ANALYSIS

Single species conservation in a multispecies fishery: the case of the Australian eastern gemfish

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Abstract

The inability of fishers to avoid bycatch when operating in a multispecies fishery can create problems when the bycatch species itself is the subject of conservation-based controls. Increased protection of the bycatch species can only be achieved through a reduction in the overall catch of the fishery. Such a problem has arisen in the Australian south east fishery where the stock of one bycatch species has fallen substantially over recent years. A model of the fishery was used to estimate the costs of protecting the remaining stock and the potential commercial benefits from the recovery of the stock. The analysis suggests that the costs of protecting the stock may be greater than the commercial benefits arising from a recovered stock. Non-market benefits of stock conservation are not included in the model analysis, but are considered in the discussion. © 2000 Elsevier Science B.V. All rights reserved.

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1. Introduction

The objective of fisheries management in most countries is to ensure the continuing existence of the resource, whilst improving the economic and financial status of the fishing operators. In most cases, these objectives are complementary. While for some species there may be an economic argument to deplete a stock based on commercial use values (Clark, 1990), for most species improving the economic performance of a fishery involves increasing, or at least maintaining, the size of the stock.

In a multispecies fishery, the criteria for optimal exploitation is less obvious. The more commonly advocated forms of stock management are inappropriate given that managers are concerned with the exploitation of a range of species (Kirkley and Strand, 1988). This in itself would not be a problem, except that many multispecies fisheries are characterized by joint production (Kirkley and...
Strand, 1988; Thunberg et al., 1995). As fishing gear is not species specific and stocks of different species occupy the same habitat as the ‘target’ species, fishers usually catch a range of species that they were not intentionally targeting.

The inability of fishers operating in a multispecies fishery to avoid the incidental capture of some species can cause problems when a bycatch species is itself the target of conservation measures. Incentives exist for individual fishers to continue fishing for the target species at the expense of the bycatch species, provided the marginal value of the catch (excluding the discarded bycatch) exceeds the marginal cost (including the opportunity cost of targeting different species groups) (Pascoe, 1998). This is essentially a principal-agent problem arising out of the agent’s incentive structure differing from the principal’s preferences (Moyle, 1998). The provision for inclusion of unique stocks in endangered-species legislation in many countries around the world, even though the species as a whole may not be endangered, has exacerbated the problem. Regional depletion of one species in a multispecies fishery can result in severe restrictions being placed on the fishery as a whole. This problem is currently being experienced in the USA through the listing of some stocks of salmon along the Pacific coast. The listing of Snake River sockeye salmon as endangered in 1991 has seen a continuing debate over whether to increase measures to protect the species (at the cost of reduced hydroelectricity generation) or maintain the status quo and risk the extinction of the stock (Lewis, 1994). The listing of Snake River chinook salmon resulted in the fishery being closed during the spawning run in 1994. The closure resulted in the estimated loss (in terms of foregone capture) of 23,000 king salmon in order to save eight chinook (Ess, 1994).¹ One-third of the salmon stocks in the Columbia River are extinct, whilst a further 40% are at risk of extinction (Bragg, 1994).

A similar problem has developed in the Australian south east fishery, where the stock of eastern gemfish (Rexea solandri) has fallen to a level where fears for the survival of the stock have been raised (Bureau of Resource Science, 1998). The eastern gemfish stock was considered for listing under the Endangered Species Protection Act (1992) at the fourth meeting of the Endangered Species Advisory Committee (ESAC) and Endangered Species Scientific Sub-committee (ESSS) in September, 1994. The species was rejected for listing at that time on the basis that management plans had been introduced to further halt its decline. Managers had responded to the stock collapse by imposing a zero total allowable catch (TAC) on eastern gemfish. However, eastern gemfish are also caught as bycatch with many other species. As a result, it is expected that considerable quantities of eastern gemfish are still being caught as bycatch and discarded (Bureau of Resource Science, 1998). This has caused concern to fisheries managers, scientists, fishers, and conservation groups, some of whom have called for further measures to protect the stock.

