaps most basic are model extensions, intended to achieve greater flexibility to capture the specific characteristics of particular multilateral trans-boundary marine fisheries. An initial move in this direction has been to elaborate the information structures that can be simulated (McKelvey and Cripe, submitted).

We refer again to the life-cycle diagram for the split-stream:

$$\begin{tikzcd}
 & & R_\alpha = \theta_\alpha R \arrow[r] & S_\alpha \arrow[dr] & & \\
S^- \arrow[r] & R = F(S^-, \mathrm{b}^-) \arrow[ur] \arrow[dr] & & & S = S_\alpha + S_\beta \arrow[r] & R^+ \\
 & & R_\beta = \theta_\beta R \arrow[r] & S_\beta \arrow[ur] & &
\end{tikzcd}$$

We assume that, prior to the decisions which must be made to harvest the current R_α and R_β, S^- will be known to the fleets, but that, in general, R will not be. Furthermore, both the switching parameter θ_v and the growth parameter b form random Markov chains, neither of which is observed directly by the fleet managers. Here we will briefly sketch out the nature of these assumed information structures, for imperfect measurement of the random variable $b(t)$. Analogous structures are assumed for measuring θ_v (t).

Concerning b, each v-fleet observes a random measurement variable b_v which takes on the same realized values as b, but with observation error. The probabilistic characterization of the instruments' joint accuracy is expressed as

$$\mathrm{prob}[b_\alpha = b_j \,\mathrm{and}\, b_\beta = b_k \mid b = b_i],$$

for all b_i, b_j and b_k in the range of b. It is known to both fleet managers, and contains information on the accuracy of each fleet's observations as well as on the extent to which their measurements are correlated.

Restricting attention initially to a stochastic version of Levhari-Mirman's classical fish-war model (Levhari and Mirman, 1980), we have carried out simulations comparing the outcomes of private information games, transparent information games and the baseline case of full current information games. In this case we have been able to push the mathematical analysis far enough to obtain intuitively-meaningful closed-form expressions for the fleets' harvesting policies.

Another topic we are pursuing is gaining an understanding of the role of the fleets' attitudes toward risk in determining the outcome of a harvesting game. In the case of the generalized Levhari-Mirman model, both fleets are always assumed to be (logarithmically) risk-averse, and we have developed a useful new way of interpreting this feature and its implications. With more general risk-sensitive objectives, simulated results are sometimes counter-intuitive, especially when the fleets have differing attitudes toward risk. We continue to examine these questions in our ongoing studies.

Another channel for future research involves elaborating the network structure of the model, from the simple split-stream structure. In particular we intend to construct an incomplete-information stochastic version of our 'Hit-and-Run' game model, which can be used to simulate the so-called 'new-member problem', concerning negotiations between a multilateral Regional Fisheries Management Commission and a distant-water fleet wishing to enter the regional fishery. This bargaining process becomes particularly interesting when the Commission controls access to a major portion of the fishing grounds, so that without cooperative agreements the Distant Water Fleet (DWF) is confined to international waters of the high seas. Deterministic versions of the Hit-and-Run model have been published (in 2002 and 2003) by McKelvey, Sandal and Steinshamn.

Finally, we wish to apply appropriate versions of our incomplete-information stochastic harvest games to a variety of marine fisheries operating across international boundaries. It seems clear that the outcomes of such harvesting games will depend heavily on particular circumstances in the fisheries involved. These differences will sometimes relate to the cyclic patterns and intensities of oceanic environmental conditions, sometimes to the biological characteristics of the harvested fish stock or stocks, and usually upon the economic interests of the nations involved in the fishery, either as harvesters or as countries exercising control over their coastal waters.

Plainly, a 'one size fits all approach' will not be adequate here – the models must be adapted to particular circumstances. On the other hand, the models we are building will remain, as they must, as highly stylized abstractions from reality: beyond the usual abstractions met in bioeconomic models, non-cooperative game models must make even more heroic assumptions about human aspirations and behaviour. Their role, then, is not prescriptive in the physical science mode. Rather, they must remain merely suggestible, as a window into an artificial world – one which, we hope, may in some ways resemble our own.

ACKNOWLEDGEMENTS

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10. Prototype of an integrated model of the worldwide system of small pelagic fisheries

Christian Mullon and Pierre Fréon

INTRODUCTION: THE WORLDWIDE SYSTEM OF SMALL PELAGIC FISHERIES

We consider the 'worldwide system of pelagic fisheries' as the main pelagic fish resources, the main pelagic fisheries, the main markets for the products (canned fish, fishmeal, fish oil and fresh fish), the human factors (fishers, managers, fishery biologists, conservationists, and so on) and the bioeconomic and political interactions of these components (Figure 10.1). Marine fisheries exploiting small pelagic fish (for example anchovy, sardine, herring) produced some 34 million tonnes per year over the period 2000–2003, 53 per cent of the world's marine fish catch (excluding molluscs, crustaceans and elasmobranches). Pelagic fisheries are found in all oceans, mainly on the East coasts of continents and often related to upwelling processes (Table 10.1). The catch is used to produce fishmeal, canned fish, fish oil, fresh fish and smoked fish (Table 10.2). Important changes are anticipated for the fishmeal and oil markets during the present decade (Table 10.3).

Catches of anchovy and sardine (local, regional and global) are highly variable and prone to massive peaks and troughs (Csirke, 1988; Fréon and Misund, 1999; Schwartzlose *et al.*, 1999). Recent analyses of small pelagic fisheries have generally concluded that neither fishing pressure nor demand for fish products should increase (FAO, 2002), and have highlighted the inherent instability of the pelagic system. More specifically, small pelagic fisheries worldwide have been characterized by (i) overcapacity in many fleets, one of the major issues in world fisheries management (Gréboval, 1999; Lindebo, 1999), (ii) changes in the destination of catch product, particularly following the development of aquaculture and its demand for fishmeal (Holmes, 1996; Durand, 1998; Rosamond *et al.*, 2000), (iii) lack of knowledge of the effects of climate change on the dynamics of the populations (DeAngelis and Cushman, 1990; Bakun and Weeks, 2004), and

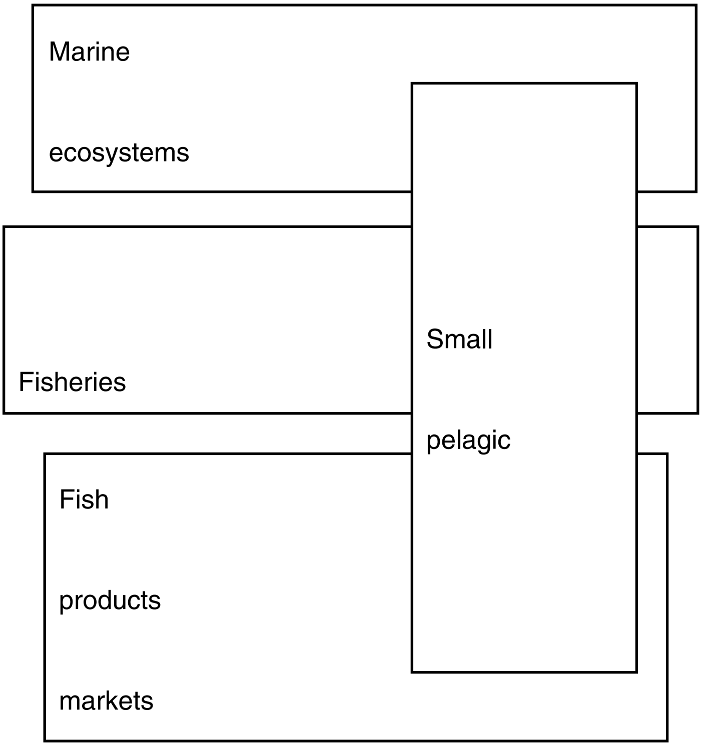


Figure 10.1 The system of small pelagic fisheries as it interacts with other ecological and economic systems

(iv) great discrepancies, mainly in management systems, between developed countries and developing countries, mainly at the level of small-scale fisheries (Garcia and de Leiva Moreno, 2003). Particular and urgent attention needs to be given to Chinese fisheries, which are important, growing fast (Figure 10.2), and poorly known (Watson and Pauly, 2001). The future of pelagic fisheries is clearly difficult to predict, though certain events or cascades of events can be foreseen.

Owing to the effects of global climate change, the dynamics of exploited marine ecosystems are becoming more unstable, some stocks are collapsing, supply to the markets that rely on them are drying up, and there is increasing pressure on other stocks, making them in turn less resilient. Such a scenario was observed when Californian sardine collapsed, pelagic fishing pressure being removed first to the Mexican ecosystem, then to the Peruvian ecosystem, both of which weakened as a result (Troadec *et al.*, 1980; Cisneros-Mata *et al.*, 1995).

Table 10.1 World catches of small pelagic fish in 2000

Country	Production (1000 tons)	(%)
Peru	7637	39.28
Chile	2317	11.92
Japan	1441	7.41
United States of America	1036	5.33
China	828	4.26
Norway	648	3.33
Russian Federation	605	3.11
Indonesia	475	2.44
Morocco	465	2.39
Denmark	397	2.04
Philippines	355	1.83
Thailand	348	1.79
Mexico	334	1.72
South Africa	325	1.67
India	322	1.66
Turkey	281	1.45
Sweden	279	1.43
Korea, Republic of	259	1.33
Canada	232	1.19
Spain	232	1.19
Senegal	230	1.18
Iceland	217	1.12
Ghana	181	0.93

Source: FISHSTAT (FAO, 2002).

Table 10.2 Use of small pelagic fish catches in 2000

	Production	(%)
Canned fish	411 491	24
Dried, salted or smoked fish	149 368	8
Fresh, chilled or frozen fish	447 542	26
Fishmeals	396 580	23
Fish oils	327 269	19
Total	1 732 250	100

Note: Percentages add up to more than 100% due to rounding.

Source: FISHSTAT (FAO, 2002).

Table 10.3 Destination of fishmeal and fish oil in 2002 and projection for 2010

	Fish meal 2002 (%)	Fish meal 2010 (%)
Aquaculture	34	48
Poultry	27	15
Pigs	29	22
Ruminants	1	0
Others	9	15
	Fish oil 2002 (%)	Fish oil 2010 (%)
Aquaculture feed industry	56	79
Industrial	12	5
Edible	30	14
Pharmaceutical	2	2

Source: International Fishmeal and Fish Oil Organization (IFFO).

