Management of Multi-species Tropical Fisheries

Final Report

MRAG Ltd

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FINAL REPORT

Title of project : Management of multispecies tropical marine fisheries

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Table of Contents

List of Figures

List of Tables

1 FINAL REPORT

1.1 Executive summary

The final report summarises the purpose, activities and results (outputs) of the project. Detailed guidelines for management are given here. General conclusions from practical and theoretical studies are drawn (contribution of outputs). A second part of the report describes the detailed results.

The effects of fishing on multispecies resources are not clearly understood. Models for assessing and managing them are frequently data hungry, and unsuitable for developing country fisheries institutions. This project aimed to assess fishing effects, derive biological management guidelines, and describe minimum data requirements for demersal bank and deep slope reef fisheries, a relatively simple multispecies example, but with widespread applicability. To achieve this, data typically collected by developing country fisheries institutions from artisanal fisheries were analysed for case study fisheries in Tonga (deep slope) and the Indian Ocean (banks). Available local literature described the socio-economic context of each fishery. Theoretical studies were performed using a multispecies model developed for the project. Input parameters for modeling were derived from case studies.

For fisheries similar to those studied, fishing removes top predators and is postulated to lead to prey release with implications for management. No evidence for prey release or other multispecies interactions was found, but species composition changes due to technical interactions were significant. Theoretical studies predicted prey release from 5 years onwards into a new fishery, but its magnitude was less than variation typically observed in available data indicating that prey release would be undetectable. These results suggested that single, and aggregate single species models were adequate to derive management advice. They also indicated that data need not be collected on all species individually, but on the most important species and guilds of others. Improved data collection in relation to technological changes in the fishery was, however, required. Biological data collection on identified key species was also important.

Management simulations have resulted in guidelines for selecting a suitable target fishing mortality for species where length at maturity is unknown and length at first capture cannot be controlled. The guidelines suggest a means of selecting the most important and vulnerable species for analysis, and a method for setting overall effort limits for a multi-species fishery, taking into account different targeting practices and conservation trade-offs. Simulations indicated that closures are primarily useful as a means of reducing effort. Pulse fishing provided no benefit and did not result in long term changes in species composition. Recovery rates for depleted stocks were shown to be lower for deep slope than for banks species. Predator control was shown not to be a viable management option.

Outputs of the project were promoted through seminars, presentations at international meetings, scientific papers and technical reports, and will be presented to collaborating institutions through working groups. They contribute towards ODA's developmental goals through the provision of management guidelines towards the aim of achieving optimum sustainable yields from capture fisheries. Benefits will be long term and relate particularly to the stabilisation of the fishery and the operational and economic benefits this implies.

1.2 Background

Multispecies fisheries and the interactions within them are complex and are not clearly understood. They have been the focus of meetings and workshops (eg. FAO, 1978; Mercer, 1982; Pauly and Murphy, 1982; May, 1984; FAO, 1992; ICES, 1992) and of a considerable literature addressing the effects of fishing on multispecies resources (see Russ, 1991; Jennings and Lock, 1996). Models have been proposed to assess and manage them, yet no clearly defined management tools or strategies are currently available. Larkin (1982), suggested future research be confined to empirical investigations to examine the responses of multispecies tropical fisheries to fishing pressure and management. Gulland and Garcia (1984), in discussing research data requirements, highlighted the fact that in the multispecies situation the amount of information required is enormous and beyond the means of small fishery departments in developing countries.

The effects of fishing on multispecies fish resources may be detected as changes in the relative abundance (species composition) of species in the stock. Such changes are related to fishing intensity, the relative catchabilities of the species, and the level of interaction between them. Competition and predation are considered the most likely forms of interaction in models of multispecies fisheries and may result in species composition changes due to changes in recruitment, natural mortality (e.g. of prey), and growth, or cominations of these (e.g. May *et al*., 1979, in an example of a southern oceans fishery; Polovina 1984, in an example of a coral reef ecosystem).

In addition to biological interactions, technical interactions through mechanisms such as targeting (e.g. by depth) and gear selectivity (ie catchability of one species to different gears, or different species to the same gear) have an important influence on species composition of multispecies resources. Changes in species composition of the catch arising from technological changes must not be confused as reflecting changes in fish abundance in the multispecies community. Those species with higher catchabilities to a particular gear will be subject to greater fishing mortality and will be removed first leading to a greater proportion of other species in the catch (but not necessarily a greater abundance of other species).

For study fisheries, and many others, fishing will tend to remove top predators (Russ 1991). They are targeted as highly valued food fish (although many examples of where prey fish are the target also exist). Furthermore, aspects of their biology tend to make them more vulnerable to fishing: they tend to be aggressive predators with high catchability to gear such as baited hooks and lines; they are slow growing so even if the number of fish remains constant, the biomass will be reduced and the relative proportion of predators to fast growing species in the catch will change; they have relatively lower reproductive capacity and take longer to reach maturity so higher total fishing mortality will occur before spawning resulting in a lower spawning biomass - recruitment may be reduced.