The purpose in this paper is to present a method for assessing the commercial benefits and costs of attempting to protect a single species in a multispecies fishery. The method involves the use of a linear programming model to estimate the value of a fishery with and without recovery of the threatened bycatch species, and the costs of introducing measures to enable the species to recover. The model is applied to the Australian southeast fishery to examine the commercial benefits and costs of gemfish recovery.

2. Gemfish and the south east fishery

The south east fishery is the main supplier of fresh fish to the Sydney and Melbourne markets, the largest fish markets in Australia. Over 100 species are caught in the fishery, although over 80% of the value of landings is accounted for by 15 species (ABARE, 1993a). Individual transferable quotas (ITQs) were introduced for these species in 1992 in order to protect the stocks and improve the economic performance of the fleet.

Gemfish have been a major species harvested by the inshore sector of the fleet since the early 1970s. Two stocks of gemfish have been identified

¹ Ironically, Snake River King salmon has also subsequently been added to the endangered species list (Ess, 1994).
in the fishery (Kailola et al., 1993). Eastern gemfish is caught off the south coast of New South Wales, with most catch occurring during the winter spawning run. Western gemfish are caught mostly off the west coast of Victoria and are largely taken as bycatch of other species.

A 3000 tonne TAC was introduced for eastern gemfish in 1988 following concerns that the stock was either over exploited or being affected by unknown environmental factors (Kailola et al., 1993). The TAC was subsequently revised downwards in response to falling recruitment in the fishery, with a zero TAC being imposed in 1993. In 1997, the TAC was increased to 1000 tonnes was part of an adaptive-harvesting strategy to validate the models used in the stock assessments (AFMA, 1997). However, only 374 tonnes were caught (Bureau of Resource Science, 1998). A zero TAC was again proposed for 1998 (AFMA, 1997), but a 300 tonne TAC was imposed in recognition that bycatch was unavoidable as long as the fishery remained open (Bureau of Resource Science, 1998).

The recovery of the stock will largely rely on the conservation of the remaining resource. In such a case, overquota catch and subsequent discarding of eastern gemfish could threaten the biological status of the stock. The discarding of an otherwise valuable fish is also seen by many groups (including fishers) as a waste of the resource. Most discarded gemfish die before being returned to the water, mostly as a result of the effects of decompression.

3. Modeling benefits and costs of stock conservation

The possibility of extinction of a fish stock is a function of prices, costs, biological features of the fish stock, and the rate of entry and exit to the fishery (Cheng et al., 1981). Given problems of non-malleability of capital (Clark et al., 1985; Schellberg, 1993), fish stocks can be driven below their minimum viable biomass under certain price and cost conditions (Cheng et al., 1981; Weisbuch et al., 1997). As the cost of capturing bycatch species is negligible (since they were caught inci-dentally), the survival or extinction of the bycatch species in an unmanaged (open access) fishery will largely depend on their relative contribution to the income of the fisher.

Management measures can be introduced, however, to protect individual species within a multi-species fishery. These measures can include area, seasonal, or total closures. However, such measures also impose costs on fishers in terms of foregone revenue from the catch of the target (and other bycatch) species. Estimating these costs are not straightforward, as fishers have the capability to adjust their activity to compensate (to some extent) for the restrictions imposed. Closing down a fishery, or prohibiting the landing of a particular species, will result in a reallocation of effort to other fisheries or other target species.

Given the complexity of most multispecies fisheries, some form of model is required in order to estimate these effects. This also requires assumptions about fishers’ behavior to be made. An assumption that is often made is that fishers are profit maximizers, subject to constraints imposed by management, the environment (such as weather), and social factors. Given this assumption, a mathematical programming model can be developed to estimate changes in the allocation of effort between alternative activities in response to various management changes. Such models have been developed for a number of multi-species fisheries (Brown et al., 1978; Siegel et al., 1979; Sinclair, 1985; Murawski and Finn, 1985; Frost et al., 1993; Pascoe, 1997; Mardle et al., 1999).