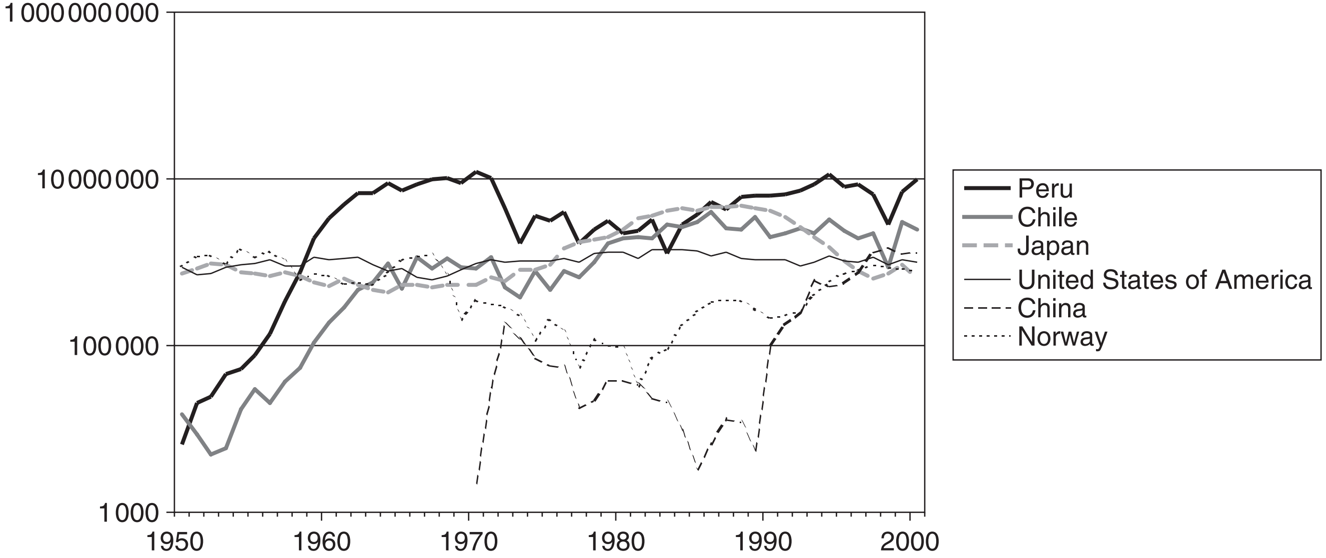
Global climate change results in a latitudinal shift in ocean temperature (Bakun, 1990; Mendelssohn and Schwing, 2002; Mote and Mantua, 2002; Snyder *et al.*, 2003; Diffenbaugh *et al.*, 2004), then in a corresponding latitudinal drift of the stocks, and consequently in fishing rights failing to adapt to the new biological situation. Some developed countries manage to reduce their national fishing effort as required to enhance the principle of sustainability, but exert increased political pressure for fishing rights in developing countries. At the same time, the development of fishing capacity in emerging countries can proceed uncontrolled.

The development of aquaculture resulted in a huge increase in the demand for fishmeal, while consumer preference for feeding poultry on soya meal rather than fishmeal resulted in a virtual collapse of the market for fishmeal in the poultry industry.

Developing demand for some pelagic fish product (for example for fish oil Omega 3) competes with demand for fishmeal; in contrast, development of new demand (for example for surimi) can result in the development of a specific fishery to feed the demand (Alaska pollock).

The globalization of trade resulted in the uncontrolled opening of the world's fisheries; perhaps now the globalization process has ended, resulting in curtailment of fishing rights given to foreign fleets by developing countries.

These uncertainties have led and will continue to lead to increasing negotiations and conflict, and highlight the need for tools to be applied to consensus building. Collectively, we must find a way to predict the effects of the



Note: This figure shows the distribution of yields, with one fishery (Peru) producing half the world's catch (Note the log scale for the y-axis), the high variability of pelagic catches, and the emergence of Chinese fisheries during the past 10 years.

Figure 10.2 World production of anchovies and sardines in the most important national fisheries variability in small pelagic fish stocks attributable to climate change, within the current context of the economic globalization that makes the fisheries of the world interdependent. Tools are required to unify the views of stakeholders and decision makers (fisheries, fish product consumers, politicians, conservationists, scientists), to open dialogue (to address specific questions and to develop appropriate concepts), and to develop relevant hypotheses (from the knowledge of all the stakeholders). These needs are key to the sustainable management of fisheries (World Bank, 2004).

MODELLING PRINCIPLES

With the objective of providing such tools and concepts, and dedicated to the global management of small pelagic fisheries, an integrated model of the worldwide system of small pelagic fisheries is being designed. Of course, building a fully predictive model of such a complicated and open system is not possible. To support discussion and negotiation, mainly through role-playing game sessions, the model has to be (i) realistic, (ii) able to reproduce typical past events and (iii) sensitive to parameterization. It should also allow consideration of the consequences of various hypotheses, at least in terms of trends and directions.

The approach to building the model is participative and step-by-step, and aims to involve stakeholders at every step: definitions of goals, entities and processes, assessment of results, and ideas for improvements. The first step has been to build a prototype of the model that runs with approximate data. This has allowed us to make explicit the components of the model, to explore which databases can support the model (in terms also of parameterization and validation), to show its technical feasibility from a computing point of view, and to discuss the theoretical background.

Pauly *et al.* (2000) and Watson *et al.* (2004) made the point that 'mapping marine fisheries onto marine ecosystems' represents possibly the most efficient tool for consensus building. Therefore, the computer interface of the model is designed to produce 'kinetic maps' as a representation of the dynamics of a system, specifically of changes or shifts. An advantage of this 'geographic' approach is that it implies explicit definition of entities represented as (1) a trade-off between extension and resolution: only few ecosystems or fisheries or markets can be mapped together on a global map, (2) a trade-off between appropriateness of the model structure and the availability of data: existing data are based on specific typologies, defining entities, so are not the most adequate to represent the dynamics.

We have selected national or regional fisheries, FAO marine areas (rather than Large Marine Ecosystems), and national or regional markets for fish products. Because it allows us to stress the process of communicating scientific results to stakeholders, the model is designed to support role-playing game sessions (Kagel and Roth, 1995; Duffy, 2001; Barreteau *et al.*, 2003) that group together several stakeholders involved in the management of a complex marine area with students attending courses in environmental management. Model simulations are used to provide the framework of the play and to make explicit the consequences of players' decisions. Role-playing games efficiently reveal the behaviour and the motivational drivers of stakeholders.

Bioeconomic models provide a simple and efficient way to display the dynamics of renewable resources (Clark, 1990). They relate biological variables, such as productivity or carrying capacity, to economic variables, such as the social rate of discount. Practically, they lead to aggregated models, which are currently not particularly relevant within the context of the global management of fisheries. In contrast, disaggregated supply-demand models focus on how equilibrium occurs in several interlinked markets, where economic agents maximize profits; see, for example, Dey *et al.* (2003) and Briones *et al.* (2005) for applications in the context of fisheries, and the FISH2020 model, which provides projections of the state of world fisheries until 2020 (Delgado *et al.*, 2002). These models work as follows:

- quantitative characteristics of supply (for ecosystems) and demand (for markets) functions are assessed from time-series of data;
- general hypotheses are set about the evolution of supply (for example, changes in the productivity of ecosystems) and demand (for example, increasing or decreasing pressure on specific markets);
- finally, for each simulated year, a global equilibrium is computed on the ecosystems and the markets, to yield detailed projections of fishing effort, production, prices and income.

These are models of behaviour, not of strategy. This approach allows us to consider non-individual entities, such as national fisheries, as agents, and to adapt them to help us describe a complex dynamic system.

Supply-demand models are mostly static; they do not address the ecological, economic and bioeconomic feedbacks of the systems. They therefore cannot show the impact of any resulting equilibrium (fishing effort, production or prices) on the evolution of stocks or markets. With a similar disaggregated approach, computable equilibrium models (Shoven and Whalley, 1992; Floros and Failler, 2004) close the macroeconomic loop, relating production, consumption, investment and savings in a dynamic perspective. However, their relevance to modelling a specific subsystem such as that of small pelagic fisheries is not obvious; being quite complicated, they can easily lead to neglecting characteristics that are perhaps specific to pelagic fisheries systems, for example their instability.

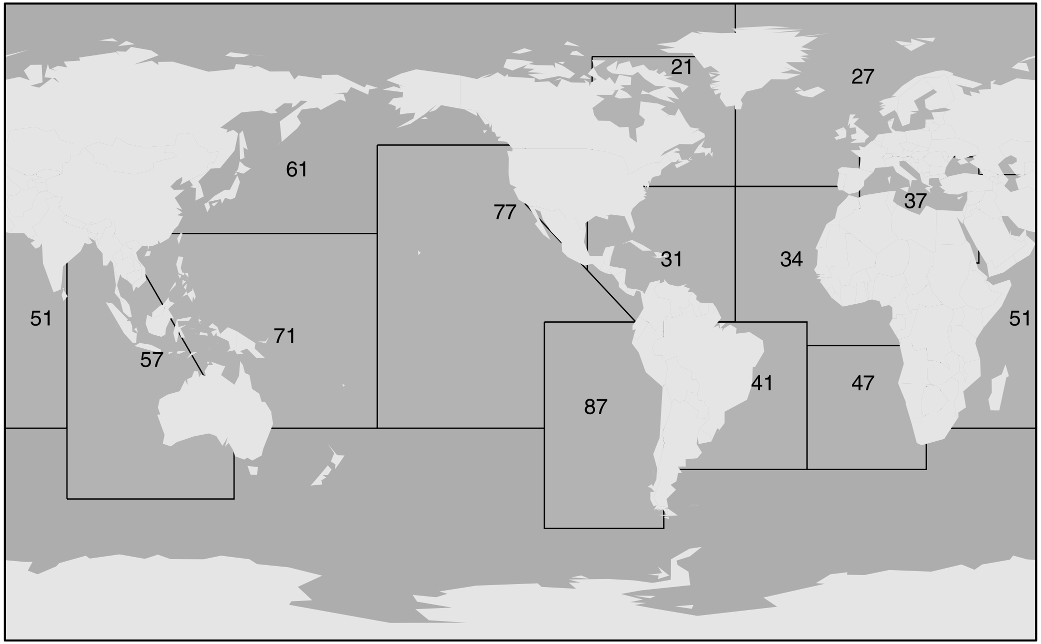
To take into account biological and economic feedback, our model of the worldwide system of pelagic fisheries is one of supply and demand, integrating worldwide pelagic stocks, small pelagic fisheries and markets for fish products. Inter-temporal dynamics are represented by simple deterministic equations that describe how pelagic stocks evolve, the behaviour of fisheries and the demand on the markets for fish product.

MODELLING CHOICES

Entities, Scales and Mechanisms

On one hand, the model must be spatially disaggregated; on the other hand a coherent set of data is needed to calibrate and validate the model; as a resulted of taking this dilemma into account, a model of intermediate complexity has been designed, involving less than 100 entities. The prototype integrates the behaviour of the following entities, all defined in the FAO database, FISHSTAT (FAO, 2004):

- 13 marine areas: Eastern Central Atlantic, Northeast Atlantic, Northwest Atlantic, Southeast Atlantic, Southwest Atlantic, Western Central Atlantic, East Indian Ocean, West Indian Ocean, Mediterranean Sea, Eastern Central Pacific, Northwest Pacific, Southeast Pacific, Western Central Pacific (Figure 10.3);
- 15 national and regional fisheries: Central America, China, Mediterranean, North Africa, North America, Northeast Asia, North Europe, Russia, Southeast Africa, Southeast America, Southeast Asia, South Europe, Southwest America, Southwest Asia, West Central Africa;
- 40 markets for fish products: Central America (canned, fresh), China (fresh, meal, other), Mediterranean (canned, fresh), North Africa (canned, fresh), Northeast Asia (canned, other), North Europe (canned, fresh, meal, oil), North America (meal, other), Russia (canned, fresh), Southeast Africa (canned, fresh, meal, other), Southeast America (canned, meal, oil, other), Southeast Asia (canned, other), South Europe (canned, fresh), Southwest America (canned), Southwest Asia (canned, meal, other), West Central Africa (fresh, other). Here, Fresh depicts fresh, chilled or frozen fish, and Other depicts dried, salted or smoked fish.