Removal of predators is postulated to lead to increase in prey (e.g. Beddington and Cooke 1982; Beddington 1984; Munro and Smith 1984). Grigg et al. (1984) and Munro and Williams (1985) advocated the fishing of predators as a management strategy in order to enhance catches of prey. Observations supporting these predicted effects, however, are limited (Russ 1991).

Models for the management of fish resources have principally been designed for single species. Polovina (1992) examined the applicability of a number of models to tropical multispecies fisheries. They include those applied to single species in parallel assuming no interaction, those which group all species together and treat them as a single species, and complex ecosystem box models incorporating all interactions. The more complex models require a large number of input parameters with significant requirements for data collection, often beyond the means of small developing country fishery departments. It is therefore necessary to address the question of to what extent it is necessary to use complex models, and whether more simple assessments and management guidelines are appropriate.

Bank reefs and deep reef slopes, the subject of this project, support demersal multispecies fisheries for lutjanids, serranids and lethrinids. They are extensively distributed throughout the tropical and sub-tropical waters of the world and are important in developing countries where resources for stock assessment and management are limited. Demand for better assessment and management of these resources is high. Species of these families share the common characteristics of being long lived slow growing predators. They are generalist feeders but piscivory is more common in serranids and lutjanids than lethrinids and predator-prey interactions may occur within the multispecies complex. The fisheries based on these highly valued food fish tend to be economically important, and fishing may selectively target the most valuable fish with uncertain effects on the community structure. Understanding multispecies responses to fishing is necessary in order for fisheries managers to make informed decisions about appropriate resource assessment and management models, and to develop a suitable management strategy for them. More generally, outputs from this project may be applicable to other types of multispecies fishery as suggested from recent work by Jennings *et al* (1995) and Jennings and Polunin (1996) through FMSP project R5024 which examined shallow water multispecies reef fisheries.

1.3 Project purpose

Using data typically collected by developing country fisheries institutions from artisanal fisheries, the project aimed to undertake an investigation of the responses of relatively simple multispecies fish communities (demersal bank and deep reef slope fisheries) to fishing, and to provide guidelines for the rational management of such fisheries. Specific aims were to examine changes due to fishing pressure in species composition, abundance and associated demographic variables and to compare them with expected changes in these parameters derived from appropriate fishery models; to examine the effects of different management strategies, and using models of the system to highlight possible management measures to enhance yields and ensure sustainability; to identify the minimum data requirements enabling sensible management of multi-species stocks.

The study was confined to examination of biological management controls to be implemented as a component of any wider strategy addressing other issues such as ecological, social, economic, and post harvest factors. Discussion was presented of social and economic factors, but the application of socioeconomic management tools or the development of bio(socio)economic management models was beyond the scope of the study.

1.4 Research activities

In order to describe the observed effects of fishing at the community level, case study examples of demersal bank and deep reef slope fisheries were examined. To examine expected effects and to develop guidelines for management theoretical studies were undertaken.

Case study locations were :

- The Tongan deep reef slope fishery, based principally on volcanic seamounts occurring on the Tonga Ridge in the south Pacific. This fishery is export oriented and 6 target species (*Pristipomoides filamentosus, Pristipomoides flavipinnins, Etelis coruscans, Etelis carbunculus, Epinephelus morhua* and *Epinephelus septemfasciatus*) constitute around 80% of the catch. A shallower banks fishery also operates providing fish for the domestic market. The seamount fishery is believed to be fully exploited. Yield was in the range 65-166% of MSY during the period 1987-91;

- The Indian Ocean Banks Fishery of Seychelles and Mauritius. Species composition varies with latitude in this region (Birkett 1979) and the heavily fished Mauritian fishery on the southerly Saya de Malha, Nazareth and St Brandon banks is virtually mono-specific (*Lethrinus mahsena;* Yield 40-102% of MSY, 1985-94). The Mauritian mothership-dory fishery in the Chagos Archipelago (yield 17-42% of MSY, 1985-94) and the Seychelles fishery, are multispecies in nature : Mahe Plateau (yield 17-56% of MSY, 1985-94) and banks (1-50%), Amirantes (2-23%), Providence/Farquahar (intermittent, up to 109% in 1992) and Astove/Cosmoledo (25% in 1992 only). The Seychelles fishery is the most complex involving a number of boat/gear categories and a large number of species, but like the Tongan and Mauritian fisheries, baited hooks and lines are the principal fishing method. Except for ‘inshore’ areas of the Mahe Plateau close to the centres of population (yield consistently in excess of MSY), the Seychelles banks are lightly exploited.