The model used in this study, formulated as a linear programming problem, was developed for a multispecies fishery exploited by several boat types (sectors), each facing different technological constraints and cost structures. The fishing gear employed is not species specific, so each boat catches a number of different species with each haul. The catch composition varies by area fished, depth, and month as the relative local abundance of the species change. Catch composition also varies by fishing gear. Stock dynamics are not included in the model, so the model has only an annual time frame. The model allocates effort each month from a given fleet size and structure to the different activities (i.e. areas and depths
fished) across the fishery. As noted above, the advantage of using a mathematical programming model in this case is that effort allocation is endogenous, and hence changes in effort allocation arising from changes in management can be estimated. This is more realistic than many simulation models that assume a constant effort allocation or impose an exogenously determined allocation based on a priori expectation of effort reallocation following management changes.

The objective function of the model is the maximization of total fishery gross margins (TOTGM), given by

\[ TOTGM = \sum_k (REV_k - COST_k) \]

where \( REV_k \) is the total revenue in sector \( k \) and \( COST_k \) is the total variable costs in sector \( k \). The total revenue in each sector is given by

\[ REV_k = \sum_i \sum_m \pi_{i,m} LAND_{i,k,m} \]

where \( \pi_{i,m} \) is the price received for species \( i \) in month \( m \), and \( LAND_{i,k,m} \) is the quantity of species \( i \) landed in month \( m \) by boats in sector \( k \). The total variable cost of fishing for each sector is given by

\[ COST_k = FUEL_k + CREW_k + MARK_k \]

where \( FUEL_k \) is the fuel cost incurred by boats in sector \( k \), \( CREW_k \) is the payments to crew in sector \( k \) and \( MARK_k \) is the marketing costs incurred by sector \( k \). The estimation of these costs is given in Eqs. (4)–(6):

\[ FUEL_k = \beta_k \phi_k \sum_g \sum_m DAYS_{k,g,m} \]

\[ CREW_k = \rho_k REV_k \]

\[ MARK_k = \mu_k REV_k \]

where \( DAYS_{k,g,m} \) are the number of days employed by a boat in sector \( k \) fishing for species group \( g \) in month \( m \), \( \beta_k \) is the number of boats in sector \( k \), \( (k) \) is the average fuel cost per day for boats in sector \( k \), \( \rho_k \) is the crew share of revenue, and \( \mu_k \) is the marketing commission incurred by boats in sector \( k \). The groups are defined by the main target species caught in a particular area and depth range, but include all other species caught in the same area and depth range. Several species may be targeted simultaneously within a group. The proportional distribution of the species within these groups may change from month to month, but the general species composition within each group is fairly stable over the year.

The number of days that can be fished each month and over the year is also limited. The number of days that a boat in sector \( k \) can fish for group \( g \) is given in Eq. (7) and Eq. (8):

\[ \sum_k DAYS_{k,g,m} \leq \delta_k \]

\[ \sum_g \sum_m DAYS_{k,g,m} \leq \psi_k \]

where \( \delta_k \) is the maximum number of days a boat in sector \( k \) can fish in any month, and \( \psi_k \) is the maximum number of days that can be fished by a boat in sector \( k \) over the year.

There are no direct catch–effort relationships at the individual species level, as the species are targeted by some boats and caught as bycatch by others. Instead, the catch of species \( i \) by sector \( k \) in month \( m \) (\( CATCH_{i,k,m} \)) is estimated as a linear function of effort exerted on each species group (i.e. defined by area, depth and main species targeted) and catch per unit of effort of each species in each group, given by

\[ CATCH_{i,k,m} = \beta_k \sum_g \lambda_{k,m,g} \alpha_{g,i} DAYS_{k,g,m} \]

where \( \lambda_{k,m,g} \) is the average catch per day of a boat in sector \( k \) fishing for group \( g \) in month \( m \), and \( \alpha_{g,i} \) is the proportion of species \( i \) caught in group \( g \). As the relationship is linear, additional constraints on the amount of effort that can be applied by each sector each month are required in order to prevent effort expanding indefinitely. Given the complex nature of the species group structure, it is impractical to introduce a non-linear catch–effort relationship for each species. An advantage of this approach, however, is that changes in catch composition following either stock collapse or recovery can be simulated by varying \( \lambda_{k,m,g} \) and \( \alpha_{g,i} \).
The catch can be either landed or discarded, such that:

\[ CATCH_{i,k,m} = LAND_{i,k,m} + DUMP_{i,k,m} \]  