Notes: The marine areas are Eastern Central Atlantic (34), Northeast Atlantic (27), Northwest Atlantic (21) Southeast Atlantic (47), Southwest Atlantic (41), Western Central Atlantic (31), East Indian Ocean (57), West Indian Ocean (51), Mediterranean Sea (37), Eastern Central Pacific (77), Northwest Pacific (61), Southeast Pacific (87) and Western Central Pacific (71). Those not defined are not used in the model.

Source: ftp://ftp.fao.org/fi/stat/by_FishArea/Default.htm.

Figure 10.3 FAO marine areas used in the model

Table 10.4 The mean annual production of FAO marine areas during 1991–2000

Area	Mean production of small pelagic fish	%
Atlantic EC	1 504 606	9.45
Atlantic NE	722 684	4.54
Atlantic SE	394 924	2.48
Atlantic SW	89 527	0.56
Atlantic WC	123 192	0.77
Indian E	245 382	1.54
Indian W	287 989	1.81
Mediterranean	659 309	4.14
Pacific EC	441 884	2.77
Pacific NW	2 480 930	15.58
Pacific SE	7 981 059	50.11
Pacific WC	997 019	6.26

Source: FISHSTAT (FAO, 2002).

- One super-species grouping of pelagic species, that is, the species aggregated in the category 'Herrings, Sardines and Anchovies' of FISHSTAT.

The model simulations provide results every year for 15 years. This duration may be debated from a biological or an economic point of view, but from a management perspective, it is straightforward and allows simplification at all levels. Catches by area are listed in Table 10.4.

General Principles

The model integrates (i) the dynamic processes, that is, biological (population dynamics) and economic (evolution of investment, activity, demand), and (ii) behavioural processes, that is, fisheries behaviour (distribution of effort in several marine areas and the yield in several markets). At each time-step:

- The states of marine areas, fisheries and markets evolve according to deterministic rules;
- The behaviour of fisheries is related to how they select marine areas in which to fish and markets in which to sell. The result of the equilibrium between supply and demand is a consequence of their competition;

Table 10.5 Variables of the model

Variable	Denomination	Unit
X_e	Stock	Tons
q_e	Fishing efficiency	1/(Boat × Ton)
P_{ef}	Access costs to marine areas	Euro/Boat
Q_{fm}	Access costs to market	Euro/Ton
E_f	Number of boats	Boat
A_m	Demand price (intercept)	Euro/Ton
B_m	Demand price (slope)	Euro/(Ton × Ton)
τ_{ef}	Repartition of effort	No unit
μ_{fm}	Distribution of product	No unit
R_f	Income	Euro
Y_f	Yield	Ton

- Biological dynamics are governed by a conventional production function, whose parameters may depend on climate.

The resulting model is quite simple; all variables are defined in Table 10.5.

Representation of Marine Areas

Marine area dynamics are represented through the conventional formalism of production models (Beverton and Holt, 1957; Hilborn and Walters, 1992). A marine area e is characterized by a stock X_e and a fishing efficiency q_e. A fishery f applying effort E_f on this marine area obtains a yield $Y_{ef} = E_f q_e X_e$. The total production from this marine area is therefore $Y_e = \Sigma_f Y_{ef}$. A stock of a marine area evolves according to a conventional production model:

$$X_e(t+1) = X_e(t) + R_e(X_e(t)) - Y_e(t) \tag{10.1}$$

Production functions are logistic:

$$R_e(X) = r_e X\left[1 - \frac{X}{K_e}\right]$$

where K_e is the carrying capacity and r_e the natural rate of renewal.

Representation of the Markets for Fish Products

There are many approaches, both theoretical and practical, to show the behaviour of fish product markets, focusing on both elasticity of prices/

quantities and the supply/demand relationship (Asche and Bjørndal, 1999; Tacon, 2001; Tvetaras *et al.*, 2002). In the model, markets for fish products are represented by a simple demand function. Let Y_{fm} be the product sent by fishery f to market m. Then, at market m, the supply is $Y_m = \Sigma_f Y_{fm}$. Prices are related to supply by the functional equation $P_m = V_m(Y_m)$. Currently this equation is linear: $V_m(Y_m) = A_m - C_m Y_m$, where A_m and C_m are the parameters intercept and slope respectively, intercept being related to the demand, and slope to elasticity. The evolution of the demand function of a fishery depends on global economic trends, and is expressed through time-dependent functions: $A_m(t)$, $C_m(t)$.

Representation of Fisheries

A fishery f is determined by (i) its fishing capacity (the number of 'standardized' boats), E_f, (ii) its access costs (per 'standardized' boat) to the different marine areas, P_{ef}, and (iii) its access costs (per unit sold) to the different markets for fish products, Q_{fm}. Access costs to ecosystems are the sum of transport costs (fuel) and royalties, and to markets the sum of transport costs and importation taxes.

Each year, fishery f selects its own strategy $S_f = \{\tau_{ef}, \mu_{fm}\}$, where τ_{ef} is the distribution of its effort among the marine areas it is permitted to access and μ_{fm} is the distribution of its yield among markets for fish products (of course, $\Sigma_e \tau_{ef} = 1$ and $\Sigma_m \mu_{fm} = 1$). Its yield from marine area e is $Y_{ef} = E_{ef} q_e X_e = E_f \tau_{ef} q_e X_e$, and its total yield is

$$Y_f = \sum_e Y_{ef} = E_f \sum_e \tau_{ef} q_e X_e.$$

To market m, fishery f sends

$$Y_{fm} = \mu_{fm} Y_f = \mu_{fm} E_f \sum_e \tau_{ef} q_e X_e,$$

and it receives $Y_{fm} V_m(Y_m)$. The income of a fishery f with strategy $S_f = \{\tau_{ef}, \mu_{fm}\}$ is equal to its sales $\Sigma_m Y_{fm} V_m(Y_m)$ minus its transportation $\Sigma_m Q_{fm} Y_{fm}$ and exploitation costs $\Sigma_e E_f \tau_{ef} P_{ef}$, that is:

$$R_f = \sum_m Y_{fm} V_m(Y_m) - \sum_m Q_{fm} Y_{fm} - \sum_e E_f \tau_{ef} P_{ef} \qquad (10.2)$$

If $\tilde{Y}_{fm} = Y_m - Y_{fm}$ represents the sales of other fisheries at market m, this results in:

$$R_f = \sum_{me} \left[V_m \left(E_f \sum\nolimits_e \mu_{fm} \tau_{ef} q_e X_e + \bar{Y}_{fm} \right) - Q_{fm} \right] \mu_{fm} \tau_{ef} q_e X_e E_f - \sum_e E_f \tau_{ef} P_{ef} \tag{10.3}$$

Modelling fisheries investment behaviour assumes a relationship between fishing capacity and income. For the current implementation, this is: $E_f(t + 1) = \kappa E_f(t) + \lambda R_f(t)$, where κ and λ are parameters that reflect the depreciation of capital and the portion of income reinvested, respectively. Coefficient κ is set at 0.95, but λ can be set by the user.

Obtaining the Competitive Equilibrium

The income generated by a fishery depends on both its own strategy as well as the strategies of other fisheries. The system is therefore a competitive game (for example Mueller, 1997) in which one may be interested in its non-cooperative or Nash's equilibrium, that is, the sets of choices by all fisheries and strategies, in such a manner that a given fishery cannot change its strategy unilaterally without diminishing its income.

There are theoretical and computing difficulties in the equilibrium model. The author will supply on request an algorithm which results in a Nash's equilibrium for the above income functions. The other, deterministic, part is simple and can easily be implemented with such tools as Stella, or even Excel.

PARAMETERS

The model has been designed to simulate scenarios that result from various hypotheses concerning the future of marine areas (for example their productivity, in relationship to climate change), the future of fisheries (for example their investment behaviour), and the future of the markets for fish products (for example demand). In the present implementation of the model, simulations are based on the parameters listed in the following sections.

Marine Areas

- Changes in carrying capacity. Coefficient υ represents a continuous (constant rate) increase or decrease in carrying capacity for all marine areas: $K_e(t + 1) = [1 + \upsilon_e(t)]K_e(t)$.

- Changes in renewal rate. Coefficient η represents a continuous (constant rate) increase or decrease in renewal rate for all marine areas: $r_e(t+1) = [1 + \eta_e(t)]r_e(t)$.
- Fishing efficiency changes. Coefficient ϑ represents a continuous (constant rate of) increase in fishing efficiency for all marine areas attributable to technological improvements: $q_e(t+1) = [1 + \vartheta_e(t)]q_e(t)$.
- Latitudinal climate change. Coefficient ω represents how the carrying capacity of marine areas changes according to latitude: $K_e(t+1) = [1 + \omega_e(t)][lat - 30°]K_e(t)$.
- Recruitment variability. Coefficient σ represents the randomness of the recruitment function. All stocks of marine areas evolve according to the formulation

  $$X_e(t+1) = [X_e(t+1) + R_e(X_e(t+1)) - Y_e(t)][1 + \theta_e(t)],$$

  where θ is a random number normally distributed with mean 0 and variance σ^2.

Fisheries

- Changes in fishing capacity. This is the portion of income that is reinvested, coefficient λ in the formula $E_f(t+1) = \mu E_f(t) + \lambda_f(t)R_f(t)$. Coefficient μ, representing depreciation of capital, is set to 0.95.
- Compliance. This takes into account how quotas are respected. Compliance by fisheries is one of the principal issues in management of any fisheries sector, so it is specially important to reflect the differences in means of enforcement between developing, emerging and developed countries.
- Changes in flexibility. A differential parameter that represents how fisheries adapt to new strategies.

Markets

- Demand function changes (intercept). In the formula $V_m(Y_m) = A_m - C_m Y_m$, this shows how parameter A_m evolves over time: $A_m(t+1) = [1 + \alpha_m(t)]A_m(t)$.
- Demand function changes (slope). In the formula $V_m(Y_m) = A_m - C_m Y_m$, this shows how parameter C_m evolves over time: $C_m(t+1) = [1 + \chi_m(t)]C_m(t)$.
- Growth of fish meal markets. In the formula $V_m(Y_m) = A_m - C_m Y_m$, this shows how parameter A_m evolves over time, in order to represent a specific increase (or decrease) in the demand for fishmeal in the market: $A_m(t+1) = [1 + \alpha_m(t)]\,[1 + \psi_m(t)]A_m(t)$.

Access to Marine Areas

- Changes in fishing rights. This parameter quantifies a uniform increase (or decrease) in exploitation costs: $P_{ef}(t+1) = [1 + \zeta_{ef}(t)]P_{ef}(t)$, and is used to take into account the trend in fuel prices or the evolution of royalties.

Access to Markets

- Changes of importation taxes. This parameter quantifies uniform increase (or decrease) of access costs: $Q_{fm}(t+1) = [1 + \zeta_{fm}(t)]Q_{fm}(t)$, and is used to take into account the effects of globalization.

SCENARIOS

A scenario involves setting the above parameters to given values. These values are the same for all steps of a given simulation, and are the same for all entities.

SENSITIVITY ANALYSIS

The model allows sensitivity analysis. Conventionally, such analysis consists of:

- Choosing a sensitivity parameter among those listed above;
- Fixing a minimum and a maximum value for that parameter;
- Fixing single values for all other parameters;
- Running the simulations for 11 values of the parameter between minimum and maximum values;
- Generating global views of the resulting dynamics.

