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Application of an experimental approach to management of a tropical multispecies fishery with highly uncertain dynamics

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In most multispecies fisheries there is considerable uncertainty in selection of an appropriate model to represent the dynamics of the resource. Many model structures and/or parameterizations may be consistent with ecological principles and the available data. These models may have very different management implications, but may be impossible to distinguish by process-oriented research at reasonable cost and on a time frame of relevance to management. In some circumstances an adaptive experimental management regime can be economically beneficial by allowing empirical learning about resource dynamics and discrimination between alternative models. However, in any particular situation it must be determined whether an experimental regime is economically viable, and which management actions and research observations should be included in the regime. The development and application of an experimental management regime for the fisheries operating on a tropical fish community in northwestern Australia are described. The history of exploitation is summarized and a number of simple models are suggested which can mimic past changes observed in fish community composition. These models include interspecific, intraspecific, and habitat modification mechanisms. Possible socio-economic responses of the fishing industry to changes in the resource state are important to evaluation of a prospective fishing regime, and these are also modelled. The models are used to evaluate options for management of competing trap and trawl fisheries on the Northwest Shelf. It was found that if an experimental management regime were adopted for about 5–15 years (during which time key uncertainties in the resource dynamics and socio-economic responses could be resolved) a larger expected value could be obtained from the resource than if the existing management regime were continued. Some experimental management regimes also provided a greater expected value than would be obtained from immediate application of the management regime that is optimal for any of the individual resource models. Experimental management periods of less than about 5 y did not allow sufficient resolution of uncertainties to be worthwhile, and periods of longer than about 15 y often resulted in the costs of obtaining the additional resolution exceeding the value of the expected improvement in returns from the resource.

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Managing the species composition of a fish community is fundamental to management of a multispecies fishery because species composition broadly determines the types of fishing industry that are possible and usually determines the gross economic value of the catch. However, the dynamics of marine communities are very poorly understood. There is little agreement even about the general class of mechanisms that drives community dynamics (e.g. Diamond and Case, 1986; Sainsbury, 1988), and in the available data mean it is often impossible to assess the validity of inferences drawn about these mechanisms (Schaffer, 1981; Gardner *et al.*, 1982; Sugihara *et al.*, 1984). Typically, ecological theory can be used to derive a number of different hypotheses about mechanisms and dynamics that are all consistent with the observations of an exploited community. The available data usually cannot support objective discrimination between the competing hypotheses because of confounding among supposed control variables (e.g. fishing patterns, technological developments, species abundances, and environmental variables), the aggregation of space, time, and species in most fisheries data, and the absence of some species from the observations. These problems occur in most fisheries but are particularly severe in tropical regions, where the harvested communities are very diverse and fishery data sets are often incomplete and contain highly aggregated data. Indeed, the profound lack of scientific understanding of the mechanisms controlling community structure and dynamics is perhaps best illustrated by the absence of a widely accepted explanation for one of the most obvious ecological features on earth: the latitudinal gradient in species diversity (see Stevens, 1989, for a recent review), of which the high species diversity of tropical fish communities is a part.

The lack of a widely accepted scientific theory of community dynamics and structure has two important implications for fishery management aimed at controlling the species composition of a harvested community:

(i) *A priori* it is not at all clear to what extent the species composition of a harvested community can be controlled by fishery management. Assumptions about "controllability" are obviously the basis for any management action (or lack thereof), but the value of resolving uncertainty in these assumptions is little considered.
(ii) Management decisions concerning community structure will be, and have been, made on the basis of assumptions that would not be widely accepted by the scientific community of ecologists because there is no widely accepted scientific theory of community dynamics. This weak scientific basis for management decision-making provides ample opportunities for criticism of any decision by opponents of that decision. Furthermore, it is very unlikely that process-oriented scientific research will provide clear answers to the questions of community dynamics on a time frame that is relevant to most management decisions and at reasonable cost (if at all). This is a serious dilemma for management agencies whose decisions are open to public scrutiny, and is probably a major reason why management of the species composition of exploited communities is usually not made an explicit management aim. Instead, management objectives are usually focused primarily on single species and the economic performance of individual fisheries, and community composition is left to emerge "naturally" from the decisions based on these reasonably tractable considerations.

A notable exception to this focus of management is provided by the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR), which has among its aims the "maintenance of the ecological relationships between harvested, dependent and related populations" and "prevention of changes in the marine ecosystem which are not potentially reversible". However, these objectives have not resulted in management actions aimed at achieving a specified ecological community or a specified aspect of community dynamics. Rather, management actions have been based on single-species considerations and the few discussions of possible actions to meet the community level aims have simply highlighted the difficulties of reaching defensible management decisions when there is a high level of uncertainty in scientific understanding of resource dynamics (e.g. see CCAMLR 1989, pp. 41–45).

This paper describes an attempt to manage the species composition of an exploited fish community on the Northwest Shelf of tropical Australia, and particularly the way in which uncertainty was incorporated into the scientific evaluation of prospective management actions. The fish community has highly uncertain dynamics, and the socio-economic response of the fishing industry to changes in the fish community composition is also very uncertain. This paper provides the basis for scientific management advice given for the Northwest Shelf fisheries in 1985 and describes some of the subsequent management changes. Scientific advice to management was based on an experimental or actively adaptive approach (e.g. Bar-Shalom and Tse, 1976; Walters and Hilborn, 1976; Walters, 1986). In this approach, management actions are explicitly regarded as having dual aims: to provide an economic yield and to allow empirical learning about aspects of the dynamics of the resources that will lead to improved achievement of management aims. Prospective management actions are evaluated across alternative hypotheses about resource dynamics and monitoring schedules. The evaluations consider economic benefits, the value of distinguishing between the alternative hypotheses in leading to improved management decisions, the ability to distinguish between the alternative hypotheses on the basis of observations made under the proposed management action, and the cost of observations.

The paper is in three main sections: the first describes the ecological and fishery background to the Northwest Shelf resource and the management question that emerged; the second outlines the method used to evaluate management options; and the third identifies alternative models of resource dynamics, details the evaluation of some management options and describes the management actions taken. A simplification of the methodology is briefly described in Sainsbury (1988), but here the original calculations are described. In particular, the full set of models is described, details of model parameter estimation are provided, transient yields following experimental manipulation are included in the evaluations, and the economic uncertainties are treated more fully.

1. The Northwest Shelf and its fisheries

1.1. The Northwest Shelf

The continental shelf of northwestern Australia (between longitudes 114°E and 123°E to a depth of 200 m, Fig. 1) has a substrate consisting mostly of calcareous

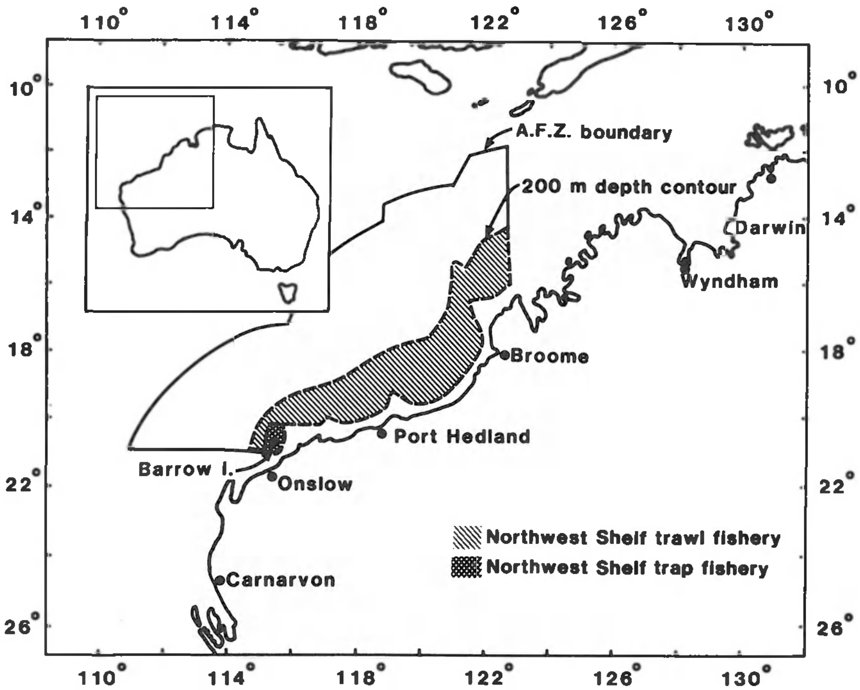

Figure 1. The 200-nmi Australian Fishing Zone, the 200-m depth contour and the Northwest Shelf. The Northwest Shelf region was defined by cluster analysis of the fish species composition recorded during research-vessel cruises in 1978–1980. The areas exploited by trap and trawl fisheries are also shown.

sands (Jones, 1973; McLoughlin and Young, 1985) with calcareous coral reef occurring only in restricted patches in depths less than about 30 m. The hydrographic regime is tropical (Wyrtki, 1961) and highly dynamic. Strong semidiurnal tides and internal tide waves cause high current speeds (Holloway, 1983a, b) and the vertical movement of isotherms through 50 m within some days (Holloway, 1987). The water column is well mixed between about May and October each year, and the annual temperature range is 21–26°C near the seabed at about mid-shelf (e.g. Holloway and Nye, 1985). Interannual variability of water temperature and sea-level height is strongly influenced by the El Nino–Southern Oscillation (Bye and Gordon, 1982; Pariwono *et al.*, 1986). Biological productivity is high (Tranter, 1962; Kabanova, 1968), and the diverse demersal fish community, comprising over 600 species, has strong Indo-West Pacific affinities (Sainsbury *et al.*, 1985). The Northwest Shelf supports a characteristic fish fauna that is clearly distinguishable from the tropical fauna to the east of 123°E by cluster analysis of the species composition of research vessel catches, and obviously different from the more temperate fauna of the west coast of Australia.