(10)

where \( LAND_{i,k,m} \) and \( DUMP_{i,k,m} \) are the quantity of catch of species \( i \) in month \( m \) by boats in sector \( k \) landed or discarded, respectively. Discarding occurs as the level of landings of each species is constrained by its TAC. However, as the species are caught jointly, it may be profitable to continue fishing and only land the species for which the TACs have not been filled, discarding the over-quota catch. Hence, the level of landings may be less than the level of catch. The quota constraint is imposed at the total landings level, such that:

\[ TLAND_i = \sum_{k,m} LAND_{i,k,m} \leq \tau_i \]  

(11)

where \( TLAND_i \) is the total level of landings of species \( i \) and \( \tau_i \) is the TAC.

4. Modeling the south east fishery

The model was applied to the Australian south east fishery in order to assess the benefits and costs of conservation of gemfish.

A clustering technique was used to identify unique species groups by main target species, location and depth. Sixteen species groups are incorporated into the model. Each group is made up of various combinations of the quota species, as well as a composite ‘other species’ component. Average monthly catch rates for each group were determined from logbook data over the period 1986 to 1991. Data after this period were excluded because of concerns over reliability of reported catches following the introduction of ITQs. Only data for 1991 were used to determine the monthly catch rate for species groups that were dominated by gemfish and orange roughy. This is because stocks of these species have been in decline since 1986. An average over the entire period would result in higher catch rates being incorporated into the model than occurred in the fishery.

In the model the fleet is split up into three sectors \( M \), the inshore trawler sector, the offshore trawler sector and the Danish seiner sector. Effort, expressed as days fished, was limited each month based on historical effort levels. The total level of effort that could be employed by a boat over the year was also limited based on historical levels. The species groups to which boats could apply effort was also limited, and was based on historical fishing patterns of the sectors. In some cases, only one boat group had access to a particular species group. For example, only offshore boats had access to species groups containing orange roughy, while only Danish seiners had access to the species group dominated by whiting. The set of available species groups for each sector was defined by an activity map, \( \Omega_{g,k} \), which had a value of 1 if a boat in sector \( k \) could harvest a particular group \( g \), or 0 (zero) if the boat in sector \( k \) could not harvest the group \( g \).

A usual feature of fishery optimization models is that effort becomes concentrated in the activities that are the most profitable. As operators do not have perfect information on which stocks are most productive at any one time, it was assumed that at least 1 per cent of a sector’s available effort would be applied to each stock to which it had access. In the case of the inshore fleet, which is spatially dispersed, it was also assumed that no more than 30% of the available effort in any one month would be applied to any one species group. These assumptions were based on the observation that, while effort was applied to all activities in the fishery, effort tended to be concentrated by the inshore sector on three or four activities in any one period. Effort is applied to an activity as long as the marginal benefit exceeds the marginal cost, including the opportunity cost associated with applying effort to an alternative activity. Anderson (1982) demonstrated that where fishing boats have a range of alternative uses, effort will be allocated between the alternatives such that the marginal returns in all cases are equal. With non-linear production functions, such a spatial equilibrium can be estimated using the model (see for example Pascoe, 1997), as the returns from each activity decrease as the amount of effort increases. As the model in this study has, by necessity, a linear production function, constraints on the amount of effort that could be applied in any activity in any period were required. Concentrating effort in one group may have resulted in a
substantial overestimate of the catch. Restricting effort ensures that, in the model, a variety of species are caught in each month and reduces the potential for catches to be substantially overestimated.