ROLE-PLAYING GAME

A role-playing game needs a set of 12–20 players gathered around a table, with several computers between them, and a game leader to guide them. Players are:

- Representatives of fishing industries for a given economic area, that is, West Asia, East Asia, North America, South America, North Atlantic and South Atlantic, who set an investment behaviour for their principals, accepting or refusing quotas. Their goal is to ensure a positive annual income from the fisheries they represent;

- Representatives of fish product industries, for example canned fish, fishmeal, fish oil and transformed fish. They reorientate the demand function in the markets, so modifying the cost of access to markets. Their goal is to generate a sufficient supply of fish product from the markets each year;
- Representatives of conservation societies for the extended marine areas North Pacific, South Pacific, North Atlantic, South Atlantic and Indian Ocean. They pressurize governments to implement appropriate quotas and ensure that stock levels remain above sustainability thresholds;
- Representatives of governments, in the political zones Europe, America, East Asia, Asian developing countries and so on. They implement quotas and define taxes. Their goal is to ensure sufficient income and supply, and to avoid stock collapses in the region they manage.

The players are not directly the agents (that is the fisheries) involved in the model. Overall there are two levels: the level of the role-playing game itself (the agents are the players), and the level of the model (the agents are the fisheries).

Play proceeds as follows. First the game leader randomly selects a scenario with climate, investment and demand components. Then he or she distributes roles to the players, that is, the representatives of fishing and fish product industries, conservation societies and governments. A game has nine rounds. For every normal round (10 minutes, one year), (i) the game leader presents a specific context, (ii) each player determines his or her own strategy (which results in setting the values of scenario parameters for that time-step), (iii) the simulation is run according to these strategies, and (iv) each player is given the results and asked to analyse them. For special rounds (third and sixth rounds, 30 minutes), meetings are set up to coordinate strategies and to allow alliances to be forged. At the end of play, a meeting is organized so that feedback on what happened during the game can be given. This process is easily implemented as a functionality of the above model. Players have access to the parameters for the entities they represent: in any round, the representative of Asian fisheries gives a value to the parameters Adaptation of fishing capacity and Compliance for fisheries in China, Northeast Asia, Southeast Asia and Southwest Asia. Concomitantly, the representative of fishmeal industries in Europe gives values to the parameters of demand (slope and intercept) for the corresponding markets for fish products.

DATA COLLECTION

A rough but testing data set is constructed with existing data when they are easily available, and with reconstructed data from very general hypotheses when this is not the case.

Marine Areas

To characterize a marine area in the model, it is necessary to quantify the renewal rate r_e, the carrying capacity K_e and the fishing efficiency q_e. Most of these characteristics can be reconstructed from FISHSTAT, the FAO Fishing Effort database, and some modelling with a production model such as CLIMPROD (Fréon *et al.*, 1991).

Fisheries

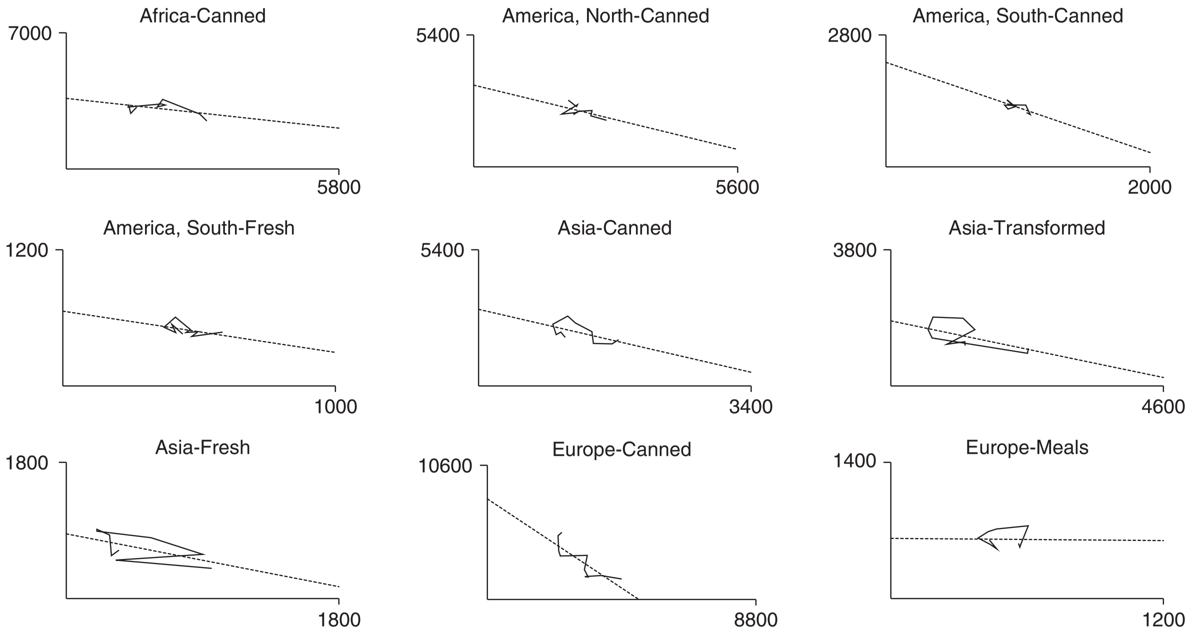
To characterize a national or a regional fishery in the model, it is necessary to quantify the fishing capacity, E_f. It can also be extracted from the FAO Fishing Effort database. In this version of the model, a proportional relationship between fishing capacity and average yield is assumed at the start of any simulation. A standardized boat is defined as producing on average 200 tons per year over the period 1990–2000.

Markets

To characterize a market for fish products in the model, one must quantify the parameters of the demand function (slope and intercept), A_m and C_m. These too can be extracted from FISHSTAT, which gives the volumes of exchanges, expressed either in tonnes or in a currency unit for recent years and for many markets for fish products. Prices and, by linear regression, coefficients A_m and C_m, can then be calculated (Figure 10.4). For several series, one knows only the mean price, mean quantities and the price/quantity elasticity, $\bar{P}, \bar{Q}$ and e, so one must use the formulation $A = \bar{P}(e+1)$ and $C = e(\bar{P}/\bar{Q})$.

Access to Marine Areas and to Markets

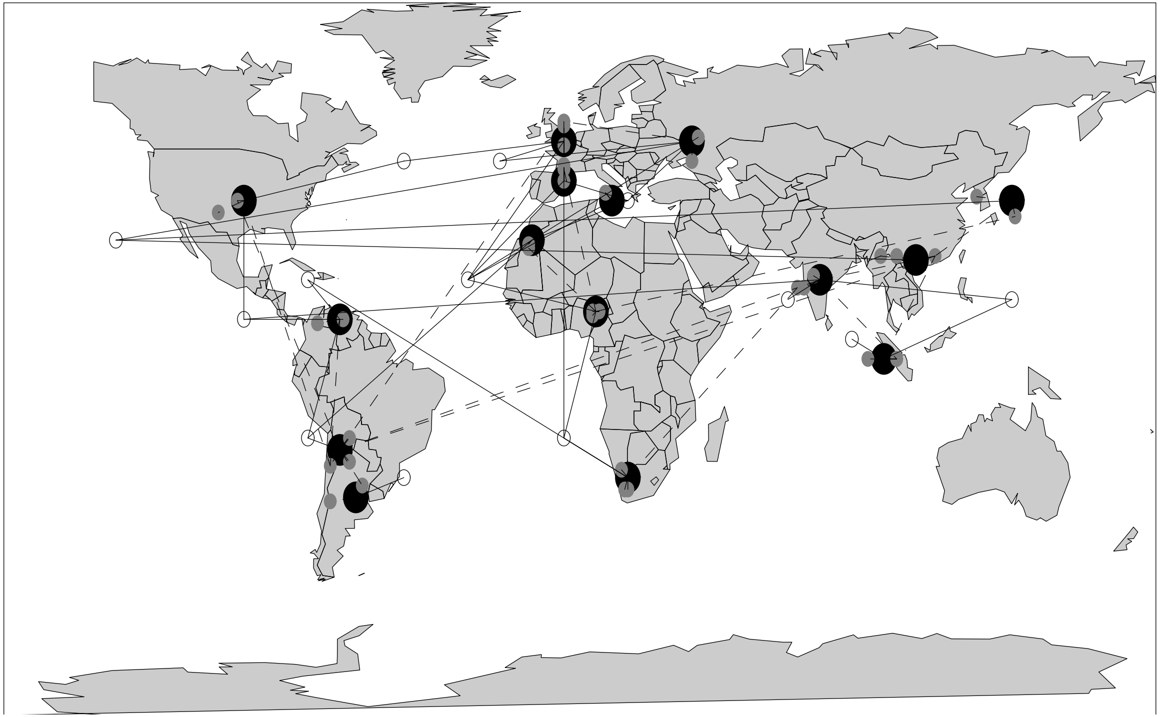
It is assumed that transportation costs from marine areas to fisheries and from fisheries to markets were proportional to the geographic distance used in the past (Figure 10.5).



Note: Price/quantities relationships for several fish products markets: Prices (abscissa) are expressed in dollars. Quantities (ordinates) are expressed in tons.

Source: After FISHTAT (FAO, 2004).

Figure 10.4 Relationships between prices (y-axis) and quantities (x-axis) during the past 15 years for several fish product markets, and the related demand functions



Note: Marine areas are represented by circles, fisheries by grey discs, fish product markets by black discs, usual routes from fisheries to marine areas by lines, and usual routes from fisheries to markets by dashed lines.

Figure 10.5 Network structure of the small pelagic fisheries system

MODELLING RESULTS

The model allows scenarios to be simulated and quantitative views of the resulting dynamics of pelagic fisheries to be generated. Here we present the results of two contrasting scenarios and a sensitivity analysis. However, in the model's current state, with non-validated data and with an algorithm that has not been fully checked, the results are simply indicative and caution must be applied to their interpretation. However, they do provide information on the model's dynamic behaviour, its plasticity and its sensitivity. The resultant individual dynamics (of marine areas, fisheries and markets) have to be interpreted in a speculative context: for instance, where we refer to the behaviour of the North European fishery, we have defined that entity with some properties of the real one simply to give some realism to the approach.