1.2. The fisheries of the Northwest Shelf

The histories of the fisheries on the Northwest Shelf are described in Sainsbury (1987). Briefly, they are:

(i) A Japanese trawl fishery from 1959 to 1963 targeted on fish of the genus *Lethrinus* (tropical emperor) and mostly operated between 116°E and 117°30′E (Fig. 2). *Lethrinus* comprised about half of the catch weight, and the catch rate did not decline during the period of the fishery (Suzuki *et al.*, 1964; Robins, 1969). However, there was a considerable change in the size composition of *Lethrinus*, with animals heavier than 0.6 kg disappearing from the catch after eight months of fishing (Suzuki *et al.*, 1964). This may have been due to the loss of *L. nebulosus* from the catch (Sainsbury, 1987). A total of about 7 600 h of trawling was exerted during the about 3-year duration of the fishery, for a total catch of about 16 700 t.

(ii) A Taiwanese trawl fishery from 1972 to the present takes a wide range of species. The retained catch mostly comprises the genera *Nemipterus* (threadfin bream), *Saurida* (lizard fish), *Lutjanus* (tropical snapper), and *Lethrinus* (Liu *et al.*, 1978). The fishery began in international waters and came under Australian jurisdiction in 1979 with declaration of the 200 nmi Australian Fishing Zone. The fishery has operated throughout the Northwest Shelf between depths of about 30 and 120 m, but effort has been particularly concentrated between 115°E and 120°E (Liu *et al.*, 1978). All fishing vessels are pair trawlers of 200–500 gross tons and have similar fishing power (Yeh and Chin, 1982); typically a pair of vessels 0.5 km apart tow a 30 m headline net on 850 m of wire cable. Codend mesh sizes of about 45 mm

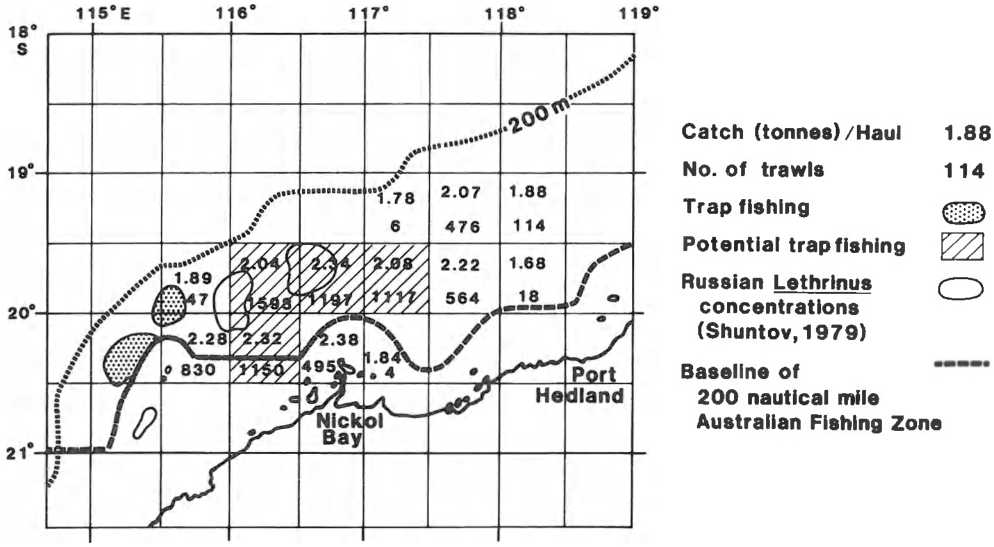

Figure 2. The western portion of the Northwest Shelf showing the 200-m depth contour, the coastal baseline of the Australian fishing zone, and fishery information from Japanese commercial fishing and Russian surveys. For each 30′ square the catch rate (upper figure) and number of trawls (lower figure) obtained by the Japanese fishery in 1959–1963 are given (after Robins, 1969); about half of all catches were of *Lethrinus*. The areas of high *Lethrinus* concentration identified during Russian research surveys in 1962–1973 (Shuntov, 1979) are also indicated.

were common until the introduction of a 60-mm minimum mesh size in 1981 (Sainsbury, 1984). A log-book programme to record fishing effort and retained catch by broad commercial category (generally at the generic level of taxon) was initiated by the National Taiwan University in 1974, and similar records have been collected by the Australian Federal Department of Primary Industry since 1979. There is no information on discards. Estimates of the catch and effort for 1972 and 1973 were compiled retrospectively by the National Taiwan University from the available fishing company log-books, but these data are not considered as reliable as those for later years (H. C. Liu, National Taiwan University, pers. comm.). Fishing effort increased rapidly to a peak of about 80 000 h of trawling in 1974, and subsequently declined to less than half this value (Table 1). Management of the pair trawl fishery since 1979 has been by total retained catch quota and minimum mesh size regulation. Both were calculated from combined analysis of a non-interacting, dynamic pool model for each of 10 major species groups (Sainsbury, 1984, 1987).

(iii) A domestic Australian trap fishery began in 1984, which targets fishes of the genera *Lethrinus* (mainly *L. nebulosus* and occasionally *L. choerorynchus*), *Lutjanus* (mainly *L. sebae*), together with the

Table 1. Fishing effort and retained catch (total and for four important genera, in tonnes) for the Taiwanese pairtrawl fishery on the Northwest Shelf. Data for 1972–1979 were obtained from annual reports of the National Taiwan University (Anon., 1972–1979) and for 1980–1985 from the Australian Department of Primary Industries and Energy database of fishing operations in the Australian Fishing Zone.

Year	Effort (h)	Total catch (T)	*Nemipterus*	*Saurida*	*Lethrinus*	*Lutjanus*
1972	500	272.0	39.2	16.1	29.9	33.5
1973	64 545	37 143.0	8 377.3	2 711.4	4 076.4	3 944.1
1974	79 860	31 256.3	7 934.7	4 276.4	2 653.3	2 784.8
1975	57 767	21 288.6	5 033.5	3 355.7	2 865.7	2 255.0
1976	46 592	18 929.2	4 530.9	3 061.9	1 840.7	805.3
1977	56 413	19 080.0	4 517.9	3 199.4	2 000.0	1 333.3
1978	40 998	14 488.3	3 431.5	1 951.9	1 701.2	1 351.3
1979	33 500	10 764.0	2 168.5	1 937.2	753.6	707.7
1980	36 173	12 512.6	3 663.6	886.3	1 707.1	1 332.1
1981	30 652	10 929.2	4 007.9	733.6	1 162.0	1 057.8
1982	38 991	13 418.1	3 884.4	558.4	1 855.1	1 886.6
1983	29 890	9 745.4	3 634.4	386.5	1 113.8	1 137.5
1984	32 743	8 899.7	3 286.5	525.1	791.9	1 372.4
1985	27 974	7 237.2	2 370.8	532.1	725.5	1 025.5

Table 2. The year, mean total catch rate ($kg\ h^{-1}$), mean catch rate for the main taxa, number of trawls (N), and net headline length (m) for nine research surveys of the central Northwest Shelf (116–119°E). for each of four taxa the mean catch rate ($kg\ h^{-1}$) is reported and the relative abundance (percentage of the total mean catch rate) is given in parentheses.

Vessel and source	Year	Total catch rate	*Nemipteridae*	*Saurida*	*Lethrinidae*	*Lutjanidae*	N	Headline
RV "Oshoro Maru" Anon. (1964) Masuda *et al.* (1964)	1962	206	2.3 (1.1)	11.3 (5.5)	42.6 (20.7)	73.3 (35.6)	15	38.2
RV "Oshoro Maru" Anon. (1965) Masuda *et al.* (1964) Suzuki *et al.* (1964)	1963	564	36.6 (6.5)	14.6 (2.6)	91.3 (16.2)	111.7 (19.8)	20	38.2
RV "Nagasaki Maru" Abe *et al.* (1967)	1966	316	23.7 (7.5)	31.6 (10.0)	88.1 (27.9)	69.2 (21.9)	14	33.6
RV "Hai Ching" Shu *et al.* (1972)	1972	134	9.9 (7.4)	7.5 (5.6)	33.1 (24.7)	40.6 (30.3)	54	34
RV "Hai Ching" Shu *et al.* (1973)	1973	92	5.8 (6.3)	7.2 (7.8)	21.7 (23.6)	29.1 (31.6)	73	34
RV "Oh Dae San Ho" Anon. (1980)	1979	235	62.0 (26.4)	19.3 (8.2)	24.6 (10.5)	14.6 (6.2)	24	38.5
RV "Hai-Kung" Chen *et al.* (1979)	1979	359	45.6 (12.7)	128.9 (35.9)	24.7 (6.9)	64.3 (17.9)	12	27.5
RV "Soela" 1/83 CSIRO unpubl. data	1983	327	40.8 (12.5)	30.1 (9.2)	22.2 (6.8)	37.9 (11.6)	62	25.9
RV "Soela" 4/83 CSIRO unpubl. data	1983	250	30.0 (12.0)	34.2 (13.7)	13.5 (5.4)	20.2 (8.1)	70	25.9

serranids *Plectropomus maculatus* (coral trout) and *Epinephelus multinotatus* (tropical cod). The traps are baited and operated to depths of about 80 m and in areas that have been subjected to little trawling (Fig. 1). There is no by-catch of *Saurida* and an extremely small by-catch of *Nemipterus* (M. Moran, Department of Fisheries and Wildlife, Western Australia, pers. comm.). There is interest in expanding this fishery, but to date fishing effort has been low and the total annual catch is about 300–500 t.

1.3. Changes in the resource community

The composition of the Northwest Shelf fish community since 1960 can be broadly determined from the results of various research vessel trawl surveys (Sainsbury, 1987). Nine surveys were conducted and reported in a manner that provides reasonably consistent information and survey coverage of the central part of the Northwest Shelf (116–119°E). All surveys were conducted by similar-sized stern trawlers using similar-sized nets, and compatible catch records are available at the generic level of taxon for *Saurida* and at the familial level for Lutjanidae, Lethrinidae, and Nemipteridae (Table 2).