Given this, additional constraints were added to the model to restrict the maximum and minimum number of days that can be fished for any one group. These constraints are given by

\[
\text{DAYS}_{k, m, g} \geq v_k \quad \text{for } \Omega_{g, k} \neq 0 \\
\text{DAYS}_{k, m, g} \leq 0.3\delta_k
\]

for \( k = \text{inshore sector} \) and \( \Omega_{g, k} \neq 0 \).

\[
\text{DAYS}_{k, m, g} = 0 \quad \text{for } \Omega_{g, k} = 0
\]

where \( v_k \) is the minimum number of days fished by a boat in sector \( k \) on any one group in any month.

Eq. (13) applies to boats in the inshore sector only, and is imposed to prevent concentration of effort on any one species groups.

5. Model validation

The process of model verification and validation is not straightforward. Oreskes et al. (1994) claim that verification and validation of numerical models of natural systems is impossible. They argue that the existence of uncertain parameters in a model ensure that it can never be verified as a true representation of the system. While the ability of a model to replicate known events is often used to validate models, this does not prove that the model accurately represents the system (Oreskes et al., 1994). Nevertheless, the converse would hold: if a model cannot reasonably replicate known outcomes, then it definitely does not represent the system.

To test the ability of the model to replicate actual behavior, the model was used to estimate the landings of each quota species over the period 1991–93, and these were compared with actual landings. The model estimates were derived given the actual prices received each month in each year, fuel and other variable costs in each year, and restrictions imposed by the TACs in each year.

Prices for each species in each year were obtained from the Sydney and Melbourne markets. Fuel costs were estimated on a cost per day basis, and derived from logbook and survey data (ABARE, 1993b). Crew costs and marketing costs were estimated as a percentage of the total revenue in each year and were derived from surveys of the fishery over the time period examined (ABARE, 1993b).
The estimates of landings derived from the model were compared with recorded landings for each year (Table 1). ITQs were introduced on a broad basis in 1992, but TACs for orange roughy and gemfish (both eastern and western combined) were in place in 1991. It was found that limits on landings were also necessary on ling and royal red prawns to prevent overestimation of landings of these species in the 1991 simulation. For the 1992 and 1993 simulations, both model estimated landings and recorded landings were less than the TAC for most species. In the majority of these cases, the estimated landings were closer to the recorded landings than the TAC. The model does appear, however, to consistently overestimate landings of a number of high value but low quantity species.

The significance of any divergence between the level of landings estimated using the model and recorded landings is difficult to determine. The newness of the ITQ system, problems in allocation and the lack of an established quota trading market may have resulted in difficulties in quota leasing (Pascoe, 1993). This in turn may have resulted in less catch than might otherwise have been taken, even though quota may still have been available. Where quota could not be obtained by operators, catch of some species may have been discarded, resulting in recorded landings being lower than the true catch. An implicit assumption in the model is that quota can be transferred between boats within each sector of the trawl fleet and between sectors with no impediments or transactions costs.

The estimates of revenue, costs, and gross margin derived from the model were also compared with estimates derived from an economic survey of the fishery (ABARE, 1993b) (Table 2). In most cases, the model estimates of revenues and gross margins were higher than the survey derived estimates, but were generally of similar orders of magnitude. As one estimate is based on a calendar year (January–December) and the other on a financial year (July–June), there is no expectation that the estimates should be identical. The relatively higher revenues estimated for the offshore sector using the model is largely a result of using market prices for orange roughy. Most orange roughy is sold directly to processors and incurs little or no handling charges once it leaves the boat. As processor prices were generally not available at the level of detail required in the model, it was assumed that the market price would be similar to the processor price.
once the marketing charges had been deducted. As the revenue estimate is before marketing charges are deducted, the total revenue is substantially higher than the revenue received from processors. This higher revenue is largely offset by the higher costs, most of which is as a result of the higher marketing charges assumed in the model. As a result, the gross margin estimated using the model is similar to the survey estimate for the offshore sector.

6. Simulations and results

A number of different simulations were required in order to estimate the relative benefits of gemfish conservation. Conservation of the stock will only result in benefits to the industry (and by implication society, assuming no non-pecuniary benefits) if the future flow of benefits exceeds the costs in terms of foregone profits in the short term. The benefits of recovery and the costs in terms of foregone production need to be determined relative to the profits that could be earned by maintaining the status quo.