Scenarios

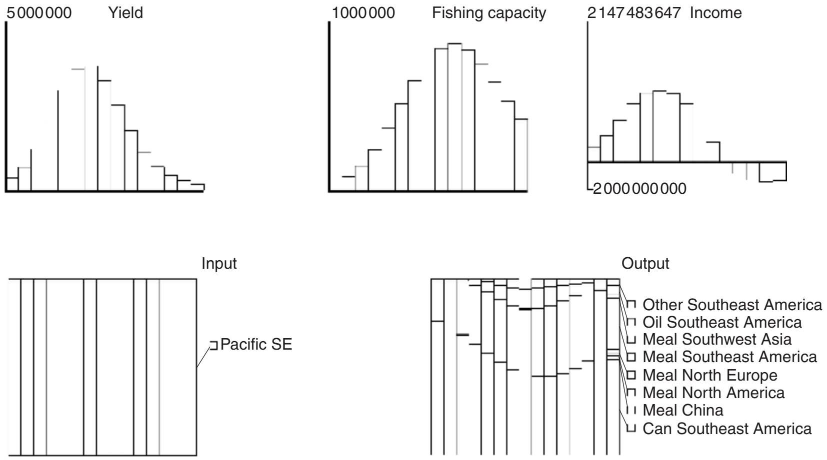
Black scenario

The black scenario is based on the following assumptions:

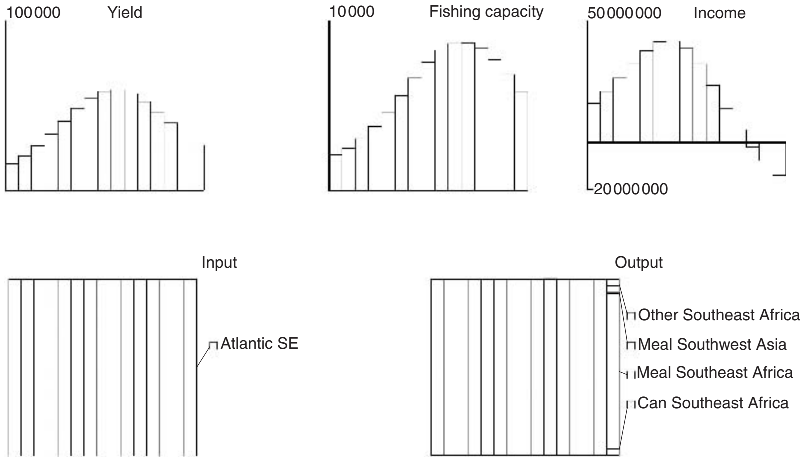
- Climate change results in an increasing productivity of the marine areas. This is represented by setting for all areas the parameters Changes in renewal rates and Changes in carrying capacity to −6 per cent.
- Globally, fisheries are quite rigid; they do not immediately adapt their fishing capacity to be in line with their income. This situation is represented by setting for all fisheries the parameter Adaptation of fishing capacity to + 3 per cent.
- There will always be some pressure on fisheries from fluctuations in the price of fuel. This situation is represented, for all routes to marine areas, by setting the parameter Changes of access costs to + 5 per cent.

Simulating the consequences of these hypotheses with the integrated model provides detailed results for marine areas, fisheries and markets. For all marine areas, stock biomasses decrease (not shown), as expected. Fisheries (Figure 10.6) increase their fishing capacity until they overexploit their resources. Incomes in each area mirror their yield, except at the end of the simulation for the Southwest Asian fishery, which recovers from losses at the start because it reorientates its production exclusively towards fresh fish for China. Moreover, the same fishery shifted its effort from the Western Central Pacific to the Western Indian Ocean then back to Western Central Pacific.

As a result of the simulation, one can obtain views of the network structure of the system, through kinetic maps. Figure 10.7 represents two



Southeast America

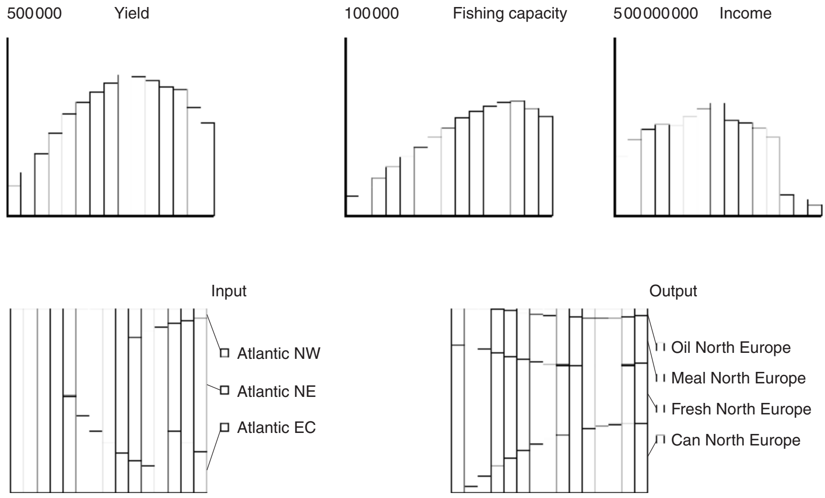


Southeast Africa

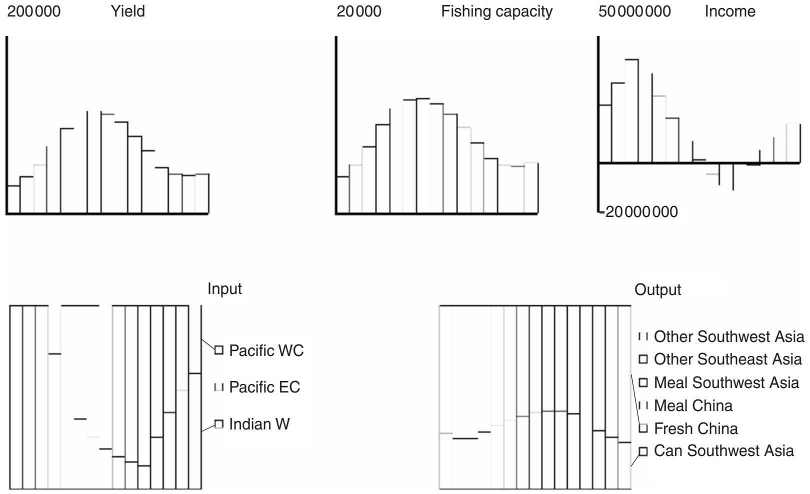
Notes:
The input figure represents how fishing effort is distributed, the output figure how the sales of fishing products are distributed.

Yields are expressed in tons; fishing capacities are expressed in normalized boats; incomes are expressed in dollars; input and output are expressed in percentages.

Figure 10.6 Black scenario. Results of simulations for Southeast American, North European, Southeast African and Southwest Asian fisheries



North Europe



Southwest Asia

Figure 10.6 (continued)

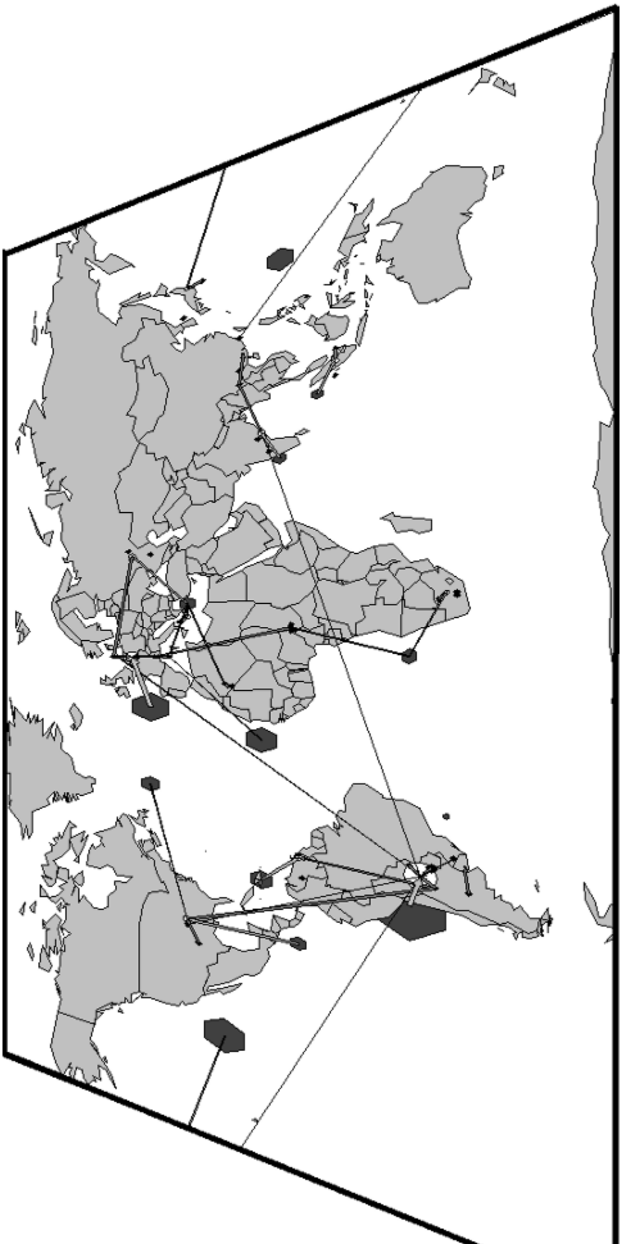


Figure 10.7a Black scenario: map of production and flow in 2006

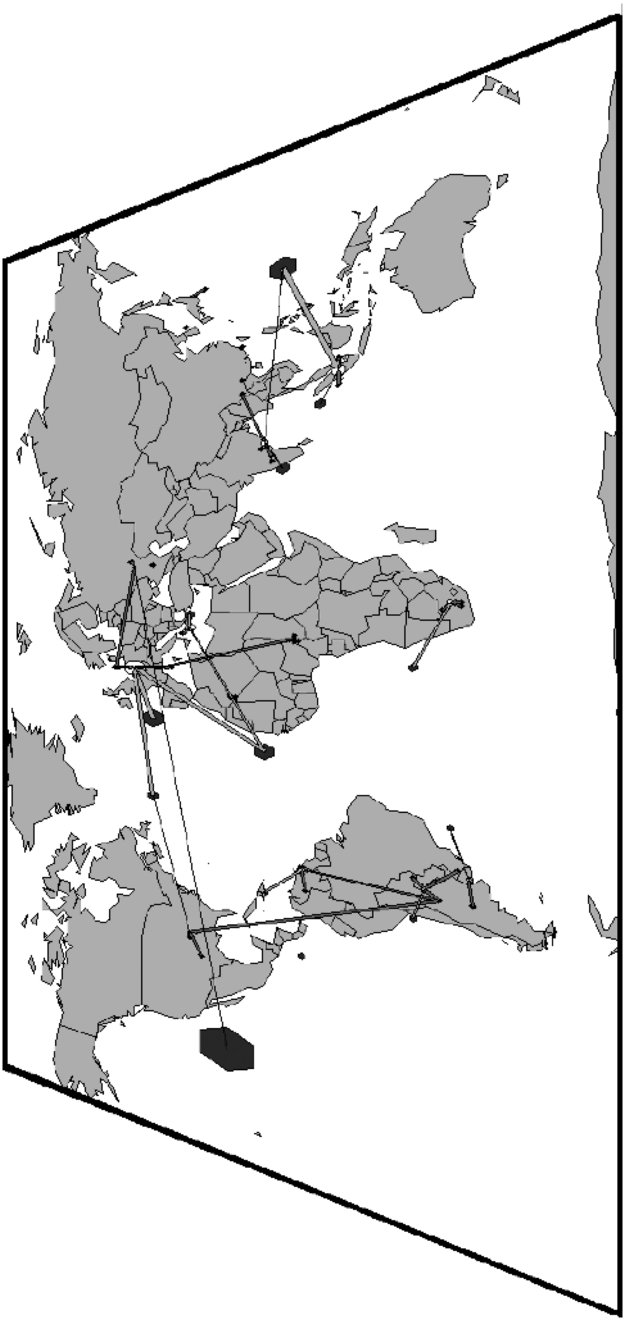


Figure 10.7b   Black scenario: map of production and flow in 2019

snapshots of the animations: for the years 2006 (start of simulation) and 2019 (end of simulation). These maps allow identification of structural patterns, for example some changes in the North Atlantic, which are attributable to the collapse of the North European fishery and the subsequent supply of corresponding markets from other fisheries.