In order to obtain a sufficient time series, historical and continuing catch/effort and biological data sets were employed. Case study locations were visited annually to implement and monitor data collection programmes, and to obtain this information, and available local literature. In Tonga, data collection had ceased in 1991 due to financial constraints and was resumed with an improved format during November 1993 of this project. Seychelles data collection was well established, but again due to financial constraints, it was necessary towards the end of the project to provide support for the continued collection of biological data from key species. Collaboration with Mauritius was not agreed until the second year of the project and historical data were not available in computerised format. It was necessary to develop a database for catch and effort data prior to 1988 and for all length frequency data, and to provide computing facilities for local data entry. A database for catch and effort data after 1988 existed but had not been employed. Due to internal restructuring at the Albion Fisheries Research Centre, Mauritius, which delayed computerisation, only catch and effort information for the period 1992-1994 was available in a suitable format in sufficient time for inclusion in the analysis. No length frequency data was available.

Analysis of case study fisheries described fish ecology and provided abundance and population demographic parameter estimates for key species to be used as input parameters for the theoretical studies. Information from case studies was used to explore the effects of fishing, and to illustrate different existing management approaches and socio-economic conditions. Single and aggregate single species approaches were employed with case study data. Length based methods were used. Theoretical studies explored fishing effects, management strategies and the importance of input parameters (minimum data requirements). The Multispecies Interactive Dynamic Age Structured simulation model (MIDAS) was developed during the project for this purpose with expertise from within MRAG. The model simulates a number of stocks being fished by a number of gear types and will determine equilibrium and transient dynamics of the stocks. Simulation and sensitivity studies were performed. Simulations assumed deterministic recruitment, and did not incorporate the potential for natural and unpredictable variation.

1.5 Outputs

The outputs of the project were achieved: detailed analyses of case study fisheries were reported; multispecies community responses to fishing were described for study demersal fisheries; key population parameters were derived and management guidelines developed; data requirements were described.

1.5.1 Summary of results

Population demographic parameters for key species from case study locations of Tonga, Seychelles and Mauritius were derived. Depth association, community structure and potential interactions were described. The project confirmed the difficulty of using length based methods of assessment for long lived, slow growing species. At the population level, these methods were inadequate to indicate differences in growth rate related to fishing pressure. Similarly, with the exception of mortality rates, other population demographic parameters could not be correlated with fishing pressure. It was highlighted that uncertainties in estimates of growth parameters, in particular, lead to uncertainties in other parameter estimates and ultimately in assessments upon which management decisions must be made.

Theoretical studies in the literature predict "prey-release" as a significant effect of fishing on multispecies communities. The project indicated that no detectable multispecies responses occurred that could be related to biological interactions and fishing, but that significant detectable responses due to technical interactions did occur. Theoretical studies with MIDAS confirmed that for typical study species, prey release would be undetectable. While prey release was predicted (from around 5 years into a new fishery, onwards), its magnitude was less than variation typically observed in available data.

The results indicated that single species and aggregate single species models were adequate to derive management advice for the case study demersal fisheries without the need for more complex ecosystem models accounting for all multispecies interactions. However, standardisation to account for technical interactions was essential. Simulations were performed to investigate the effect of controls on effort and length at first capture, and a simple rule for determining the ideal fishing mortality for individual species was derived. A set of criteria was formulated for selecting critical species for which such a single species analysis should be performed. A method is presented whereby the most appropriate overall effort level for the multispecies fishery can then be determined from this analysis of some of the component species. These procedures are designed to optimise biological yield, but will be of assistance in making the necessary trade-offs if economic value is to be maximised. Simulations examined the potential benefits of closed areas and seasons, where possible, but generally found such benefits to be indirect.

Management simulations for key (single) species from case study locations were performed, and the fishing mortality producing maximum sustainable yields determined (F_{msy}). Single species recovery rates following fishing were also examined. Deep slope species took longer to recover than banks species, and in each case recovery times were greater than the time required to deplete the resource at varying effort levels.

In relation to minimum data requirements, the finding that single species and aggregate single species models are adequate to describe and manage multispecies demersal bank and deep slope reef fisheries has important consequences. The results indicated that it is sufficient to obtain catch and effort data from the most important species and aggregations of other species without the need for detailed information on every species. However, due to the importance of technical interactions, other details included in the catch and effort data collection need to be improved, particularly those relating to technological changes in fishing methods. Sampling strategies could also be improved to collect better depth and spatial information to compensate for lack of spatial and temporal heterogeneity in both length frequency and catch/effort data which limits the ability to standardise adequately for variation in these factors. However, improvements in sampling design from the fishery may be limited if the patchy nature of the data reflects fishing patterns rather than lack of appropriate sampling. For length frequency and biological data collection, the findings similarly suggested that the number of species from which data is collected need only be confined to the most vulnerable and economically important. For individual species the analysis indicated that length frequency data and age and growth assessment was essential. The more costly biological data to provide parameters such as length at maturity, whilst useful, was not seen as essential for management. An estimate of density dependence in stock recruitment would be very useful for refining management thresholds. A key deficiency in existing data collection related to uncertainty in growth parameter estimates from length based methods.