The model analyses took the form of three simulations. The first simulation represented the existing situation. This was to provide a benchmark against which the other scenarios could be assessed. The second simulation represented a fully recovered stock. This was to estimate the potential commercial benefits of stock recovery relative to the benchmark scenario (i.e. status quo). The third simulation involved imposing additional restrictions on the fishery that may be required to achieve stock recovery. This was to estimate the relative cost of the stock recovery.

In the simulation assuming that the stocks of gemfish had fully recovered, the catch rate of the spawning run group \( (l_{k, m, g}) \) was increased until an estimated 3000 tonnes of gemfish were caught in total in the fishery (including bycatch). It was assumed that real prices would not change from their base level as a result of the increase in gemfish supply. Increased landings of gemfish is likely to have some impact on price. However, as gemfish were exported prior to the decline in catches, it is likely that the gemfish price will be largely determined by the world price. Further, while gemfish prices have risen as a result of the decline in catches in recent years, this increase has been moderated by the existence of close (largely imported) substitutes.

In order to estimate the costs of gemfish recovery, it was assumed that catches of gemfish (as compared to landings of gemfish) would need to be reduced to zero to provide the greatest chance for the recovery of the stock. This was run to estimate the relative costs of taking further action to protect gemfish stocks. By setting a zero catch on gemfish, an implicit assumption is made that managers can prevent fishers from targeting groups in which gemfish are caught. The difference between the gross margins under the current situation (zero landings) and the zero catch scenario are indicative of the costs associated with stock recovery.

The fishery is subject to random fluctuations in abundance from year to year for most species. To incorporate this feature of the fishery, the model was run with different stock levels. As there are 16 species, there are potentially thousands of combinations that could be examined. Catch distributions \( (z_{g, i}) \) were randomly varied between three different levels — the base level and up or down by 20%. The model was run 1000 times with a random number determining whether a particular stock in a simulation should be increased, decreased, or remain the same.

The results of the model simulations are presented in Table 3. These represent the mean values of the 1000 stochastic simulations associated with each scenario. Variability in the results are indicated by the coefficients of variation (i.e. the standard deviation expressed as a percentage of the mean). The model is designed to estimate the possible range of catch and discarding rather than to forecast the actual level that may occur (this will ultimately depend on actual species abundance and market factors for the year). As a result, the relative values of the model outcomes are of more significance than the actual values themselves.

Preventing catches of gemfish by reducing the TACs of other species is estimated to reduce the fishery gross margin by about 25% each year relative to the situation under current manage-
Table 3
Estimated gross margins, catch and landings

<table>
<thead>
<tr>
<th></th>
<th>Current management (zero TAC)</th>
<th>Full recovery</th>
<th>Zero gemfish catch</th>
</tr>
</thead>
<tbody>
<tr>
<td>Gross margin ($m)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total fishery</td>
<td>34.7 (5)</td>
<td>39.5 (4)</td>
<td>26.0 (6)</td>
</tr>
<tr>
<td>Inshore trawlers</td>
<td>12.3 (6)</td>
<td>17.0 (5)</td>
<td>4.4 (5)</td>
</tr>
<tr>
<td>Offshore trawlers</td>
<td>21.2 (7)</td>
<td>21.1 (7)</td>
<td>21.6 (7)</td>
</tr>
<tr>
<td>Danish seiners</td>
<td>1.3 (14)</td>
<td>1.3 (14)</td>
<td>0 (0)</td>
</tr>
<tr>
<td>Total catch (kt)</td>
<td>33.1 (6)</td>
<td>34.7 (6)</td>
<td>21.4 (7)</td>
</tr>
<tr>
<td>Total landings (kt)</td>
<td>31.9 (6)</td>
<td>33.9 (6)</td>
<td>20.8 (7)</td>
</tr>
</tbody>
</table>

* Figures in parentheses are coefficients of variation.

ment (Table 3). This reduction in gross margin is not shared equally between the three sectors of the fishery, but is borne entirely by the inshore and Danish seine fleet. It may be necessary to prevent Danish seiners from operating altogether as gemfish is an unavoidable bycatch (albeit at very small levels) for these operators in all months of the year (Table 3). These results assume, of course, that such a policy could be enforced.