Pink scenario

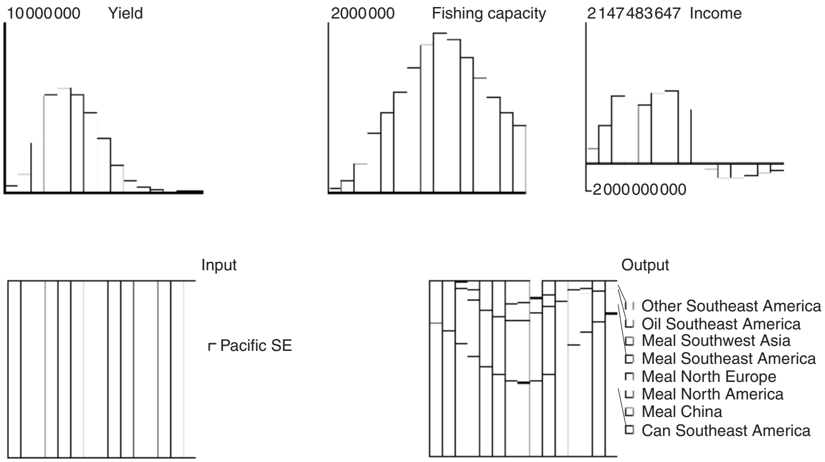
The pink scenario is based on the following assumptions:

- Climate change is beneficial for the productivity of marine areas. This is represented by setting for all areas the parameters Changes in renewal rates and Changes in carrying capacity to + 6 per cent.
- Globally, fisheries are reactive; they adapt their fishing capacity in line with their income. This situation is represented by setting for all fisheries the parameter Adaptation of fishing capacity to + 8 per cent.
- There will be some relaxation in the fuel price. This situation is represented, for all routes to marine areas, by setting the parameter Changes of access costs to −5 per cent.

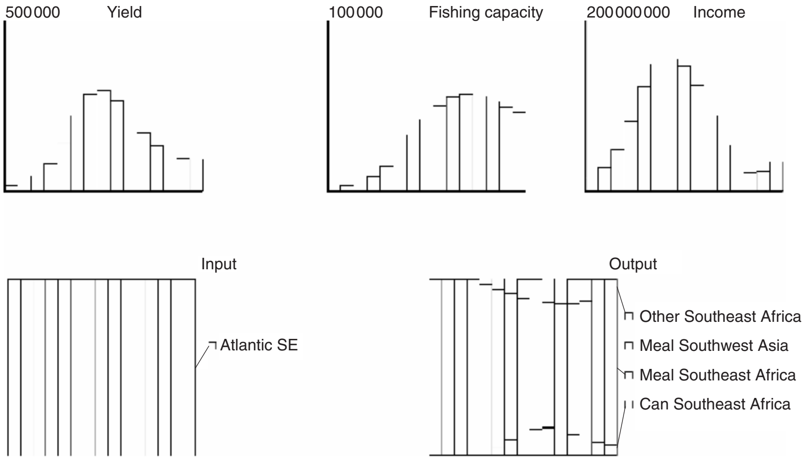
With this scenario, predictions are variable (Figure 10.8) and not all positive. Several fisheries (Southeast America, Southeast Africa) collapse owing to their high reactivity; their fishing capacity increases too much and the stocks they exploit weaken. At the opposite end of the spectrum, the North European fishery, with low income at the start of the simulation, immediately reduced its fishing capacity, and moved its fleet within the Atlantic Ocean to generate sustainable income. Its yields increased, and it was able to sell its output on different markets in a dynamic manner, without interference from other fisheries that had collapsed. In this scenario, the Southwest Asian fishery shows patterns of sustainability that are comparable to the ones of the North European fishery.

Sensitivity Analysis

The results thus far highlight the adaptation of fishing capacity to generated income as an important factor in determining the dynamic behaviour of the small pelagic fisheries system. Therefore, a sensitivity analysis was performed with this parameter, allowing it to vary from 0 to 8 per cent. Low levels of adaptation are conservative for the stocks (Figure 10.9); they favour maximum yield at the end of simulation. In contrast, high levels favour adaptation at the start of the simulation. High production with a high level of adaptation is offset by lower prices, resulting in smaller incomes.



Southeast America

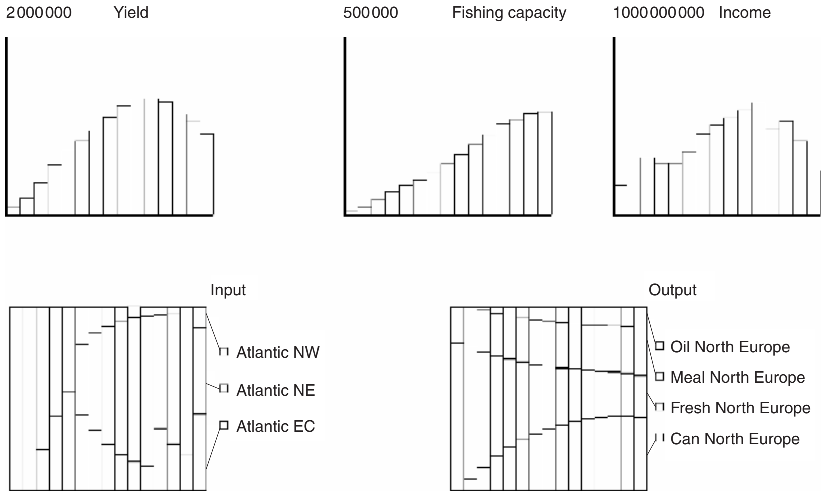


Southeast Africa

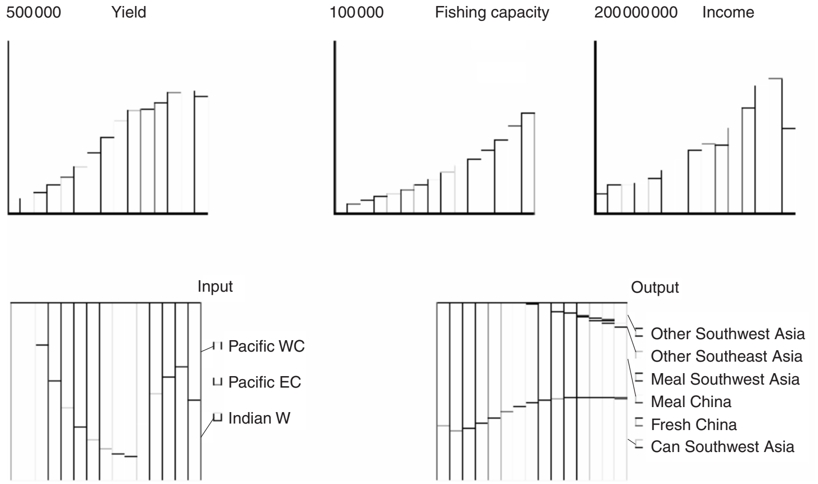
Notes:
The input figure represents how fishing effort is distributed, the output figure how the sales of fishing products are distributed.

Yields are expressed in tons; fishing capacities are expressed in normalized boats; incomes are expressed in dollars; input and output are expressed in percentages.

Figure 10.8 Pink scenario. Results of simulations for Southeast American, North European, Southeast African and Southwest Asian fisheries

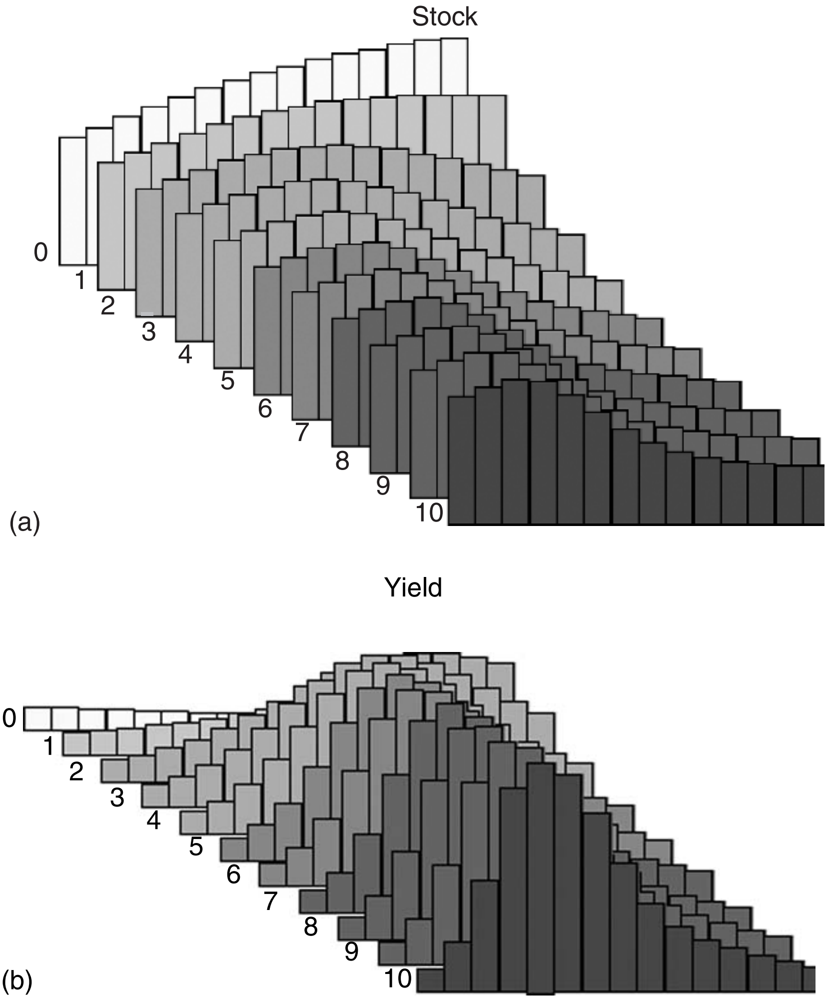


North Europe



Southwest Asia

Figure 10.8 (continued)



Note: Simulation 0 corresponds to 0 per cent, simulation 1 to 0.8 per cent, 2 to 1.6 per cent, and so on.