1.5.2 Guidelines for development of a fisheries management plan

The objectives of this project included the development of biological guidelines for management (Section 6.3.6). These should form only part of any wider management plan which would address a number of objectives of management. Whilst full evaluation of objectives other than those related to biological management were beyond the scope of the present study, guidelines for the development of a management plan are presented which highlight important activities and put into context the biological guidelines of this study. Furthermore, the case study locations illustrated a range of approaches, including, in Seychelles, a change in policies to reflect the economic realities of the fishery. These examples provide lessons of wider applicability for similar fisheries elsewhere.

1. The social, economic and biological objectives of management should be clearly defined and a management plan tailored to meet these objectives. It should be noted that demersal fisheries for lutjanids, lethrinids and serranids may be categorised as low volume, high value fisheries, and that maximisation of economic yield from the fishery is a likely primary objective. Other high volume fisheries (eg. small pelagic species, banks species in Tonga) may better meet social objectives such as provision of food and employment.

2. Social, economic and biological constraints to development and management should be identified, quantified and addressed. Appropriate physical and administrative infrastructures should be developed to address all stages of the harvest-marketing chain. The costs and benefits of different development strategies (eg. for remote locations) should be addressed in defining management objectives.

3. Co-Management should be the basis of the development of the management plan. It should be prepared by a committe including representatives from relevant Government Institutions (fisheries, legal department, licensing authority, planning department etc.) and from the fishing sector (fishermen, processors, traders and exporters).

4. Where they do not exist, fishermens associations or co-operatives should be encouraged in order to ensure common representation on such committees and for the dissemination of management advice. Many of the necessary management controls cannot be applied (without incurring excessive cost) without the full co-operation of the fishing community. It is necessary that there will be an element of self-policing of the fishery by the fishing community, and therefore that the management instruments are understood and accepted.

5. The management plan should be developed on the basis of the ‘best available information’. Data collection for monitoring, assessment and provision of advice should follow the proposed criteria for minimum data requirements (Section 6.5). Fisheries institutions should provide timely advice and there should be an annual review of the

management plan.

6. The costs and benefits of monitoring, control and surveillance of the fishery should be evaluated and reviewed annually.

7. All vessels exploiting demersal resources should be licenced:

   - Licencing criteria should be developed (number and types of vessels to be licensed, locations different classess of vessel may fish, question of allocation of licenses to foreign fishing vessels, experimental fishing, transferability of licenses, etc.)

   - Conditions of licensing should be applied to incude details of management controls (ie. specify gear types, fishing locations, etc.), compliance with legislation (conservation and fisheries, for example), reporting requirements (eg. logbooks, provision of information to fisheries officers), and monitoring (observers, surveillance officers)

   - Administration and guidance documentation should be developed

8. Fish processing and exporting establishments should be licenced and appropriate criteria, conditions, and administration documentation developed. Development of value added products will increase profitability from a limited catch.

9. In relation to defined management objectives, an appropriate combination of the management instruments suggested for biological management should be employed. Certain management instruments confer additional benefits beyond regulation of the demersal fishery e.g.

   - Gear restrictions: by regulating against demersal trawls and bottom set gill nets for example, habitat degradation, ghost fishing, and the catch of protected species (eg turtles) is prevented or limited.

   - Closed areas may protect vulnerable habitats and serve a conservation role.

10. Where possible, an adaptive approach to management should be adopted. For example experimental area closures could be monitored, high effort at particular locations could be permitted and rates of depletion and recovery monitored. This would provide additional information for management allowing over time the dynamic evolution of the management plan.

11. Biological guidelines for management

    a. Biological management advice should be based on the most important and vulnerable species in the catch and guilds of others. Data collection for monitoring, assessment and provision of advice should follow the proposed criteria for minimum data requirements (Section 6.5) which also suggests appropriate means of defining guilds.

    b. Direct catch controls are considered inappropriate, and other means, particularly effort controls should be employed. Catch levels should be monitored and, in certain circumstances, indirect catch controls could be used. A commonly applied rule is that individual species' spawning stock biomass should not be allowed to drop below 20% of unexploited levels. It is recommended that this rule be applied; a new project (R.6437) will examine the appropriateness of this and other similar rules for these fisheries.

    c. Whilst management may adequately be directed at single species or guilds of species since biological interactions are predicted to be negligible and undetectable, the multispecies nature of the fishery must nevertheless be considered in developing management criteria since management advice appropriate for one species may be inadequate for another.