Total fish catch rate showed no significant correlation with year of survey (Table 3). However, both the Lethrinidae and Lutjanidae significantly decreased in both relative and absolute abundance with time. Conversely, both Nemipteridae and *Saurida* show a significant increase in relative abundance during the same time. The

Table 3. Spearman's rank correlation coefficients (r_s) for the correlation between year of observation and the research vessel catch data give in Table 2. The relative catch rate is the catch rate for that taxon divided by the total catch rate. The probability (p) that $r_s = 0$ is given, and values statistically significant at $\alpha = 0.05$ are marked with an asterisk.

Variable	r_s	p
Total catch rate	0.1	0.14
Nemipteridae		
Catch rate	0.55	0.15
Relative catch rate	0.73*	0.04
Saurida		
Catch rate	0.55	0.15
Relative catch rate	0.75*	0.03
Lethrinidae		
Catch rate	−0.83*	0.01
Relative catch rate	−0.75*	0.03
Lutjanidae		
Catch rate	−0.75*	0.04
Relative catch rate	−0.73*	0.04

nemipterid and saurid data are quite variable and, while suggestive of an increase, neither show a significant increase in absolute abundance. However, more detailed data on the relative abundance of *Saurida* since 1978 shows that between 1978 and 1983 a small-bodied species (*Saurida* sp2 of Sainsbury *et al.*, 1985) has increased in numbers by about two orders of magnitude and a larger bodied species (*Saurida undosquamis*) has increased by about an order of magnitude (Thresher *et al.*, 1986). These changes in the community are also supported by information from research surveys conducted by the USSR between 1962 and 1973. While the available description of these surveys is insufficiently detailed to allow their inclusion in the above analysis, catches were reported to comprise 47% Lethrinidae, 12% Lutjanidae, and 3% *Saurida*, with no data being available for Nemipteridae (Anon., 1978). Overall, then, the fish community has maintained about the same biomass between 1962 and 1983, but its composition has altered from initially comprising 40–60% by weight of *Lethrinus* and *Lutjanus* combined and about 10% *Nemipterus* and *Saurida* combined, to comprising about 10% *Lethrinus* and *Lutjanus* and about 25% *Nemipterus* and *Saurida*.

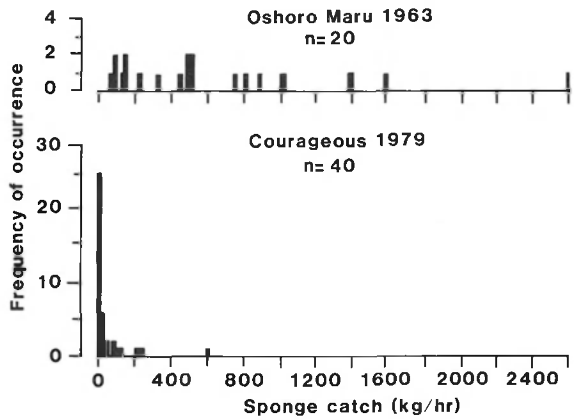

Figure 3. Sponge catch rates on the Northwest Shelf in 1963 (Anon., 1965) and 1979 (Sainsbury, 1987). Catch rates differ significantly (Mann-Whitney $U = 763$, $p < 0.01$).

The demersal habitat on the Northwest Shelf also changed in the period between these research surveys. Most of the pre-1972 research surveys reported large catches of large epibenthic fauna (mostly sponges, alcyonarians, and gorgonians), particularly from the area between 115°E and 120°E (Anon., 1965; Shu *et al.*, 1972; Shu *et al.*, 1973), and fishery inspectors observed large sponge catches in the early years of the pair trawl fishery (C. Ossel, Department of Fisheries and Wildlife, Western Australia, pers. comm.). Comparison of the sponge catches from 20 trawl hauls in 1963 (Anon., 1965) with those from 40 hauls in 1979 (Sainsbury, 1987) shows a significant reduction in sponge catch rates (Fig. 3). It is presumed that the reduction in epibenthos is due to its incidental removal by trawls. A crude measure of the use fish make of the habitat provided by the large epibenthos compared with the open sand habitat was obtained by mounting a 35 mm camera on the headline of the research trawl and taking photographs of the demersal habitat and fish in the path of each trawl. About 80 equally spaced photographs were taken on each of 108 randomly located trawls in the central part of the Northwest Shelf (116°E–119°E) in 1983. The photographs were scored for the presence and absence of large epibenthos (>25 cm along the longest axis) and the main genera of fish. From these data the probability of finding each fish genus in the presence and in the absence of the large epibenthos habitat was calculated (Table 4). This is a conservative method for detecting the pattern of habitat usage, because herding of fish by the trawl will result in any natural pattern being disturbed so that fish are likely to be observed in habitats where they would not naturally occur. This conservative bias is expected to be particularly severe in detecting usage of small, relatively isolated patches of habitat such as patches of large epibenthos. Despite this, however, the observations showed that *Lethrinus* and *Lutjanus* have a higher probability of occurring in the demersal habitat containing

Table 4. Frequency of occurrence and probability of occurrence of each of four fish genera in the presence and absence of large (25 cm) epibenthic organisms. The standard error of each probability of occurrence is given in parentheses. Estimates are based on photographs taken at regular intervals from the trawl headline during 108 randomly located research trawl hauls between 116°E and 119°E in 1983. Each photograph includes about 4 m^2 of sea floor. Standard errors were calculated assuming each photograph provides an independent observation of a binomial process, and so may be underestimated.

	Large benthos present		Large benthos absent	
Fish genus	Number of frames with this genus	Probability	Number of frames with this genus	Probability
Nemipterus	26	0.022 (0.004)	340	0.047 (0.002)
Saurida	110	0.092 (0.008)	999	0.138 (0.004)
Lethrinus	25	0.021 (0.004)	21	0.003 (0.001)
Lutjanus	9	0.008 (0.006)	22	0.003 (0.001)
	Total frames 1199		Total frames 7229	

large epibenthos than in the habitat without large epibenthos, while *Nemipterus* and *Saurida* have a higher probability of occurring in the open sand habitat (Table 4). If this habitat usage pattern remains constant it is clear that alteration in the proportion of the sea floor occupied by large epibenthos would cause an alteration in the composition of the fish community.

1.4. The management issue

In the decade following declaration of the Australian Fishing Zone management of the pair trawl fishery utilized a minimum mesh size and a total catch quota to maximize the yield of the species groups commonly retained by the pair trawl fleet (Sainsbury, 1984). Obtaining the maximum total catch in a mixed species trawl fishery will almost always result in overfishing the less productive species groups (in this case *Lutjanus* and *Lethrinus*) to obtain good yields of the more productive species groups (in this case *Nemipterus* and *Saurida*). This strategy reduces the proportionate abundance of the less productive species groups. Changes in relative abundance among the groups *Lutjanus*, *Lethrinus*, and *Nemipterus* are of little consequence to the pair trawl fishery, because all have similar values on the Taiwanese market (Liu and Lai, 1980). However, changes in the abundance of these fish groups are very important when development of the Australian trap fishery is considered.

The domestic fishery relies heavily on *Lethrinus* and *Lutjanus*, because *Nemipterus* has no net value in the Australian fishery. Consequently the trap fishery is restricted to the small areas where trawling has been light and the desired fish groups are still abundant. The historical fish community on the Northwest Shelf would have provided a suitable and large resource for the domestic fishery, but this fish community no longer exists. There is little prospect of significant development of the trap fishery with the present fish community composition. Furthermore, it is not certain that social and economic conditions in this remote part of Australia would allow expansion of the domestic fishery even if the resource had its historical composition, whereas it is known that the pair trawl fishery can generate income to Australia from the existing resource community.

The aims of management include maximizing the benefit to Australia from the resource, preferably by maximizing the involvement of the domestic fishing industry (Anon., 1985). The available data suggest that achieving these aims may require recovery of the species composition to its former dominance by *Lethrinus* and *Lutjanus*. However, the poor state of knowledge about the ecology and dynamics of the resource make prediction of the resource's response to major alterations in the management regime highly uncertain. In addition, there are substantial uncertainties associated with the social and economic factors that determine the response of the fishing industry to any recovery the resource may show. Questions relating to the management of the competing trap and trawl fisheries are:

- Can the historical composition be recovered?
- Is it worth trying to recover the historical composition (i.e. do the rewards justify the risks and costs)?
- If recovery is attempted what is an appropriate strategy to follow?

2. Evaluation of management options

2.1. The general approach

Scientific management advice in fisheries usually has an implicit assumption of "certainty equivalence" (e.g. Bar-Shalom and Tse, 1976). A scientific group reviews the data and somehow selects one model structure and one parameterization of that model that is thought to be scientifically "best". The implications of this one "best" model/parameterization are then examined with respect to the management aims, strategies, and tactics determined by the resource managers. The management actions considered feasible and desirable are then selected by the managers and applied to the whole of the resource as though this "best" model/parameterization were true. The selection of the best scientific model is often accompanied by lengthy discussion of uncertainty and confidence bounds, but this rarely influences the management decision-making process or the final management decision. In addition to being "certainly equivalent", this approach to management is also usually "passively adaptive". It relies upon errors in the management process or unplanned external events to provide the contrasts in the control variables that are necessary to identify the important processes and allow precise estimation of model parameters (Walters and Hilborn, 1978). Good examples of this passive adaptation occurred when fishing effort was reduced in the North Sea during the two World Wars. The reduction in fishing effort during World War I demonstrated the reality of overfishing, and changes in the yield after the reduction of effort during World War II provided empirical support for the method of yield per recruit analysis of growth overfishing (e.g. Beverton and Holt, 1957; Smith, 1988).

Under some circumstances an approach based on passive adaptation and certainty equivalence may provide close to optimal policies (e.g. Walters and Hilborn, 1978), but it appears unlikely that this is generally the case for resources with highly non-linear dynamics or when there is considerable uncertainty about the basic structure of the appropriate model of resource dynamics. Furthermore, such an approach usually treats the whole resource as one unit, so that all of the resource is at risk if an error is made. Possible control variables (e.g. population sizes, temporally varying environmental variables) are also confounded under this approach. Consequently, future observations of the resource have little power to discriminate hypotheses or allow recognition of suboptimal situations.