Figure 10.9 Sensitivity analysis on a 15-year (x-axis) simulation: global results (stock, yield, income, effort, price) of 11 simulations for values of parameter Adaptation of fishing capacity varying from 0–8 per cent

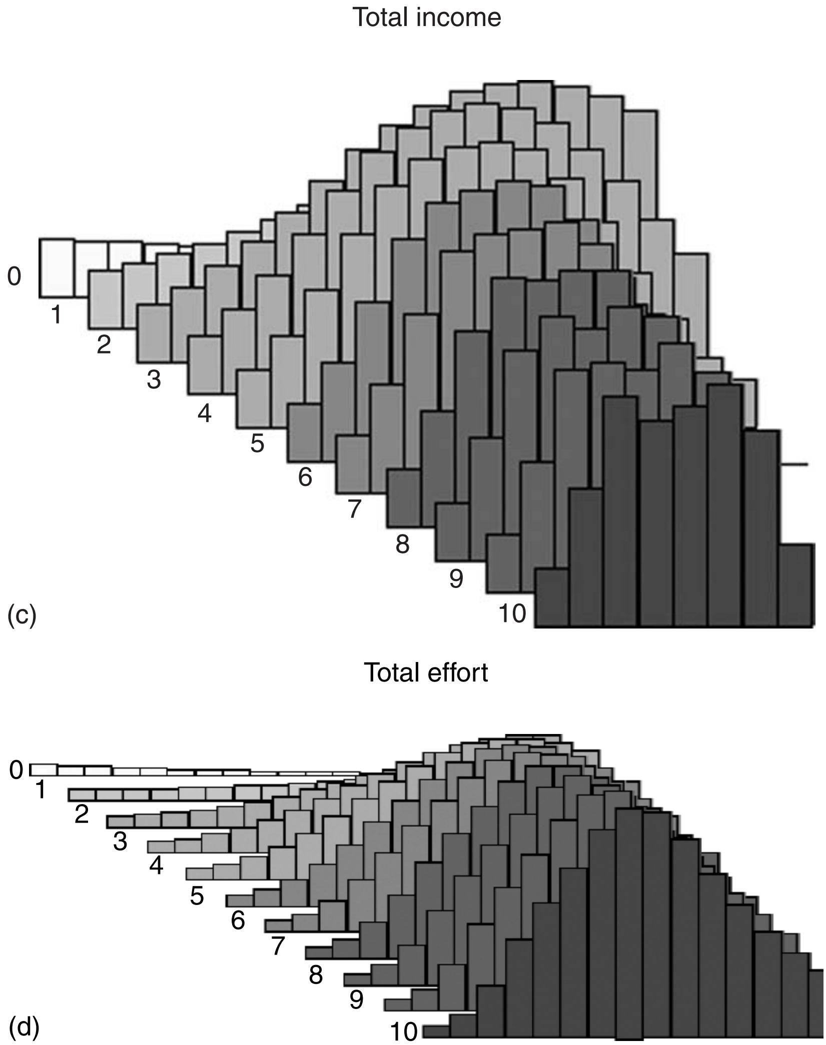


Figure 10.9 (*continued*)

DISCUSSION

One may ask whether the objectives of this preliminary step towards developing an integrated model of the worldwide system of small pelagic fisheries have been reached. The prototype has defined the components of the

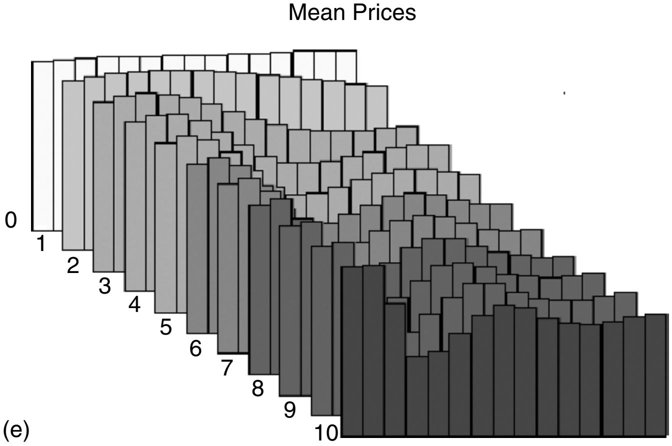


Figure 10.9 (continued)

integrated model. The selected resolution (disaggregation) allows dynamic patterns to be reproduced; more complicated modelling of the economic behaviour (macroeconomic looping) seems to be unnecessary.

The databases necessary to support the model (in terms of parameterization and validation) have been defined. Even if entities are not exactly those needed by a dynamic model, the FAO databases can provide most of the required data; complementary data can be provided by the International Fishmeal and Fish Oil Organization (IFFO) and the International Food Policy Research Institute.

The technical feasibility of the model has been evaluated: computing algorithms are fast enough to provide an interacting framework for the modelling itself. Several issues have, however, been raised by the preliminary results:

- Discussion of the main assumptions of the model are crucial: (i) the worldwide small pelagic fisheries as a system, (ii) the fisheries as active entities of the dynamics of that system, (iii) the dynamics of that system as the results of a coupling between deterministic processes and a competitive equilibrium.
- The model must be made more realistic, that is (i) tuning the definition of entities (marine areas, national or regional fisheries, markets for fish products), considering several groups of pelagic species instead of just one, (ii) improving the estimation of access costs, and (iii) using more appropriate data sets.
- Role-playing game sessions must be organized better and their progress more effectively analysed.

- The mathematical formulation should be improved. The whole formulation of the model could be rephrased in the framework of network economics (Nagurney, 1999), which allows general hypotheses, such as considering non-linear price functions, to be relaxed. A similar approach could be applied to different marine systems, at a different scale and in different contexts. For example, it may be of value to refine the dynamics of the system by focusing on one area, for example the Southeast Pacific, where fleets and species can be further disaggregated and parameterization improved.
- The main assumption of this modelling approach (the worldwide small pelagic fisheries as a system), needs in-depth discussion: do the worldwide pelagic fisheries constitute a single system? Would it not be of greater value to focus on the interactions between upwelling ecosystems, small pelagic fisheries and markets of small pelagic products rather than on the interactions between coastal upwelling ecosystems and deep-sea ecosystems, or on the targeting behaviour of fisheries switching between small pelagic resources and other fish resources, or on the interactions between all fish products, or between fish products and substitutes (for example soya meal versus fish meal)? Our preliminary modelling experiments may contribute to resolving this question. Although they are very unstable at all levels of organization, climate, biology and economics, but still highly viable (Fréon *et al.*, 2005), the system of small pelagic fisheries provides a good case study of collective management of a shifting resource. For example, it can help to address the question of overcapacity as a structural adaptation to fish variability.

According to FAO (2002), one of the most important challenges facing the world's fisheries management lies in improving the data systems. Such a model relating very different components of the system, and providing a global overview of it, can be used to reveal and minimize incoherencies between data sets. This is a similar approach to one that recently showed that Chinese catches were overestimated in the past (Watson and Pauly, 2001). A global model can be used in a systematic way with the same purpose.

The probable increase in conflicts in the worldwide system of small pelagic fisheries attributable to globalization and climate change underlines the urgent need for tools of consensus building to be developed. The present prototype of the model is a step in this direction, because it should allow discussion between the different stakeholders and favour unifying points of view within the context of an ecosystem approach to fisheries management.

ACKNOWLEDGEMENTS

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11. On the consequences of climate change in pelagic fish populations: a conclusion

Rögnvaldur Hannesson, Samuel F. Herrick Jr and Manuel Barange

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Since the 1980s, man-made climate change has been high on the agenda worldwide. Climate models have shown the possibility of a rise in average annual temperatures of several degrees over this century, albeit with wide confidence limits (IPCC, 2001). Temperature changes of this magnitude would most likely be associated with other climatic events, such as changes in rainfall, and strength and frequency of storms. This would, needless to say, affect human activity in a number of ways, in particular activities that depend directly on the forces of nature. Farming is the activity that immediately comes to mind; crop success depends on temperatures and rainfall and therefore it influences the supply of food to humans, both directly and indirectly through animal feed.

Fishing is another activity, the outcome of which depends critically on the forces of nature. Indeed, this is so to a much greater degree than farming. Despite all technological advances, fishing is still a form of hunting, the success of which depends on environmental conditions over which people have little or no control, be it through abundance of fish stocks, their migrations and accessibility, or the weather. We can plough and fertilize the land, and our crops remain in their locations until we harvest them, but we cannot truly fertilize the ocean and the fish go where they want or where carried by currents. Fish farming is responsible for a small fraction of the total supply of fish worldwide, but even in that activity we are as exposed to the vagaries of the weather as in farming on land, if not more so. As an example, inclement weather in Scotland during January 2005 released 600 000 farmed salmon from their cages, causing an economic and ecological crisis in the industry (http://news.bbc.co.uk/1/hi/scotland/4287407.stm).

Anthropogenic climate change is likely to affect the oceans and hence the environment of the fish stocks we exploit no less than it does our land environment. However, the uncertainties surrounding impacts on the oceans seem even greater, so predicting fisheries yields under climate change is even more difficult than predicting effects on land-based activities. Yet, given the high profile that climate change issues have acquired, it would be an omission not to make an effort to do so. Certainly we have enough information to indicate that climate change impacts are not negligible. For example, catches of northwest Atlantic cod during the period 1600–1900 were correlated with sea temperature, while shorter term variations in North Sea cod have also been related to a combination of overfishing and ocean warming (O'Brien *et al.*, 2000). The decadal variability in the Japanese sardine catch has been related to variability in the ocean and climate in the North Pacific, and responded synchronously with sardine catches off Chile and California (Kawasaki *et al.*, 1991) and Pacific salmon catches (Beamish *et al.*, 1999). Changes in the abundance of herring (*Clupea harengus*) and sardine (*Sardina pilchardus*) in the North Sea and Baltic Sea have been linked to variations of the North Atlantic Oscillation and the resulting strength and pattern of southwesterly winds (Alheit and Hagen, 1997). These and other examples indicate that fish resources respond partially to climate forcing at global scales, suggesting a need to conduct coordinated research and impact assessments. Suggestions that continued warming will compress the distribution of some stocks (for example sockeye salmon; Welch *et al.*, 1998), squeezing them out of their traditional habitats, highlights the urgency this matter deserves. Given the great uncertainty we face, it would be appropriate to start from first principles. As the uncertainty hopefully is resolved, we could draw upon those results and revise the conclusions we have come to from a more rudimentary knowledge.

Arnason's chapter in this volume is an attempt to derive some first principles on how to adjust to climate change in the oceans. Clear-cut results are few and far between, even in the unlikely scenario that changes in the productivity of fish stocks are deterministic; it all depends on circumstances, such as whether unit costs depend on stock size. Increased productivity of fish stocks might in some cases mean a reduction in catch rather than the opposite. Uncertainty about how climate change influences stock growth leads to further complications. One result shown in this chapter is that if changes are gradual, adjustment to these changes can be myopic; it would not be necessary to predict these changes. However, even if climate changes turn out to be gradual, the impacts on fish stocks could be abrupt. It is possible, and indeed likely, that abrupt changes in fish populations will be triggered when ocean temperatures or other environmental variables (*e.g.* critical currents linking spawning and recruitment grounds) surpass certain threshold values. Hence, nothing much need happen as ocean temperature changes gradually, until it falls below (or exceeds) a certain critical level. Then, suddenly, there is not sufficient plankton to feed the stock or its predators, on which the fishery may be based, and the resource collapses. This happened for the Atlanto-Scandian herring in the 1960s. These environmental changes are described in Arnason's chapter, while the social and economic consequences are discussed in the chapters by Hamilton and his colleagues, and Lorentzen and Hannesson.

It would indeed seem a promising avenue of research to explore the consequences of abrupt regime changes triggered by gradual environmental change that comes to surpass certain thresholds. What are the implications of being unable to forecast such changes? What happens if we can do so shortly before they occur? How lengthy a lead time do we need for such forecasts to be able to adjust exploitation strategies? What are the implications of climatic variability around a trend? What happens when thresholds are passed in both directions repeatedly, before perhaps being surpassed for good, or at least for long enough to be permanent from the point of view of the present human generation? These are largely unexplored areas of inquiry and yet are seemingly important.