d. Specific guidelines for biological management are presented in Section 6.3.6. Although the guidelines presented are conservative, the method enables informed choices to be made about the risks and benefits of allowing some species to be overfished in order to optimise yields of others. Management instruments are discussed in Section 6.1.2. The following summarises the effects of various management controls and recommends instruments for their implementation. It should be noted that the imposition of management instruments may result in an initial loss of yield to the fishery before benefits occur resulting from the growth of fish and recovery of populations. This will not be a problem where pre-emptive management is applied in lightly fished areas, but for the rehabilitation of heavily fished locations a phased approach may be required to spread the economic burden to the fishermen.

Management control : Catch

Direct catch controls and quotas are not recommended for multispecies fisheries, however monitoring of catches is still essential. In certain circumstances, indirect catch controls may need to be applied.

Management Instruments

- export controls : limit quantity by species; restrict exports of low value whole frozen fish

- improve the economic efficiency of fishing units in order to increase profitability (economic yield) and reduce the need for large catches e.g. promote use of electric reels (assess risk that some fishermen might increase catches to improve profits beyond current levels)

- gear restrictions : prohibit gear that results in high volume low grade product (e.g. trawls)

- development of alternative resources

Management control : Effort

Management control by effort is closely linked with management by length at first capture. Where possible, both forms of management are applied simultaneously. The smaller the length at capture, the less effort is appropriate. At large lengths at first capture, effort controls become less important as the stock is robust to fishing, although the returns from additional effort become small. However, in hook and line fisheries of tropical demersal bank and deep slope reef species, size controls cannot be implemented practically. Management must therefore rest on setting effort levels appropriate to the length at first capture that occurs.

Current rules for setting desirable fishing mortality for individual species rest on the relationship between length at first capture and length at maturity. Simulations for this study have confirmed that these rules are appropriate for these species, taking into account suspected levels of density dependence in recruitment. However, length at maturity data is costly to collect.

By examining the sensitivity of these rules to length at maturity, this study has shown that adequate fishing mortality specification may be achieved for these fisheries without deriving length at maturity estimates. A simple graphical formulation is given relating measured length at first capture to desirable fishing mortality (Fig. 6.9). The sensitivity of this relationship to length at maturity is shown to be small, and the yield that is lost by adopting the most conservative fishing mortality in the sensitivity range is also shown to be insignificant.

The above methods could result in a target fishing mortality for each species in the fishery. However, it is generally not possible or desirable to manage each species individually. Guidelines were formulated for setting an overall effort to be applied across all species (Section 6.3.6). These have to take into account differences in catchability between species, which are largely a result of species targeting. Where one species is targeted, applied effort will have to be limited to the level appropriate for the targeted species, which will lead to underutilisation of other species in the resource. Where species are avoided (e.g. potentially poisonous ciguatoxic fish) these species should not be included when total multispecies abundance is assessed and used to set upper effort limits. While the above guidelines are formulated to take account of differences in targeting, it is appropriate to discourage this where possible, if it is desirable to maximise the biological return from the fishery (eg. for local food requirements) or effort (eg. the number of persons employed in the fishery).

It should be remembered that in new and lightly exploited fisheries, length at first capture may be unsustainably high because of the favourable size structure of unexploited stocks. If the fishery is in such a situation, it is recommended that the fishery be managed as if $Lc=0.5\ L\infty$ until measured Lc has stabilised.

It is recommended that effort controls are the principal means of regulating catch. However, owing to the potential for uncertainty in the parameters on which these targets are based, the inclusion of certain permanently closed areas as part of the overall management strategy should be considered as a means of reducing effort in conjunction with limits on the number of fishing units. Different levels of fishing effort will be appropriate for different locations (banks, seamounts). Greatest effort is likely to be expended in locations close to centres of population. Through controls on the allocation of licences by number and size of vessel, management should aim to control the distribution of effort by location.

Management Instruments

- Licensing by number and size of vessel, and allocation of permitted fishing zones for different classes of vessel.
- Closed areas and seasons
- Price controls to reduce targeting effects (note price controls imply other economic side effects and need to be fully evaluated before implementation)
- Economic incentives and disincentives (to encourage investments in vessels capable of fishing more lightly fished locations)

Management control : Length at first capture (Lc)

In the tropical demersal hook and line bank and deep slope reef fisheries considered in this study, accurate regulation of length at first capture through gear regulation does not seem to be practical. Minimum size controls are not recommended. Owing to this, the management guidelines in this report concentrate more on setting appropriate effort for measured size at capture.