The prospect of avoiding the uncertainties in resource management by identifying the key features of the existing dynamics of a complex biological community through a process-oriented research programme seems remote. Even if achieved, it would not be certain that the identified features would remain the key factors under the novel perturbations provided by a changed fishing regime.

From these considerations it is concluded that community level management is largely empirical, even if this is not explicitly recognized, and that there is possible benefit in trying to increase the efficiency of that empiricism while maintaining revenue from the resource (Sainsbury, 1982, 1988). In particular there is a need to develop a reasoned empiricism, and avoid blind trial and error, by formally evaluating the performance of prospective management actions across alternative models while taking explicit account of future learning and the identification of policy improvements. This approach is sometimes called "actively adaptive" (e.g. Bar-Shalom and Tse, 1976; Walters and Hilborn, 1978; Walters, 1986) because it includes the possibility of taking management action that intentionally increases the contrast between key variables, so as to permit identification of important processes and therey increase the precision of model parameter estimates.

2.2. Assessment of Northwest Shelf

2.2.1. *Assessment method*

A framework for evaluating actively adaptive management is provided by Walters and Hilborn (1980) and Walters (1986), and a simple version of this methodology is applied to the Northwest Shelf. This evaluation examines a limited number of long-term management regimes (U_k, $k = 1$ to m) that are regarded as feasible, across a series of alternative models of the resource (M_j, $j = 1$ to n). At present time, T, models of resource dynamics that are consistent with the historical data are identified. The U_k examined here are the fixed harvesting regimes that are close to optimal for each of the M_j.

The expected economic value obtained from immediate application of one of the U_k regimes is compared to the expected value of first attempting to learn more about the relative credibility of the alternative hypotheses and then selecting the long-term management regime. It is of interest to determine whether or not additional learning is of economic value, and if so what combined harvesting and observation regime gives the greatest economic return. In the evaluations performed here, learning is considered to occur only during a "learning period" of duration t (i.e. the interval $T,T + t$) during which some trial fishing regime W is applied and various observations of the system are made. After the learning period, at time $T + t$, the observations are used to update the relative credibility of each of the alternative models, and the long-term regime that now appears most appropriate is selected and applied to the resource. The annual revenue during the learning period, including costs of the observations, and under the chosen U_k is calculated. This revenue flow is expressed as a present value by applying an economic discount rate to future revenues (e.g. Clark, 1976). The expected present value from the resource, under both the learning period regime W and the U_k chosen at time $T + t$, is calculated across all the models using the relative probability placed on each model at time T. These values are used to find a combination of W and t that results in selection of the most appropriate U_k at $T + t$ while also providing an acceptable overall economic return from the resource.

An ideal W regime would allow correct identification of each model if that model were true, would provide high revenues during the learning period, would not compromise the revenue from any of the regimes that might be selected at the end of the learning period, and would not involve expensive observations. The expected economic return from actively adaptive management in this context is the result of "trade-offs" between the revenue generated during the learning period, the costs of observations made during the learning period, the discrimination between models achieved from the observations, and the long-term economic value resulting from the achieved model discrimination.

The first step is to assign relative probabilities to each of the proposed models of resource dynamics. At time T all models are consistent with historical observations and can be reasonably parameterized with the historical data set, and so all models are assumed to have equal probability at time T (i.e. $P_T(M_i) = 1/n$).

The next step is to determine the relative probabilities placed on each model after application of the trial policy W for the learning period t. The probabilities placed on each model are updated on the basis of the observations made during this period. For trial regime W and model M_j, data sets $O_{j,W}$ are generated by Monte-Carlo simulation of the observation process (i.e. sampling) and the behaviour of model M_j under policy W during the interval $(T,T + t)$. The $O_{j,W}$ are examples of the data that might be available at time $T + t$ if model j is true. For each data set, the likelihood that it was generated by each of the models is calculated (i.e. the likelihood, $L(O_{j,W}|M_i)$, when model M_i is fitted to observations $O_{j,W}$). The probability placed at the end of the learning period on model M_i when model M_j is true is then calculated from Bayes' theorem.

As used here Bayes' theorem provides a method of calculating the conditional probability placed on a hypothesis from an initial statement of the probability of that hypothesis and some observed outcomes of trials (see Hays and Winkler, 1971). For example, Bayes' rule could be used to calculate how the probability placed on alternative hypotheses about the bias in a coin would change as a result of observing a number of throws. Two hypotheses are formulated about the probability of a coin returning a head on being thrown: H_1 being that the probability of a head is 0.5 and H_2 being that the probability of a head is 0.8. Because most coins are fair, it is initially thought that H_1 has probability 0.75 of being correct and that H_2 has probability 0.25. These are known as the prior probabilities placed on the hypotheses. If three throws all return heads, then Bayes' theorem allows calculation of the probability placed on each hypothesis conditional on the observed outcome of the throws. The updated probability on H_1 is

$$P(H_1 \text{ true}|\text{observations}) = \frac{P(H_1)P(\text{observations}|H_1 \text{ true})}{P(H_1)P(\text{observations}|H_1 \text{ true}) + P(H_2)P(\text{observations}|H_2 \text{ true})}$$

$$= 0.75 \times 0.5^3/(0.75 \times 0.5^3 + 0.25 \times 0.8^3)$$

$$= 0.423$$

and similarly the updated probability on H_2 is 0.577.

A slightly different application of Bayes' theorem can be used to calculate the probability placed on the alternative models of resource dynamics at time T + t, given the observations made during the learning period t, under each possible model and treatment of the resource during T. In this application

$$P_{T+t}(M_i|W,M_j) = \frac{P_T(M_i)L(O_{j,W}|M_i)}{\sum_{k=1}^{k=n} P_T(M_k)L(O_{j,W}|M_k)}$$

If W is perfectly informative then $P_{T+t}(M_i|W,M_j) = 1$ for $i = j$ and zero otherwise, while if W is totally uninformative then $P_{T+t}(M_i|W,M_j) = P_T(M_i)$.

For each model the present economic value from the resource under each combination of learning period policy and subsequent long-term policy can be calculated. For model M_j, the present value of applying policy W during (T,T + t) and then applying policy U_k is

$$V(U_k|W,M_j) = \sum_{v=0}^{v=t} R_{T+v}(W|M_j)\Phi^V + \sum_{V=t+1}^{V=\infty} R_{T+v}(U_k|M_j)\Phi^v$$

where $R_{T+v}(W|M_j)$ is the annual net economic return (including observation costs) from policy W applied to model M_j in year T + v, and Φ is a discount rate factor.

If at time T + t the decision to follow a particular U_k is made, then the data available from true model M_j will be interpreted as being due to model M_i with probability $P_{T+t}(M_i|W,M_j)$. This probability will then be erroneously associated with the annual value from the resource under U_k when model M_i is true. At T + t, perceptions of the value of the resource under the alternative long-term management regimes will be influenced by such errors, and selection of a management regime at T + t will be based on an "apparent value" made up of the revenue expected from each management regime when applied to each model and the relative probability placed on each model at that time. Here it is assumed that the resource manager is risk neutral (e.g. Lewis, 1982, p. 19) and will choose the U_k with the greatest expected value. For model M_j true the chosen U_k will maximize the "apparent value"

$$AV_{T+t}(U_k|W,M_j) = \sum_{i=1}^{i=n} P_{T+t}(M_i|W,M_j) \times \sum_{v=t+1}^{v=\infty} R_{T+v}(U_k|M_i)\Phi^{v-t-1}.$$

The $P_{T+t}(M_i|W,M_j)$ are random variables because $O_{j,W}$ is influenced by random processes such as sampling. A matrix of probabilities, $Q_{T+t}(U_k|W,M_j)$, can then be defined with elements giving the probability of selecting each of the U_k at time T + t when W is applied and model M_j is true. $Q_{T+t}(U_k|W,M_j)$ is calculated by using repeated simulations of the observation process to give data sets $O_{j,W}$, calculation of $AV_{T+t}(U_k|W,M_j)$ for each data set, and simulation of the decision process. The expected value from the policy with regime W for period t across all models is then

$$E[V(W,t)] = \sum_{j=1}^{j=n} P_T(M_j) \sum_{k=1}^{k=m} Q_{T+t}(U_k|W,M_j) \times V(U_k|W,t,M_j).$$

To this stage in the analysis it is assumed that any W or U_k can be successfully applied. In the case of the Northwest Shelf it is not at all clear that fisheries and management regimes other than the existing one are viable. The Australian fishing industry never attempted to establish fishing operations in the region prior to development of the foreign trawl fishery, presumably for socio-economic reasons; feasibility trials of Australian fish trawling soon after declaration of the 200 nmi fishing zone proved to be uneconomic (Anon., 1980); and the trap fishery is very small and would require a major increase in infrastructure to handle the ten-fold increases in catches implied by some of the U_k. Indeed, a major reason for licensing operation of a foreign fishery on the Northwest Shelf was the perception that the

Australian fishing industry could not economically harvest and market the available yield in this remote and sparsely populated part of Australia.

Consequently assessment of the fishery options for the Northwest Shelf resource needs to recognize that attempts to apply W and U_k may not be successful. In particular, social and unknown economic constraints may prevent the domestic trap fishery from expanding even if the historical species composition of the resource were restored. Failure of the trap fishery to expand was thought to be the most likely cause of failure to achieve a stated W or U_k. Other causes of failure to achieve a stated W or U_k are conceivable, such as the inability to enforce a trawl closure, but in this analysis only fishing regimes that attempt expansion of the trap fishery were considered to involve the possibility of failing. If expansion of the trap fishery is successful then the target fishing mortality specified by the W or U_k is achieved, and if unsuccessful the catch does not increase above the presently observed annual catch of 300 t.

Observation of the success or failure of the domestic fishery to expand during the learning period will alter perceptions of the possibility that the trap fishery will ever develop. These altered perceptions affect the "apparent value" of each U_k, and so will influence how long the opportunity to expand is made available to a fishery that shows no sign of expansion. If foreign access fees are foregone in providing the opportunity for the trap fishery to expand, then the success or failure of the trap fishery to expand will also affect the economic return from the resource. While the course of development of the trap fishery could be examined using a detailed economic model of fishery development, a simple *ad hoc* method was adopted here. The economic values obtained from success or failure of the regimes attempted are represented by the subscripts + for success and − for failure. So if both U_k and W fail, then the economic value obtained is represented by $V({}_{-}U_k|{}_{-}W,M_j)$, while the value of successful application of both U_k and W is represented by $V({}_{+}U_k|{}_{+}W,M_j)$.