Scientists have now recognized the persistence of multi-year regime shifts. Changes in recruitment patterns of fish populations and the spatial distribution of fish stocks have been linked to climate-ocean system variations, such as the *El Niño*-Southern Oscillation (ENSO) and decadal-scale oscillations (Lehodey *et al.*, 2003). Fluctuations in fish abundance are increasingly regarded as a biological response to medium-term climate-ocean variations, and not just as a result of overfishing and other anthropogenic factors. Sudden regime shifts as a consequence of climate change can be of various kinds; some fisheries are likely to be positively affected, and others negatively (IPCC, 2001). The direction of the effect may be linked to the biogeography of the stocks, which might be displaced from the areas where they used to be abundant (a perceived negative effect) and into others where they used to be rare (a perceived positive effect). The net effect may be negligible, but different countries could be affected in a radically different fashion, especially now that the ocean has been carved up into different national exclusive zones with controlled access. Under the old, open access regime, fishers could operate pretty much where they wished, but only the fish now have that freedom. Moreover, the repercussions of the effects in one particular fishery could spread far and wide to other fisheries and other sectors of the economy, because products derived from one fishery are often sold in competition with products from others. Hence, through these interlinks via markets, fisheries unaffected by climate change *per se* could be affected by changes in other, perhaps distant, fisheries. As fish products are traded far and wide, consumers could be affected by climate changes in distant locations. The chapter by Briones and others describes a market model that can be used to trace these links. They focus on developing countries in Asia where small pelagics are an important part of the food supply. As small pelagics typically are susceptible to climate variability, the effects of climate change on people's livelihoods are likely to be particularly strong in that part of the world.

While rising temperatures over the long term (100 years or more) may be a new trend, climate fluctuations on a decadal scale are nothing new (Hare and Mantua, 2000; Beaugrand, 2004), and neither are short-term fluctuations, such as the famous *El Niño* (Lehodey *et al.*, 2003). Ocean temperatures in Norwegian coastal waters are no higher today than they were in the 1930s, and in that time interval they have risen and fallen over a time span of a few years, ignoring of course seasonal variability. The temperatures in the seas around Iceland and Greenland were appreciably higher in the 1920s and 1930s than before and after. In that period both cod and herring colonized new areas; cod began to spawn off the north coast of Iceland and herring off western Greenland. The cooling of the ocean north of Iceland drove the herring farther east and north, with a devastating effect on what Hamilton calls in his chapter the one-time herring capital of the world, and later on the Icelandic herring fisheries in general.

While the spatial displacement of herring was due to changes in ocean currents, as shown in the chapter by Hamilton and his colleagues, the collapse of the entire stock of Atlanto-Scandian herring was probably due to overfishing, even if the adverse climate change may have played a leading role. The effect on the herring fisheries of Norway and Iceland was devastating; herring catches fell from almost two million tonnes to almost nothing over the course of a few years. No regime shift that could result from a man-made climate change is likely to produce more drastic effects on a fishery, so the consequences of that collapse are certainly of interest for what we may have to cope with in future. People adjusted to the disaster in various ways, as discussed in the chapters by Hamilton and his colleagues, and Lorentzen and Hannesson. The California sardine fishery also collapsed in the 1950s, following the demise of the Pacific sardine stock. That experience is briefly mentioned in the chapter by Herrick and others. Another such fisheries-driven collapse is that of Namibian pilchard (sardine), which was unable to cope with the pressures of the fishery and struggled to recover after severe fishery regulation (Boyer *et al.*, 2001). The economic consequences of this collapse are the subject of the chapter by Sumaila and Stephanus.

In summary, the world has learned to live with changes as dramatic as any likely to result from global warming, as far as fisheries are concerned. If anything, countries that are now considerably better off than they were at the time would seem to be better able to deal with such changes now or in the future, provided their economic development does not change gears and go into reverse. On the other hand, it could be argued that as societies develop economically, they become more specialized and employ more complicated and capital-intensive technologies that require skilled and trained labour, and have little need for the unskilled or those with skills that are not in demand. Skills in fishing are not always easily portable to other industries, so it can be argued that more economically developed societies will be less able to deal with events like fisheries collapses that suddenly throw a large number of people out of work. In earlier days, unskilled labour was in greater demand, and unemployed fishers were more easily absorbed into other sectors. On this account we might be less able now to deal with fisheries collapses than we were a few decades ago, unless we have in place adequate income-transfer mechanisms and retraining programmes for those in need of new, marketable skills.

However, the effects of regime changes need not just be adverse; it is quite possible that such effects will be positive, with higher ocean temperatures raising the productivity of stocks and enabling them to colonize new areas. Such effects were touched upon above, for the Atlanto-Scandian herring and the Icelandic cod. Some forecasts envisage the Arctic as ice-free during summer, which might make it possible for cod to colonize that area and establish new spawning grounds farther north (Stenevik and Sundby, 2003). This need not, however, imply a net increase in productivity; it could merely mean a displacement of the habitat of the stock farther north. This could have repercussions for how these stocks are being shared between the nations in whose economic zones the stocks are now found. For example, Hannesson (2004) discusses how such movements may affect the sharing of Northeast Arctic cod between Norway and Russia. The chapter by Hannesson in this volume explores how a positive effect on the growth of the Norwegian spring-spawning herring, the main component of what used to be called Atlanto-Scandian herring, together with an increased migration of the stock, might affect the sharing of the stock between the nations that now exploit it. A complicating factor here is that this stock straddles a 'hole' of international waters that lies between the exclusive economic zones of the countries around the Northeast Atlantic.

More generally, climate changes that change fish migrations and displace the habitats of stocks are likely to affect the agreements that are now in place among countries in whose economic zones those stocks are located. While the stocks are in 'transition' from one regime to another, existing agreements will probably come under strain and be abandoned, as they lose their basis in reality. Things may be further complicated if the changes occur as fluctuations around a trend (Hannesson, 2005). However, trend-free fluctuations alone can cause problems, as shown in the chapter by McKelvey and colleagues here. Under such a scenario, it is possible that competitive fishing by two agents would result in the extinction of the stock they fish. Surprisingly, this would be more likely to happen, the better the information on the actual distribution of the stock the two agents have. We do not have to invoke global warming as a reason for such problems emerging; the fluctuations in oceanographic conditions in the Benguela Current, the Humboldt Current and elsewhere are known to affect the distribution of anchovy and sardine stocks between the economic zones of Namibia and South Africa, and Peru and Chile respectively (Schwartzlose *et al.*, 1999). This is likely to put agreements on shared stocks under strain (in many cases there are not even now agreements on stocks that migrate between the zones of two or more countries). Herrick here notes how the Pacific sardine spreads all the way from Mexico to Canada's British Columbia when it is plentiful. Conflicts over the allocation of such highly fluctuating stocks can arise between and perhaps even within a sovereign country. Such problems have been approached by game theory, which for quite some time has been applied in the form of experiments. The chapter by Mullon and Fréon outlines an ambitious plan to conduct such experiments about the sharing of stocks globally, recognizing the interconnectedness in the fishmeal market. Effects of climate variability are among the features built into his model.

Even if global warming follows the lower end of the IPCC predictions (IPCC, 2001), fluctuations in ocean climate and the associated regime shifts will surely be with us in the future, as they have in the past. Most of the chapters in this volume do in fact deal with climate fluctuations rather than global warming, but much that can be said about climate fluctuations can be applied to global warming, especially if it is thought of as a trend with substantial short- and medium-term deviations from that trend. Fluctuations in ocean climate and the associated dynamics of fish stocks, sometimes referred to as 'regime shifts', are topics worthy of research in their own right, and they pose questions of great relevance for fisheries management. One of these questions is what meaning we can give to the term 'sustainable fishing'. There is a suspicion that this term owes much to the classic deterministic fisheries models where the concept of sustainable yield is clear and unproblematic. Once we allow for stochastic variations in growth, things become complicated. With stochastic fluctuations, as we surely have in the real world, it would be difficult, probably undesirable, and maybe impossible to take a constant catch of fish year after year; our fishing would have to vary according to the situation at each particular time. Sustainability could be given the meaning of not exceeding the productivity of the stock over a long time perspective so as not to drive it down to levels at which it cannot sustain itself. However, some implications of fluctuations may run contrary to the sustainability idea. As shown in the chapter by Sumaila and Stephanus, it is conceivable that the industry and in fact its fisheries managers would prefer a catch profile where an abundant stock is fished down heavily to a low and unproductive level over a profile that sustains catches at an even level over a long time period.

Another question raised by fluctuations in stock growth concerns the optimal fleet capacity for fishing such stocks. Much of the contemporary discussion on problems in fisheries management identifies excessive fleet capacity as the root cause and gives the impression that if we only bring this under control, all should be well. However, catching the same quantity year after year is not likely to be the best way of utilizing fish stocks that fluctuate substantially over time; a sustainable catch that is constant irrespective of its associated environmental conditions would, for some stocks, have to be set ridiculously low. Hence, the optimal utilization of stocks that fluctuate substantially would also have to vary, depending on the conditions at each particular time, especially if the stock consists of one or only a few year classes. In turn, however, this will almost certainly demand fleet capacity that in some years is 'excessive' and in other years insufficient to take the catch that it would be advisable to take.

All this presupposes that fisheries and fleet capacity are either managed by some central authority or else that there are incentives in place to induce the industry to avoid investing in excessive fleet capacity. In the case of the former, Herrick discusses development of the US harvest policy for Pacific sardine. The latter is likely to apply in fisheries controlled by individual transferable quotas, but this is clearly a far cry from what would happen in open access fisheries. In those, it is highly likely that environmental fluctuations would exacerbate the development of overcapacity of fishing fleets. Good years are likely to breed over-optimism and so cause investment in greater overcapacity than would transpire under the textbook-model case of deterministic stock growth.

Particular and interesting problems are posed by regime shifts in which the dominant species is replaced. Regime shifts occurred long before man's exploitation of the stocks began (Baumgartner *et al.*, 1992), and must therefore be caused by environmental changes beyond human control. Such shifts have since occurred in modern times in the North Pacific (Hare and Mantua, 2000) and North Atlantic (Beaugrand, 2004), when human exploitation was an important forcing. These recent events question whether such shifts could be brought about or accelerated by fishing. How should we regulate our exploitation, given that such regime shifts are natural? If the shifts are caused by environmental effects that are currently difficult or impossible to predict, they would seem to make sustainable exploitation a lot less interesting a subject. In other words, if catches cannot be sustained anyway, should we not take the fish while the stocks are there (see Johnston and Sutinen, 1996)? This way of thinking becomes all the more relevant as some of these shifts involve species that are raw material for fishmeal, a product that does not depend much on the fish from which it is derived. Sardine and anchovy are pairs of species that appear to fluctuate out of phase in several parts of the world: in the Benguela Current, the Humboldt Current, the California Current and off Japan. De Oliveira's chapter deals with the trade-offs between return and risk for the mixed fishery for sardine and anchovy off South Africa, taking into account that sardine (also called pilchard locally) are a by-catch when young in the fishery for anchovy. His approach takes into account explicitly the uncertainty surrounding the fluctuations that generate the regime shifts in the sardine-anchovy complex off South Africa.

We have mentioned a number of research issues that were broached or were brought to the fore by the various papers given at the workshop. One of the most obvious conclusions is that much fisheries-related research is going on in various places, but little of it addresses the economic effects of climate change or climate variability. This is likely to be due to the great uncertainty regarding the predictability of the effects of global warming. However, global warming notwithstanding, climate variability in the ocean is, and has long been, a real issue. This variability can have, and has had, major economic consequences, and it is of major interest to deal with them and thus to avoid their most serious consequences, if possible. It is our impression that this aspect of fisheries management and economics has received too little attention. We hope that this volume will stimulate further activity along those lines.

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