However, simulations were also performed to investigate what size limits should ideally be set in a multi-species fishery of this sort. The larger the length at capture the less the likelihood of overfishing, but this is at some cost to the optimisation of yield. For single species, length at capture should be set equal to the length at maturity, with effort no greater than twice the mortality rate in order to maximise yield. An advantage to this configuration is that the stock is tolerant to increases in effort, so where effort is variable or hard to control, risk is minimised.

In a multi-species fishery length at first capture cannot be regulated by species. Even if a gear catches all species at the same absolute size, this size may be far less than the length at maturity for some species, and greater for others. By setting the minimum size limit to that appropriate for the *smallest* species, and the maximum fishing mortality to that appropriate for the longest lived one (often the *biggest*), yield would be maximised. If fishing mortality exceeds twice the natural mortality of the longest lived species, then the overall length at first capture should be increased to the appropriate level for that species (i.e. its length at maturity), or it may become endangered.

Management Instruments :

- gear restrictions generally, and for hook and line fisheries, since there is some evidence that larger hooks will catch larger fish, regulation against small hook sizes may be beneficial.
- price controls : avoid higher price for smaller (plate size) fish where feasible
- rather than formal closures, fishermen should be encouraged to avoid areas known to be juvenile nursery grounds and to selectively target areas and depths with larger fish

Management control : Closed seasons

Current simulations indicated no benefit from closed seasons. This follows from the nature of the stock recruitment relationship and the fact that spatial variations leading to local differences in abundance, and sex ratio at size differences, such as occur in spawning aggregations, were not taken into account in the model. Intuitively it would seem sensible to protect known spawning aggregations, and this would appear necessary from reports in the literature. Species specific management controls in relation to reproductive strategies, as have been proposed elsewhere, are important in this context (see Section 6.1.3).

Management Instruments

- Area based seasonal closures (co-management)

Management control : Closed areas

Closed areas are expected to enhance the stock recruitment relationship, but to what extent, and whether this would compensate for loss of yield due to closures could not be assessed. However, the allocation of formal closed areas would provide an additional buffer to that of effort controls against overfishing, and would therefore be an additional useful tool. Furthermore, they would maintain a spawning stock of vulnerable species within the multispecies complex, thereby preventing their elimination. In lightly fished locations no loss of current yield would be necessary due to closures. Permanent closures in heavily fished locations would be more difficult to justify due to loss of yield and disruption of economic activity of fishing fleets. The contribution of larval recruitment from unfished areas (eg. the trawlable areas of banks in the Indian Ocean which are not fished) and lightly fished areas (eg. the deep reef slopes of the Tongan banks) needs to be assessed in this context, and may obviate the need for, or reduce the necessary total area of closures. However, these may have a different species composition and so will not benefit all exploited species equally. Further research is required on the optimum number, size (e.g. sufficient to protect 20% of the stock) and spatial relationships of closed areas.

Management Instruments

- permanent closed areas : Marine Protected Areas (co-management)

Management control : Pulse fishing

Pulse fishing is neither recommended nor discouraged. It confers no benefit in terms of additional yield over continuous fishing and implies economic and organisational disruption to the fishery. Equally, however, pulse fishing implies no disadvantage in terms of unwanted changes in species composition and if rotational fishing can be applied without economic disruption to the fleet (eg. in the case of nearby Tongan seamounts) then it need not be discouraged. Indeed, there may be benefits to management from pulse fishing in that short periods of high catch and effort will result in strong temporary depletion, which will enable more accurate production model stock assessments to be made.

Temporary closures equivalent to pulsed fishing may be necessary where overfishing has been identified. Recovery rates from low biomass (SSB approaching 20% SSB_0) are approximately 8% per year for deep slope and 12% per year for banks species. In order to allow a depleted resource to recover to 50% SSBo, the fishery should thus be closed for around 3 years for both species. Such closures, particularly for isolated locations where external recruitment is uncertain, should be applied before SSB falls below 20%

SSB_0. Similarly if pulse fishing occurs, short term heavy fishing is not a signal that serious depletion of the resource is acceptable. Where pulse fishing is permitted the inclusion of certain permanently closed areas can be regarded as a buffer against this.

Management Instruments

- Temporary closures (enforcement may be a problem and this implies the need for co-management)

Management control : Resource manipulation

Predicted effects of fishing indicated that prey release was not significant, and therefore that predator control was not a viable management option. Other means of resource manipulation, such as altering the catch composition by shifting length at capture to favour a particular species, or by targeting (catchability) effects similarly would result in reduced total yields, and may lead to eradication of certain species, reducing also potential future yields. Such management measures may nevertheless be appropriate if, for example, optimisation of economic yield is the objective of management.