Three probabilities must be specified to allow incorporation of this *ad hoc* treatment of the socio-economic responses into the assessment of (W,t).

(i) S_T, the probability that the trap fishery can expand.
(ii) S'_t, the probability that a trap fishery will have shown evidence of expansion by t years, given that expansion is possible.
(iii) S''_t, the probability that a manager who observes t years of non-expansion of the trap fishery places on that fishery ever expanding.

S_T and S'_t determine the probabilities of the various outcomes of applying W then applying U_k; the probability of ${}_{+}W$ followed by ${}_{+}U_k$ is $S_TS'_t$; the probability of ${}_{-}W$ followed by ${}_{+}U_k$ is $S_T(1 - S'_t)$; and the probability of ${}_{-}W$ followed by ${}_{-}U_k$ is $(1 - S_T)$. It is considered that ${}_{+}W$ followed by ${}_{-}U_k$ (which corresponds to successful expansion during the learning period then failure to expand after the learning period) cannot happen. Guidance on appropriate values for S_T and S'_t could perhaps be obtained from detailed economic analysis of the fishing industry. However, these probabilities are the result of numerous micro-economic and social decisions by companies and individuals, and it is notoriously difficult to determine such quantities. Here it is assumed that $S_T = 0.5$, giving development and failure an equal chance, and $S'_t = 1 - \psi^t$ with $\psi = 0.7$, because it seems reasonable to assume that signs of expansion would be about 80% probable to occur after 5 years if they are ever to occur. There is no necessary connection between S''_t and the real chance of ultimate expansion of the trap fishery, because S'' relates to management perceptions rather than fact. However, it is assumed here that the manager has knowledge that is consistent with the processes actually occurring, so that S''_t is the conditional probability that expansion will ultimately occur given that t years of non-expansion have been observed, i.e. $S''_t = S_T(1 - S'_t)/(1 - S_TS'_t)$. Even in the highly simplified decision-making process considered here, the manager's perceptions have a major influence on the decisions made, and it is interesting to note that irrational perceptions (i.e. those that are not based on evidence and that favour or discount certain outcomes) can give greater value from the resource than rationally based perceptions if the irrational perceptions are correct but can give a much lower value if wrong. It is difficult to improve on a lucky guess based on a correct but irrational bias.

With these additional considerations, the "apparent value" of applying U_k at time T + t, on which the manager bases policy selection, is

$$AV_{T+t}(U_k|{}_{+}W,M_j) = \sum_{i=1}^{i=n} P_{T+t}(M_i|{}_{+}W,M_j)V({}_{+}U_k|{}_{+}W,M_i)$$

for calculation of $Q_{T+t}(U_k|{}_{+}W,M_j)$ after observation of successful expansion of the trap fishery during the learning period, and

$$AV_{T+t}(U_k|{}_{-}W,M_j) = \sum_{i=1}^{i=n} P_{T+t}(M_i|{}_{-}W,M_j)\left[S''_tV({}_{+}U_k|{}_{-}W,M_i) + (1 - S''_t)V({}_{-}U_k|{}_{-}W,M_i)\right]$$

for calculation of $Q_{T+t}(U_k|{}_{-}W,M_j)$ after observation of no expansion of the trap fishery during the learning period. Consequently, the overall expected value of W for time t is

$$E[V(W,t)] = \sum_{j=1}^{j=n} P_T(M_j) \times$$

$$\times \left\{ S_T \left[\sum_{k=1}^{k=m} S_t' Q_{T+t}(U_k|_+W,M_j) V(_+U_k|_+W,M_j) + \right.\right.$$
$$\left. + (1 - S_t') Q_{T+t}(U_k|_-W,M_j) V(_+U_k|_-W,M_j) \right] +$$
$$\left. + (1 - S_T) \sum_{k=1}^{k=m} Q_{T+t}(U_k|_-W,M_j) V(_-U_k|_-W,M_j) \right\}.$$

It is often useful to calculate the expected value of perfect information (EVPI, see Hays and Winkler, 1971; Walters, 1986) to indicate the upper bound on the management value of distinguishing the alternative resource models and to gauge the absolute performance of a particular learning period regime. The expected value of perfect resource information from W applied for time t is

$$\mathrm{EVPI} = \sum_{j=1}^{j=n} P_T(M_j) \{ S_T S_t' V(_+U^*_{k,j}|_+W,M_j) +$$
$$+ S_T(1 - S_t') V(_+U^{**}_{k,j}|_-W,M_j) +$$
$$+ (1 - S_T) V(_-U^{**}_{k,j}|_-W,M_j) \}$$

where $U^*_{k,j}$ is the U_k that gives maximum $V(_+U_k|_+W,M_j)$ and $U^{**}_{k,j}$ is the U_k that gives maximum $S_t''V(_+U_k|_-W,M_j) + (1 - S_t'')V(_-U_k|_-W,M_j)$. This is the expected value if W perfectly discriminated the resource models after t years. The EVPI from resolution of the uncertainties in both the resource dynamics and the expansion of the trap fishery is given by the above expression evaluated with $S_t' = 1.0$ and $S_t'' = 0.0$.

2.2.2. Models of the Northwest Shelf fish resource

Four models of the dynamics of the four main species groups comprising the Northwest Shelf resource were examined, each model reflecting a different interpretation of the available data. The models are very simple because the data do not permit estimation of many parameters. Model parameters were obtained for each model by separately examining the likelihood surface for the two sets of data available, the research vessel catch rates and the commercial catch and effort (Tables 1 and 2). Each model was expressed in difference equation form for biomass (B_T) and fishing effort (E_T) at time T,

$$B_{T+1} = B_T(f(B_T) - q_f E_T),$$

with the predicted commercial catch (C_T) being

$$C_T = q_f E_T B_T \epsilon$$

where q_f is the catchability coefficient for commercial vessels and ϵ is an error term such that $\log_e \epsilon$ has a normal distribution with mean zero and standard deviation 0.4. No allowance for discarding was made and the commercial effort data were regarded as being reliable. The predicted research vessel catch rate is given by

$$\mu_T = q_r B_T \gamma$$

where q_r is a research vessel catchability and $\log_e \gamma$ has a normal distribution with mean zero and standard deviation 0.34. The standard deviations for the error distributions were chosen in the belief that each research catch rate observation was probably within half to double the value implied by the true biomass, and that the recorded commercial catch observations were probably somewhat worse than this.

The likelihood of each model was calculated by fitting data to the non-equilibrium form of the model directly, beginning in 1960 and assuming that the populations were initially at equilibrium under no fishing effort. The models were fitted to the research survey and commercial data sets separately, with the log likelihoods for n observations being calculated from

$$-0.5n \log(2\pi\sigma_\epsilon^2) - (2\sigma_\epsilon^2)^{-1} \sum_{T=1}^{T=n} \log(C_T/qfE_T B_T)$$

for the commercial data set and

$$-0.5n \log(2\pi\sigma_\gamma^2) - (2\sigma_\gamma^2)^{-1} \sum_{T=1}^{T=n} \log(\mu_T/q_r B_T)$$

for the research vessel data set.

In most cases a single set of parameter values corresponding to the maximum of both surfaces could be found; in other cases the parameter set corresponded to the maximum if each surface was accepted so that two parameter sets were considered for some models. Each parameter set/model combination was treated as a different model in the evaluation of management options. The commercial catch data beyond 1979 for *Saurida* were not included in the analysis because of changes in commercial retention practices (see Thresher *et al.*, 1986).

Model 1

All species groups are controlled by "intraspecific" processes. This model represents the multiple single species approach to assessment.

Lethrinus and *Lutjanus* follow the difference equation for logistic population growth,

$$B_{T+1} = B_T[1 + r - (rB_T/K) - q_f E_T]$$

where r is the intrinsic population growth rate and K is the carrying capacity. Best estimates of the three parameters of this model are given in Table 5.

Nemipterus and *Saurida* increased in abundance with fishing, which, assuming only intraspecific population processes, suggests that the population growth rate depends upon population age structure. This might be due to high levels of cannibalism by old individuals which are subsequently removed by the fishery, or a highly domed relation between population egg production and recruitment combined with strongly age-dependent individual fecundities. A simple model of this is

$$B_{T+1} = B_T[1 + r - (a_T r B_T/K) - ((1 - a_T) r B_T/\partial K) - q_f E_T]$$

where a_T is the proportion of the population biomass at time T made up of animals younger than some critical age (T_{crit}) at which they strongly retard population growth and ∂ determines the strength of this retardation. Inclusion of a_T and ∂ is a crude method of allowing a production model to exhibit some age structure behaviour. From a simple age structured population model

$$a_T = \sum_{t=0}^{t=T_{crit}} \left[\exp\left(-Mt - q_f \sum_{i=T-t}^{i=T} E_i\right)(1 - \exp(-kt))^3\right] \div \sum_{t=0}^{t=\infty} \left[\exp\left(-Mt - q_f \sum_{\substack{i=T-t \\ i>1971}}^{i=T} E_i\right)(1 - \exp(-kt))^3\right]$$

with the values for the rate of natural mortality (M) and the von Bertalanffy growth coefficient (k) taken from Sainsbury (1984). The best estimates of the five parameters of this model (r, q_f, K, T_{crit}, and ∂) are given in Table 5. The model parameter values suggested by the commercial catch data were substantially different from those suggested by the research survey data, so each was used to give two different parameterizations of the model. Parameter set 1a of Table 5 provides the best fit of the model for *Nemipterus* and *Saurida* to the commercial data, and set 1b provides the best fit to the research survey data.