A fully recovered gemfish stock may only increase total annual gross margins by about 13% relative to the situation under current management (Table 3). The benefit of such a recovery will be borne solely by the inshore fleet.

From this, there are no net benefits from recovery of the gemfish stock to the Danish seiner fleet or the offshore fleet. There would only be a net benefit in reducing harvest now to the inshore fleet if the increased returns from the recovered stock could be realized in 4–6 years, depending on the discount rate (Fig. 1). The net benefits in Fig. 1 were derived by assuming that the lower returns were realized prior to stock recovery and the higher level of returns after stock recovery for an indefinite period. The net benefits to the inshore fleet were estimated for various periods of stock recovery, and ignore the costs to the Danish seine fleet. Including the costs to the Danish seine fleet would further reduce the benefits, requiring even a faster recovery period to break even. As the expected recovery period is not sufficient to offset the costs to even the inshore fleet, the benefits of recovery are clearly not sufficient to offset the costs to the fishery as a whole.

7. Discussion

The model of the fishery is fairly simplistic so the results have to be viewed with some caution. The assumption of the linear relationship between catch and effort in any one month may result in an overestimation of the catch. While similar assumptions are often employed in bioeconomic models (see, for example, Brown et al., 1978; Siegel et al., 1979; Vestergaard, 1996), a preferable approach would be to have the catch per unit of effort varying with the size and vulnerability of the stock. Unfortunately, the complexity of the interactions between species over the year and the absence of information on stock size, growth, and natural mortality of most species inhibited the development of a more complex model. Given the constraints placed on the amount of effort that could be employed in any one month on any one
group of species, it is likely that the catches are not substantially overestimated. From the model validation, the model was able to produce estimates of catch that were considered reasonable when compared with the recorded catches over the same period. However, the model could not be expected to provide reasonable estimates of catch and discarding in the future if the fleet size and behavior differed substantially from that which occurred over the period of the data.

While the model allowed fishers to change their allocation of effort between trawling activities in response to the landings restrictions, it did not allow operators to change fishing gear. This is an often seen response in fisheries managed under ITQs. However, in the case of the south east fishery, quotas were initially allocated only to trawler operators. While other gear types existed in the fishery (e.g. long line), these were subject to a different management regime. Permits to operate in the fishery were gear specific. This prevented trawler operators using alternative gear types to take their catch.

The model results suggest that reducing gemfish catch to zero would only produce benefits to the inshore sector if the stock could recover to a level that can sustain a catch of 3000 tonnes a year within 4–6 years, depending on the discount rate used. It could be assumed that Danish seiners would be exempt from the ban on catching gemfish as they catch only a small quantity, whereas the effects of such a ban would be to totally prevent them from operating. Therefore, the estimated costs to this sector can be effectively ignored as they would not, presumably, be incurred. The offshore fleet is likely to be largely unaffected by such a ban.

It may be many years before a normal age structure of the fishery emerges, and there remains a risk that the stock may never recover (Rowlings, 1995). The 1997 stock assessment indicated that the spawning biomass is only 40% of its target level (Bureau of Resource Science, 1998). Nicholls (1993) estimated that that five or six strong cohorts would need to be sustained for at least 4 or 5 years for there to be any viable fishery in the future. This would require a minimum of 9 years (i.e. the fifth cohort survives 4 years), assuming that the first of the strong cohorts appeared the first year of the additional conservation measures. Given this, it is not likely that the stock will recover in the necessary period to ensure a net commercial benefit from additional conservation measures.

Non-market benefits of conserving the gemfish stock (for example, existence value and option value) have not been considered in the above analysis. It is conceivable that the nonmarket benefits may be substantial, particularly if there is a high risk of stock extinction. Loomis and White (1996) suggest that, based on a meta-analysis of contingent valuation studies of a range of species, the benefits of species preservation outweigh even the most expensive preservation efforts. If the non-market benefits from conserving the resource are estimated to exceed the loss to the commercial fishers, there may still be benefits to society as a whole in conserving the resource. However, because this is likely to impose a cost on the commercial fishery, it is likely that some form of compensation will need to be paid.