Management Instruments

- gear controls
- price controls

1.6 Contribution of outputs

The ODA RNRRS goal to which this project relates has been defined as "to enhance the productivity and productive potential of the land/water interface through improved management of aquatic resources". The project directly addressed the FMSP indicative output of understanding interspecific relationships in tropical, multispecies fisheries, and of developing strategies for the effective management of multispecies artisanal fisheries in order to meet the RNRRS purpose of achieving optimum sustainable yields from capture fisheries. Implementation of management guidelines and other recommendations should contribute towards the long term sustained exploitation of these fisheries, and thus the project contributes to the RNRRS goal. However, the project has also indicated the requirements for additional information in order to derive biological management strategies before this goal can be reliably achieved. The time frame over which benefits of management will be effective depends upon the degree of exploitation of the fishery and the degree to which management instruments are applied. For example, where overfishing has occurred, a fishing ban of around 3 years is required. The potential for increasing catches at case study locations was low (except Seychelles) and benefits relate to stabilisation of the fishery and the operational and economic benefits this implies.

The results are particularly important, and widely applicable, to the situation of developing countries where resources for stock assessment and management are limited. Demersal banks and deep slope fisheries for snappers, groupers and emperors are extensively distributed throughout the tropical and sub-tropical waters of the world. The results indicate that a stock assessment programme targeted at the most important species and a complex of all species is adequate. Complex approaches to the biological management of these fisheries are not required. The observations relating to the effects of fishing, if applicable to other types of multispecies fishery, as would appear to be the case from other studies (Project R5024) suggest that these results may have broader relevance.

Outputs of the project have been promoted through a combination of seminars, presentations at international meetings and workshops, scientific papers and technical reports (details follow). Project results will be disseminated to collaborating institutions directly through short workshop/discussion groups tailored for each situation, and more widely by distribution of reports to regional bodies such as South Pacific Commission; University of the South Pacific (Marine Studies Programme); Pacific Islands Marine Resources Information System; International Centre for Living Aquatic Resource Management; Food and Agriculture Organisation; RECOSCIX. Tonga has not yet implemented a management plan for its fishery and the results of this project should assist Tongan policy makers and fisheries managers in devising an appropriate management plan. The need for such has been clearly recognised within Tonga and there is a high probability of uptake of the findings. The situation is similar in Seychelles. Mauritius has already implemented a management strategy. The development by case study institutions, fine tuning and adoption of management plans which incorporate the biological guidelines suggested in this study is necessary in order for the findings of the work to achieve their developmental benefit. This implies identification and prioritisation of local management objectives, evaluation of social and economic constraints to be addressed in the plan, evaluation of economic yields from the fishery, and ensuring that an appropriate infrastructure is in place from initial harvesting through the marketing chain to the consumer. These are potentially factors which could form the basis of further studies including an adaptive project whereby biological guidelines are practically implemented.

Further biological research suggested by this study arises from the finding that it is adequate to aggregate species into guilds for management. Whilst suitable for total biomass production models, length and age based assessments from guilds imply deriving aggregate demographic parameters, and raises the question of the practicality of this. This is also the case for production models requiring estimates of mortality. The best means of aggregating species into guilds could form the basis of further study. A project (R.6465) has begun to address the question of the reliability of growth estimates and the extent to which this limits our ability to manage demersal fish stocks. Another project (R.6436) will also further develop the guidelines for management of fisheries such as those in Tonga. The genetic basis of fish populations on banks and isolated seamounts, and the sources of recruitment to specific locations are also in need of further understanding.

1.6.1 Seminars/Workshops/Conferences

1993 Management of Tropical multispecies fisheries - project objectives and overview seminar. Imperial College Centre for Environmental Technology.

1993 Demersal Fish Stock Assessment in Seychelles. International workshop on tropical groupers and snappers, 26-29 October 1993, Campeche, Mexico.

1994 The Tongan deep reef-slope fishery. Seminar, CSIRO, Hobart, Tasmania.

1995 Optimisation of yield of *Pristipomoides filamentosus* from the Tongan Seamount Fishery. Joint FFA/SPC workshop on the management of South Pacific Inshore Fisheries, 26 June - 12 July 1995, SPC, Noumea.

1996 Multispecies responses to Fishing at Indian Ocean and Tongan offshore reefs. 8th International Coral Reef Symposium, Panama, 23-29 June 1996

1996 Seminar/working groups to disseminate project results to collaborating institutions are planned for later in 1996 to include Tonga, Seychelles and Mauritius.

1.6.2 Internal Technical Reports

In addition to the regular quarterly annual and field visit reports, the following technical reports were prepared. These contain additional detail to that presented in this report, and may be referred to for more information :

1994. Multi-species Interactive Dynamic Age-structured Simulation (MIDAS) : Model description and simulation documentation. MRAG Ltd, Jan 1994.