Model 2

Lethrinus and *Lutjanus* have intraspecifically controlled population growth, as in model 1, while the population growth of *Nemipterus* and *Saurida* is also negatively influenced by the combined biomass of *Lethrinus* and *Lutjanus*. Population growth for *Lethrinus* and *Lutjanus* is given by model 1, and for *Nemipterus* and *Saurida* is

$$B_{T+1} = B_T[1 + r - (rB_T/K) - (\alpha r W_T/K) - q_f E_T]$$

where α is an interaction coefficient and W_T is the combined abundance of *Lethrinus* and *Lutjanus* at time T. Parameter estimates for r, q_f, K, and α are given in Table 5. Parameter set 2a in Table 5 provides the best fit of the model for *Nemipterus* and *Saurida* to the commercial data, while 2b provides the best fit to the research survey data.

Model 3

Nemipterus and *Saurida* have intraspecifically controlled population growth, while the population growth of *Lethrinus* and *Lutjanus* is also negatively influenced by the combined biomass of *Nemipterus* and *Saurida*. Population growth for *Nemipterus* and *Saurida* is as in model 1, and for *Lethrinus* and *Lutjanus* is

$$B_{T+1} = B_T[1 + r - (rB_T/K) - (\alpha r W_T/K) - q_f E_T]$$

where W_T is the combined abundance of *Nemipterus* and *Saurida*. Parameter estimates are given in Table 5. The parameter set 3a in Table 5 follows from *Nemipterus* and *Saurida* having parameters 1a, while parameter set 3b follows from *Nemipterus* and *Saurida* having parameters 1b.

Model 4

The carrying capacity of all groups is determined by the amount of suitable habitat, and habitat abundance is altered by the physical effects of trawling. For all fish groups population growth is

$$B_{T+1} = B_T\left[1 + r - \left(rB_T \Big/ \left(\Delta \sum_{i=1}^{i=2} h_{T,i}\lambda_i\right)\right) - q_f E_T\right]$$

where $h_{T,i}$ is the proportion of the area occupied by habitat i at time t, λ_i is the relative density of the species group in habitat i, and Δ is a constant. Only two demersal habitats were considered, the first containing large (>25 cm) epibenthic organisms and the second not containing these organisms. The estimated probabilities of observing a fish in each habitat type (obtained from the photographic survey data in Table 4) were taken to be estimates of the λ_i. The $h_{T,i}$ were obtained from a simple model of the benthos in which the sea floor is considered to comprise a number of small, independent patches of size equal to the area covered by each survey photograph (about 4 m^2). In this model each patch has a fixed probability (s) of receiving a recruit each year and each benthic organism has a fixed probability (d) of dying each year. If it takes T_{25} years for the organisms to grow to 25 cm then the proportion of patches with one or more individuals of large benthos is

$$1 - \prod_{\phi=T_{25}}^{\phi=\infty} [1 - s \exp(-\phi d)].$$

If each trawl sweeps proportion p of the fished area and an individual aged ϕ is removed with probability r_ϕ on encounter, then the probability of an individual not being removed by trawling that year is approximately $\exp(-pr_\phi E_T)$. Assuming r_ϕ can be written as

$$r_\phi = r_{max}(1 - \exp(-c\phi)),$$

but restricting $r_{T_{25}}$ to be 0.95 of r_{max}, so that c can be expressed solely in terms of T_{25} and r_{max}, gives the proportion of patches with one or more individuals of large benthos at time T (i.e. $h_{T,1}$ in the difference equation for model 4) as approximately

$$h_{T,1} = 1 - \prod_{\phi=T_{25}}^{\phi=\infty} [1 - s \exp(-D_\phi)]$$

where

$$D_\phi = \phi d + p \sum_{k=1}^{k=\phi} r_{\phi-k} E_{T-k}$$

and

$$h_{T,2} = 1 - h_{T,1}.$$

A similar equation can be obtained for the proportion of patches with small benthos, and since both proportions were measured in 1983 this gives two equations in four unknowns, s, d, T_{25}, and r_{max}. The proportion of the area swept by one Taiwanese trawl, p, was $0.84/8 \times 10^4 = 1.05 \times 10^{-5}$. A value of $r_{max} = 0.5$ was considered reasonable from scuba and video observations of trawls in progress, and two values for T_{25} were taken from the literature (Harrison and Cowden, 1976), 6 years and 10 years. These data were used to estimate two parameter sets (s,d), each set corresponding to the growth rate implied by the one of the two T_{25} values. A separate set of parameters for the fish dynamics model (r,Δ,q_f) was then obtained for each (s,d) pair. The estimates of the five parameters of this model are given in Table 5. Parameter set 4a of Table 5 is for $T_{25} = 6$ years and set 4b is for $T_{25} = 10$ years.

2.2.3. *Assessment of management regimes for the Northwest Shelf*

Table 5 defines eight different model/parameter value combinations resulting from two parameterizations of each of four models. Models 1 and 2 imply a low yield to a trap fishery, because the historical decline of *Lethrinus* and *Lutjanus* is interpreted as indicating that these stocks have low productivity. Models 3 and 4 give a high yield to the trap fishery because trapping removes no benthos (which under model 4 allows return of high carrying capacities for *Lethrinus* and *Lutjanus*) and catches few *Nemipterus* and *Saurida* (which under some parameterizations of model 3 allows the *Nemipterus* and *Saurida* populations to decrease to their unfished levels, with consequent reduction in the negative influence they exert on *Lethrinus* and *Lutjanus* population growth).

Assessment of management regimes for the Northwest Shelf examined four different regimes during the learning period (i.e. the W in section 2.2.1), learning periods of up to 20 years, and four different long-term regimes (i.e. the U_k in section 2.2.1).

The long-term fishing regimes considered for possible implementation after the learning period were: U_1 a trap fishery for *Lethrinus* and *Lutjanus* with a fishing mortality of 0.1 (close to maximum sustainable yield for models 1 and 2), U_2 a trap fishery with fishing mortality of 0.2 (close to maximum sustainable yield for model 3), U_3 a trap fishery with fishing mortality of 0.6 (close to maximum sustainable yield for model 4), and U_4 continuation of trawling at the present trawling effort and no further development of the trap fishery.

The yield to the pair trawl fishery is about the same under all models and so the annual return to Australia from this fishery is taken to be the 1985 annual licensing and access fee of AUS\$ 0.5×10^6. Vessels in the trap fishery obtain an after-costs value of about AUS\$ 1142 per tonne of retained catch (M. Moran, Western Australian Department of Fisheries, pers. comm.), and this was used to calculate the annual value of the trap fishery catch from the resource under each model and fishing regime. Failure of the trap fishery to expand is taken to result in the present level of catch to the trap fishery, 300 t per year, irrespective of the resource model. The discount rate factor was assumed to be 0.95 throughout the analysis.

The learning period regimes examined were: W_A indefinite continuation of U_4 (i.e. no learning period, so $t = 0$, and continuation of the *status quo*); W_B immediate application of the U_k giving greatest expected value (i.e. $t = 0$ and application of the regime presently considered best); W_C continued trawling during a learning period of t years, then after the learning period the U_k giving greatest expected value is selected and applied (i.e. learning does not disrupt revenue flow or empirically explore trap fishery expansion); W_D a learning period of t years during which trawling is stopped for a number of years and trap fishing with $F = 0.2$ is attempted, then after the learning period the U_k giving greatest expected value is selected and applied (i.e. learning disrupts revenue flow but empirically explores trap fishery expansion); and W_E a learning period of t years in which half of the area is treated as for W_D and foreign access trawling is continued in the other half

Table 5. Parameter values for each of the four resource dynamics models used in assessment of management regimes. Different parameterizations of each model that were regarded as being equally acceptable for the available data are indicated by letters. The maximum equilibrium sustainable yield (MSY) is given for each species group (i.e. single-species maximum) for each model, and for models 3 and 4 the directed MSY is also given. MSY (directed) is the yield that would be available from *Lethrinus* and *Lutjanus* if they alone were subject to fishing mortality and harvesting did not modify the habitat, and is the yield available from a trap fishery. The units for K and MSY are tonnes, and for T_{crit} are years.

Parameter	*Nemipterus*	*Saurida*	*Lethrinus*	*Lutjanus*
Model 1a				
r	1.1	1.3	0.25	0.35
q_f	0.3×10^{-5}	0.4×10^{-5}	0.4×10^{-5}	0.5×10^{-5}
K	45 000	35 000	20 000	15 000
T_{crit}	5	5		
∂	0.1	0.05		
MSY	12 000	10 200	1 250	1 312
Model 1b				
r	1.5	1.4	0.25	0.35
q_f	0.7×10^{-5}	0.8×10^{-5}	0.4×10^{-5}	0.5×10^{-5}
K	300 000	60 000	20 000	15 000
T_{crit}	5	5		
∂	0.001	0.01		
MSY	70 000	15 700	1 250	1 312
Model 2a				
r	1.4	1.5	0.25	0.35
q_f	0.5×10^{-5}	0.4×10^{-5}	0.4×10^{-5}	0.5×10^{-5}
K	35 000	45 000	20 000	15 000
α	0.5	1.0		
MSY	12 300	16 900	1 250	1 312
Model 2b				
r	1.5	1.5	0.25	0.35
q_f	0.4×10^{-5}	0.4×10^{-5}	0.4×10^{-5}	0.5×10^{-5}
K	45 000	45 000	20 000	15 000
α	1.0	1.0		
MSY	16 900	16 900	1 250	1 312
Model 3a				
r	1.1	1.3	0.4	0.5
q_f	0.3×10^{-5}	0.4×10^{-5}	0.3×10^{-5}	0.4×10^{-5}
K	45 000	35 000	45 000	25 000
T_{crit}	5	5		
∂	0.1	0.05		
α			0.5	0.2
MSY	12 000	10 200	945	1 150
MSY (directed)			1040	1220
Model 3b				
r	1.5	1.4	0.45	0.5
q_f	0.7×10^{-5}	0.8×10^{-5}	0.3×10^{-5}	0.4×10^{-5}
K	300 000	60 000	25 000	20 000
T_{crit}	5	5		
∂	0.001	0.01		
α			0.3	0.2
MSY	70 000	15 700	1 070	1 180
MST (directed)			1 940	1 840
Model 4a				
r	1.2	1.2	1.6	1.2
q_f	0.5×10^{-5}	0.2×10^{-5}	0.7×10^{-5}	0.8×10^{-5}
Δ	550 000	250 000	900 000	1 400 000
MSY	7 730	10 300	1 130	1 320
MSY (directed)			5 890	2 820
Epibenthic organisms				
r_{max}	0.5			
T_{25}	6			
s	0.1332			
d	0.0655			