The non-market benefits in this instance, however, may not be substantial. While a separate and unique stock, the eastern gemfish are the same species as western gemfish. Additional gemfish stocks are found around New Zealand. Most studies that have estimated non-use values of wildlife resources have examined the value of a species (e.g. Boyle and Bishop, 1987; Rubin et al., 1991), rather than stocks. Assuming that existence and option values are relevant at the species rather than stock level, then the existence value of a stock would only be significant if the species as a whole was threatened. The non-use value of an individual decreases as the number of individuals in existence increases (Eagle and Betters, 1998).

Because of the difficulties in estimating non-market values for endangered species, an alternative approach is to establish a safe minimum standard (Bishop, 1978). This is implicitly the approach adopted by the management authority. An assumption has been made that the existing stock is sufficient to prevent extinction. Further deterioration in the stock was to be avoided, and the zero TAC was designed to rebuild the stock to a higher, safer level. However, as noted previ-
ously, while the stock does not appear to be deteriorating further, there are few indications that rebuilding is taking place.

The analysis has not considered the relative costs of adopting some other form of management policy in order to protect gemfish. The only alternative policy could be to close the entire fishery down during the spawning run to prevent targeted and incidental catch of gemfish. This is likely to impose the same cost on inshore fishers as the zero catch of gemfish, because the gemfish ban effectively closes the fishery to inshore fishers during the spawning run. However, a total closure may also have implications for the offshore fleet (who are also capable but currently do not catch gemfish due the absence of quota) and Danish seiners who could be expected to be exempt from a ban. Hence the cost to the industry as a whole from a closure is likely to be greater than that given the current TAC system. Input controls and other technical measures have generally proved ineffective in conserving individual species within a multispecies fishery because of the technical interactions in the fishery, as demonstrated by the collapse of several major stocks around the world in recent years in input-control managed fisheries (in particular, the collapse of the Canadian cod stocks off the Grand Banks and several stocks of other species in the US north east fishery). Incentives can only be created to avoid particular species though either output controls or a penalty (tax) system for landing threatened species. Of these two options, output controls have generally been the most acceptable. Where penalties have been imposed, these have often proved to be less than the social value of the species, resulting in a sub-optimal outcome (Eagle and Betters, 1998).

8. Conclusions

The analysis highlights the potential conflict between economic and biological objectives of management. In many cases managers have an expectation that the conservation requirements of management can only be met when the stock is healthy and capable of producing either the maximum sustainable yield or maximum economic yield. In the case of a single species fishery, the economically optimal utilization of the resource generally requires a healthy fish stock (although as pointed out earlier, an optimal harvesting strategy from an economic viewpoint could involve the depletion of a slow growing stock). With a multispecies fishery, an optimal-harvesting strategy for a species may differ substantially from that of the strategy of a single species fishery (Anderson, 1975).

In most fisheries, it is very difficult to fish a stock to biological extinction. It is generally unprofitable to continue targeting a species at low stock levels, particularly if other species are available that are likely to provide a greater return per unit cost of effort. As a result, the non-market benefits associated with a stock are likely to be preserved even if a stock is fished to commercial extinction (the level at which it is no longer profitable to continue to harvest). In a multispecies fishery with imperfect gear selectivity for individual species, there will often arise situations where some stocks are exploited more heavily than others.

With most fisheries resources around the world currently being overexploited, and with the collapse of several major fish stocks in recent years (such as off the north east of the USA and off the east coast of Canada), the issue of conservation of single species within multispecies fisheries will become increasingly important. The introduction of output controls into these fisheries will be a necessary, but not sufficient, condition for conservation.

As a result, managers may need to review their conservation objectives. Fishery managers and society as a whole must decide if they want to conserve every species in the multispecies community, at what level, and at what cost. In some cases, such as the case of the eastern gemfish, the benefits may not outweigh the costs.

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References