(*continued*)

Table 5. *Continued.*

Parameter	*Nemipterus*	*Saurida*	*Lethrinus*	*Lutjanus*
Model 4b				
r	1.1	1.1	1.4	1.2
q_f	0.5×10^{-5}	0.2×10^{-5}	0.6×10^{-5}	0.8×10^{-5}
Δ	750 000	250 000	950 000	1 400 000
MSY	9 690	9 490	1 000	1 280
MSY (directed)			6 220	3 090
Epibenthic organisms				
r_{max}	0.5			
T_{25}	10			
s	0.0906			
d	0.0320			

Table 6. The present value (millions of Australian dollars) obtained from each model (M) by immediately applying each of four possible long-term fishing regimes (U). $V({}_{+}U_k|M_j)$ gives the value of successful application of U_k to a resource obeying model M_j, and $V({}_{-}U_k|M_j)$ gives the value of a failed application. U_1 is a trap fishery with F = 0.1, U_2 is a trap fishery with F = 0.2, U_3 is a trap fishery with F = 0.6, and U_4 is continuation of the licensed trawl fishery. A failed trap fishing regime occurs if the fishery fails to expand to provide the target F, and in this event a constant annual catch of 300 t per year is assumed. A discount rate of 0.05 was used throughout.

	$V({}_{+}U_k\|M_j)$				$V({}_{-}U_k\|M_j)$			
M_j	U_1	U_2	U_3	U_4	U_1	U_2	U_3	U_4
1a	48.2	57.3	26.7	9.96	6.85	6.85	6.85	9.96
1b	48.2	57.3	26.7	9.96	6.85	6.85	6.85	9.96
2a	48.2	57.3	26.7	9.96	6.85	6.85	6.85	9.96
2b	48.2	57.3	26.7	9.96	6.85	6.85	6.85	9.96
3a	49.6	54.1	29.1	9.96	6.85	6.85	6.85	9.96
3b	60.1	84.6	32.9	9.96	6.85	6.85	6.85	9.96
4a	39.9	73.8	148.7	9.96	6.85	6.85	6.85	9.96
4b	38.7	71.2	137.1	9.96	6.85	6.85	6.85	9.96

(both halves are assumed to be isolated and have identical dynamics). W_A and W_B are degenerate cases (t = 0) and do not involve consideration of further learning, while for W_C, W_D, and W_E the learning period duration giving the greatest economic benefit is of management interest. During all learning periods, observations of the status of the resource are made by an annual research trawl survey. The survey is assumed to cost AUS\$ 0.3×10^6 (based on CSIRO survey costs) and to provide a relative index of the abundance of each of the four fish groups according to

$$\mu_T = q_r B_T \gamma$$

where, as before, B_T is the true abundance, q_r is the research vessel catchability as estimated from the historical data and $\log_e \gamma$ has a normal distribution with mean zero and standard deviation 0.34. Recent surveys appear much more precise than this, but the early surveys of the Northwest Shelf appear to contain variability of this magnitude and it is prudent to take a conservative view of sampling capabilities.

The present values associated with each of the U_k applied immediately to each model (i.e. at time T with no further learning) are given in Table 6. If it were known that the trap fishery could not expand, then the existing trawl fishery gives a higher present value from the resource than a trap fishery irrespective of which of the resource models is true. If trap fishery expansion is possible, then a trap fishery with F = 0.2 gives the greatest present value when models 1–3 are true, while a trawl fishery with F = 0.6 gives the greatest present value when model 4 is true. Distinguishing between models 1 through 3 has no management value, because they all imply the same management action, but there is considerable benefit from distinguishing model 4 from models 1–3.

The expected present values from each of the W examined are given in Table 7. For W_B, immediate application of a trap fishery with F = 0.2 (i.e. U_2) gives highest expected present value from the resource (AUS\$ 35.4×10^6). Immediate application of U_2 gives a larger expected present value than immediate application of U_3 if the probability of model 4 being true (i.e. $P_T(M_{4a}) + P_T(M_{4b})$) is less than about 0.33. The expected value of perfect information (EVPI) for instant resolution of the alternative resource models in 1985 is AUS\$ 45.8×10^6. The best expected value obtainable

Table 7. The expected present value (millions of Australian dollars) of different management strategies (W) for the Northwest Shelf fishery. Policy W_A is continuation of the existing licensed trawl fishery; W_B is to immediately apply one of four possible long-term regimes (U_1 to U_4 as described in the caption to Table 6); W_C is continued trawling for a period t, conducting annual resource surveys during t and using these data to select the best U_k at the end of the period; W_D is conducting annual resource surveys for period t, during which time all fishing is stopped for t_c years and a trap fishery with $F = 0.2$ is applied for the remaining $t - t_c$ years, and then using these data to select the best U_k at the end of the period; W_E is the same as W_D except that the learning period regime is applied to only half the area and the trawl fishery is continued in the other half. The expected value of perfect information (EVPI), with respect to distinguishing the alternative resource dynamics models, is also given for each strategy.

Strategy W_A		9.96		
Strategy W_B				
	U_k		$E[V(U_k)]$	
	U_1		27.2	
	U_2		35.4	
	U_3		31.8	
	U_4		9.96	
	EVPI		45.8	
Strategy W_C				
	t		$E[V(W_C,t)]$	EVPI
	0		35.4	45.8
	5		35.6	36.7
	10		29.7	30.7
	15		25.1	25.8
	20		21.2	21.7
Strategy W_D				
	t	t_c	$E[V(W_D,t)]$	EVPI
	0	0	35.4	45.8
	2	1	35.8	42.0
	5	1	36.8	40.9
	5	2	37.4	40.5
	10	2	37.2	38.7
	10	4	36.8	37.4
	15	2	37.1	37.2
	15	4	35.6	35.6
	20	2	36.3	36.3
	20	4	33.9	33.9
Strategy W_E				
	t	t_c	$E[V(W_D,t)]$	EVPI
	0	0	35.4	45.8
	2	1	35.9	43.1
	5	1	40.5	42.0
	5	2	40.6	41.6
	10	2	40.5	40.7
	10	4	38.9	39.2
	15	2	39.8	39.8
	15	4	38.4	38.7
	20	2	38.6	38.6
	20	4	37.2	37.2

under the present uncertainty is AUS\$ 35.4×10^6 (Table 7), and in principle the difference (AUS\$ 10.4×10^6) is the amount it would be worth paying to obtain instant resolution of the specified uncertainties in resource dynamics. W_C (continued trawling at the 1985 effort level throughout the learning period) is a very uninformative strategy. Further observations of the community under continued trawling provide little opportunity to recognize model 4 when it is true, and there is no opportunity to test empirically the expansion of the trap fishery. Strategy W_D, a mixture of closing trawling and attempting to expand the trap fishery, can give expected values that are greater than W_B for learning periods less than 15–20 years. Of the strategies examined, a short (about 2 years) closure of the trawl fishery followed by 3–13 years of attempted expansion of the trap fishery, provides close to the maximum expected value from the resource. Maintaining the trawl fishery on half of the Northwest Shelf while examining the trapping option on the other half during the learning period provides a higher expected value than treating the whole shelf as one unit. Longer learning periods, and in particular longer periods of trawl closure, provide greater resolution between the alternative models but the costs of obtaining this additional resolution exceed its value to management. Very short learning periods provide highly variable outcomes because of frequent misidentification of models, and so give relatively low expected values.

The analysis outlined here was provided to the State and Federal Fisheries management authorities. The State authorities saw merit in the suggested management actions, but the Federal authorities initially regarded the *status quo* as acceptable. Both authorities agreed on the goals of management of fisheries in the Australian Fishing Zone (Anon., 1985) – to obtain the greatest benefit from the resource and support domestic fishing industries – but differed in their assessment of how these goals should be achieved. Regional, organizational, political, and individual differences in emphasis on the inevitably multifaceted consequences of a management action are to be expected, and illustrate the difficulty in specifying an objective criterion for decision-making in fisheries management. However, partly for the reasons outlined here and partly for other reasons, the Federal and State fishery managers cooperated to close a portion of the Northwest Shelf to foreign trawling from 1985 (Fig. 4). They similarly cooperated to close an additional area to foreign trawling from 1987 (Fig. 4), thereby providing three areas with contrasting fishing regimes.

Annual surveys of the fish community and epibenthos on the Northwest Shelf west of 119°E have been conducted by CSIRO since 1985. The abundance of fish in the areas closed to trawling increased in the years following the closures, and in 1988 domestic trawlers began to operate in these areas. This development was not anticipated in the 1985 assessment of options for the Northwest Shelf. Domestic trawling has subsequently been restricted to east of 116°45′E so as to maintain non-trawled areas for resource recovery and development of the trap fishery. It is anticipated that the western boundary of the foreign access zone will be further moved eastward to 120°E from 1990 to reduce interaction between the foreign and domestic trawl fisheries. Increase in the domestic trawling effort in the area

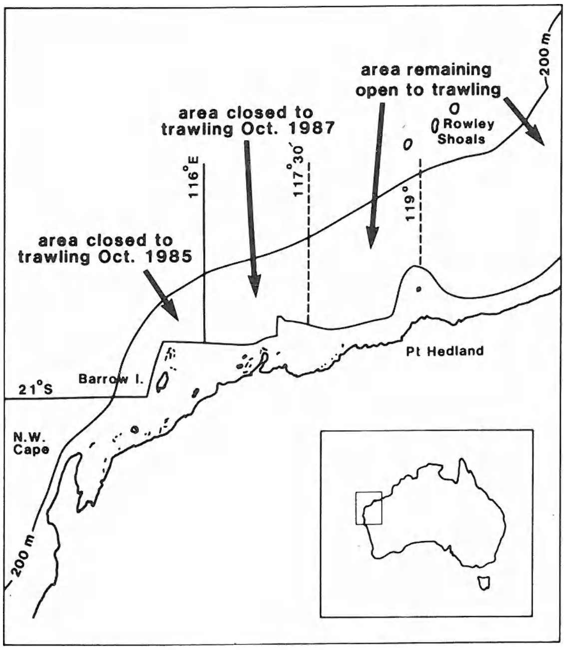

Figure 4. The zones subjected to different fishery management regimes on the Northwest Shelf.

116°45'E–120°E is expected to slow, so that 117°30'E–120°E will probably be trawled only lightly during the next few years.

In practical application, adaptive management was not formally used as the management policy for the resource. However, the adaptive management analysis strongly influenced the management regimes that were adopted. The management regime in place in 1987 was very similar to a regime that performed well in the analysis, although it did not eventuate solely for the reasons used in the analysis. Similarly, the recent development of the domestic trawl fishery was not part of any regime examined in the original analysis, but the benefit demonstrated by that analysis of separating trap and trawl fisheries was used in formulating the management response to development of the domestic trawl fishery.

The principles underlying adaptive management are the credibility placed on different hypotheses of resource dynamics, the economic return from the resource thought to be possible under each hypothesis, and the way that present management actions affect the ability to distinguish between hypotheses with different economic consequences. Practical application of adaptive management involved the application of these principles both to anticipated management options and to management issues and options as they arose.

3. Discussion

The dynamics of the Northwest Shelf fisheries are complex and highly uncertain, and by necessity the assessment of prospective management actions contained many simplifying assumptions. The resource models used were all very simplistic and could be expected to mimic only gross aspects of the dynamics of the real fish community, such as whether a popultion would increase or decrease, quickly or slowly, and with long or short time lags. None of the models used could exhibit behaviour such as multiple stable states, non-stationarity or external forcing, although such behaviour is to be expected in the dynamics of real communities. Furthermore, the complex non-linear interactions between economic investment and resource state were almost totally ignored, and the analysis used a very simple treatment of learning and the use of information by managers. The inclusion of model process error, to allow for stochastic change in the resource models and decision parameters, would be a useful improvement to the analysis. Its inclusion is likely to decrease the estimated ability to distinguish the alternative models.

The experimental designs provided under the various learning period regimes (W) examined were all very simple. More complex designs would be required for resolution of resource models involving spatially and temporally specific influences, but tractable models of these influences could not be constructed with the data available and so were not considered. Also more complex learning period regimes and designs were not considered feasible on the Northwest Shelf.

If the Northwest Shelf community is really driven by highly non-linear interactions, then the implications to parameter estimation, prediction, and the interpretation of any experiment are daunting and well expressed by Neill (1974): "the ecologist is therefore in a difficult position: on the one hand he cannot hope to measure the dynamics of the interactions between pairs of species without perturbing them, and on the other, results of the perturbation depend upon the effects of other species on the competing pair. Properly done the experiment becomes hopelessly difficult."

The approach taken on the Northwest Shelf assumed that it was possible to approximate the important features of the dynamics of the community and its fisheries by simple models. Despite the simplifications the analysis provided a very useful framework for encompassing many processes and uncertainties involved in assessing prospective management actions. It was instrumental in leading to positive management actions despite the initially bewildering level of uncertainty, and proved useful in helping management decisions involving both foreign/domestic and domestic/domestic fishery interactions on the Northwest Shelf.

More generally, there appear to be many underutilized opportunities for fisheries research and management to make active use of the feed-back between management action and empirical learning about resource dynamics. Such opportunities occur almost every time a new management measure is introduced or an old one modified, and yet it is very rare for any consideration to be given to how that measure could be introduced so as to provide the greatest benefit to long-term management (and in particular to learning about the resource's response to the measure). It is quite possible that much research expenditure is currently being devoted to questions that could have been answered more directly and cheaply (or at least posed more sharply) following direct trials using a fishery.

The Northwest Shelf provided a rather unusual situation with many features that favoured an actively adaptive management approach: there were major uncertainties in the dynamics of an extremely complex fish community which made it difficult to select a management regime that could confidently be expected to achieve the management objectives; few scientists were studying the resource so that resolution of the uncertainties by process-oriented research was highly unlikely; the fish had relatively short life spans and hence were expected to respond quickly to changes in the fishing regime; close management control was possible; the existing fishery had relatively low value; the existing fleet had alternative fishing options; and an alternative fishery had relatively high value. The large areas closed to trawling on the Northwest Shelf would not be feasible or justifiable in many fisheries, and even for the Northwest Shelf closures of more than a few years may not be justifiable.

More usually, opportunities for actively adaptive management would be expected to arise from changes in the usual fishery controls such as mesh sizes, catch quotas, and seasonal closures. For example, it may be possible to implement changes in mesh-size regulations in such a way as to maintain revenues and empirically test whether the changes bring about the expected improvements in the fishery. Very little attention has been given to this question, despite the widespread use of mesh regulations, and recent research findings of high mortality among fish that have passed through trawl meshes (e.g. Main, 1988; DeAlteris and Reifstech, 1988) suggest that there would have been considerable value in obtaining empirical support for the use of mesh regulations. Similarly, the difference in yield predictions between single-species models and multispecies models in the North Sea (see Pope, 1991) suggests that it may be possible to devise an experimental management policy that could empirically determine which of these two modelling approaches provides the most reliable management advice. Examination of this possibility appears worthwhile.

While the presented method of analysis can help expose the basic issues and alternative hypotheses for examination and review, this in itself of course does not overcome the uncertainties. Management of marine communities remains fundamentally empirical, and the analysis simply helps guide the choice of which available management actions are worth taking. However, in dealing with difficult community level management problems, where several different views of community dynamics are unresolvable at the time of decision-making, such guidance is perhaps the most constructive outcome possible. The analysis is "scientific", in that the consequences logically follow from stated assumptions, there is emphasis on testing predictions, and it focuses attention on assumptions, their consequences, and their empirical resolution. However, the analysis is not totally objective. Rather it is conditional upon the "state of the world" statements, such as the models to be included and the probabilities initially placed upon them, which cannot be determined with total objectivity. This is a fundamental difficulty with the methodology. Different people with different backgrounds and goals may choose quite different models to include in the set of alternatives, and may choose to assign different prior probability distributions even to the same models. Either of these differences could alter the outcome of the analysis. Furthermore, it is not possible to examine all possible models and prior distributions, and at best the sensitivity to some variation in prior distributions and models could be examined. However, this apparent weakness in the methodology is simply the result of making explicit the process of model selection that occurs in all fishery assessment and management. All assessments involve selection of a general approach, the models to use, and the management implications to examine. But usually it is not obvious that subjective inputs are involved and there is no clear framework for quantitatively specifying or evaluating these inputs. Methodology of the type employed here has the advantage of explicitly examining the value of distinguishing some of these subjective inputs, and determining the ability to do so under various management actions. The widespread use of such methods would perhaps encourage more frequent and rigorous tests of prediction against reality in fisheries management. In this way it may be possible to examine the limits of empiricism and the question of just how predictable and manageable fish communities are.

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Management of multispecies fisheries in New Zealand by individual transferable quotas

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A quota management system (QMS) using individual transferable quotas (ITQs) was introduced into New Zealand's fisheries on 1 October 1986. Most of the 29 species in the system are taken in trawl fisheries which are multi-species in nature. By-catch problems have been experienced because species total allowable catches (TACs) have not been set in proportion to pre-QMS landing levels and because of natural variations in stock size. Management by a QMS system rather than fishing for a total competitive TAC has brought these problems down to the level of the individual fisher rather than the entire fleet. The two major by-catch problems experienced thus far include: (1) TAC over-runs; and (2) TAC under-runs. Current attempts to resolve these problems include: (1) trading of ITQ between individual fishers; (2) fishing on behalf of another ITQ holder; (3) allowance for fishers to overcatch their ITQ by up to 10% in a given fishing year or carry over up to 10% of their ITQ to the next fishing year; (4) surrender of the port price value of over-caught fish to the Crown; and (5) exchange of uncaught ITQ of a certain species for catch of another, over-caught species. These problems and attempts to resolve them are detailed for three different fisheries: the deepwater hoki fishery, the transitional alfonsino/bluenose fishery, and a mixed species inshore fishery. The use of other methods to resolve by-catch problems (basket and sacrificial TACs) is discussed.

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Introduction

A quota management system (QMS) using individual transferable quotas (ITQs) was introduced into New Zealand's fisheries on 1 October 1986. As of 1 October 1989, there were 29 species in the QMS (Appendix A). The fishery for each species is divided into a number of different management units (defined as "fishstocks"), ranging from 2 to 8 for any given species, with a total of 169 fishstocks in the system. Each fishstock is composed of one or more quota management areas (QMAs) (Fig. 1) and may or may not have a biological basis. Details of the QMS are provided in Clark *et al.* (1988).

Most of these species are taken in trawl fisheries which are multispecies in nature. Before the introduction of the QMS, total allowable catches (TACs) were estimated for each of the species to be included in the scheme. By-catch problems have been experienced (mainly in the inshore fisheries) because TACs were not set in proportion to pre-QMS landing levels and because of natural variations in stock size. TACs for the over-exploited inshore species were set at levels from 25% to 75% of the pre-QMS catch levels, depending on the biological status and management objectives for each fishstock. TACs for under- and fully-exploited species taken in the same mixed fisheries were set at levels equal to or greater than the pre-QMS levels. This has resulted in an imbalance in the catch mix relative to the available quota.

Under New Zealand legislation TACs are based on the amount of fish that will produce the maximum sustainable yield (MSY) as qualified by other factors, e.g. economic, environmental, and social. However, only limited data are available for stock assessment for most species in the system. As of May 1988 there were little or no biological data to support stock assessments for 16 of the 29 species (Annala, 1989). For the 1988–1989 fishing year (1 October to 30 September) TACs for 18 species were based mainly on past landings data only. This limited basis for stock assessments has been due mainly to the rapid development of the QMS and has increased the risk of experiencing by-catch problems.