1994. The Tongan deep reef-slope fishery. An analysis of data from 1987-1991. Interim report to the Government of Tonga, Ministry of Fisheries. MRAG Ltd

1994. A description of the data entry programme for the Mauritian banks fishery catch and effort data prior to 1988 and for length weight data. MRAG Ltd, June 1994

1996. The Status of Seychelles demersal fishery. Technical report to Seychelles Fishing Authority for the ODA FMSP Management of Multispecies Tropical Fisheries project.

1996. The Mauritius banks fishery - A review and spatial analysis. Technical Report to Albion Fisheries Research Centre, Mauritius, for the ODA FMSP Management of Multispecies Tropical Fisheries project.

1.6.3 Publications

Mees, C.C. and J.R. Rossouw (1995) Optimisation of yield of *Pristipomoides filamentosus* from the Tongan seamount fishery by changing size at first capture. *In* Dalzell, P. and T.J.H. Adams (compilers) South Pacific Commission and Forum Fisheries Agency Workshop on the Management of South Pacific Inshore Fisheries (1995: Noumea). Manuscript collection of country statements and background papers, Volume I, SPC/Insh. Fish. Mgmt/BP7 pp 291-306

Mees, C C (in press). Demersal fish stock assessment in Seychelles - an analysis of a Mothership-Catcher boat fishery. *In* F. Arreguin-Sanchez, J.L. Munro, M.C. Balgos and D. Pauly (eds). Biology and culture of tropical groupers and snappers. ICLARM Conf. Proc. 48.

Mees, C C (in press). Multispecies responses to fishing at Indian Ocean and Tongan offshore reefs. *In* Proceedings of the eighth International Coral Reef Symposium, Panama, 23-29 June 1996.

Mees, C C and J R Rossouw (in Prep.). The potential yield of *Pristipomoides filamentosus* from the Mahe Plateau, Seychelles : management through size and effort controls

2 MATERIALS AND METHODS

2.1 Introduction

The background to this project, and the general approach through a combination of practical and theoretical studies have been described (Chapter 1). Methodologies employed for the derivation of fishery parameters for case studies are indicated in the relevant sections of the text and detailed descriptions are given in Technical reports (MRAG 1994; 1996a;b). The model used for simulation studies is described.

2.2 Available case study data

Where possible, this study aimed to utilise existing databases of information :

- A time-series of information greater than the duration of the project was required to detect any multi-species effects;

- Data typical of that collected by developing country fisheries institutions was required to establish its adequacy for multispecies assessment and management.

2.2.1 Tonga

Fishing activity is directed at shallow banks surrounding the Tongan islands and at deep sea mounts. Initially intended to relieve fishing pressure on the inshore reefs, the Tongan demersal fishery soon became export oriented with a high demand for deep reef slope snappers and groupers. This case study thus relates principally to deep slope demersal resources in contrast to the other case studies relating to banks fisheries. It was considered that due to the relatively small size of sea mounts and availability of data from early in the fishery, the Tongan example would be a good case study to test for multi-species effects of fishing.

Commercial exploitation of the Tongan banks and sea mounts began in 1980. Data collection started in 1986 with the assistance of USAID, and continued until the end of 1991. Limited data collection occurred in the early part of 1992 and resumed in November 1993 with the assistance of ODA through this project. Sampled catch and effort data from the historical data set (1772 records, 1986-1991) was made available to the project. Catch records related to 10 species :

> *Pristipomoides filamentosus, Pristipomoides flavipinnis, Etelis coruscans, Etelis carbunculus, Epinephelus morhua*, and *Epinephelus septemfasciatus*, regarded as deep water and sea-mount associated species, and
>
> *Lethrinus chrysostomus* (correctly *L. miniatus* - throughout this report scientific names and synonyms are those given in FISHBASE, 1995)*, Lethrinus rubrioperculatus, Gymnocranius japonicus* (correctly, *G. euanus*) and a grouper (serranidae, not identified to species level, Sp8), regarded as banks associated shallow water species,

and to a guild of all other species caught. The number of species in the catch was recorded, and the mean number for all trips in any one year was 7-14 species dependant upon depth (the number of species decreases with depth) and targeting effects.

Length frequency data collection (sampled quarterly) was confined to the six deep water species and *L. chrysostomus* (Table 2.1). Length-weight data was collected during 1987 only and was not available to the project. No other biological information was collected.

Catch, effort (1161 records) and length frequency data (Table 2.1) collected between November 1993 and August 1995 was also available to the project. The additional time series (continuing) was insufficient to contribute significantly to the multi-species objectives of this project, but indicated how improved data collection could provide additional detailed information for management, and how the fishery is changing (see Technical Report, MRAG, 1994).

Species	Period	N
P. filamentosus	Nov '86-Dec '91	11,842
P. filamentosus	Nov '93-Jun '95	2,233
P. flavipinnis	Nov '86-Dec '91	7,189
P. flavipinnis	Nov '93-Jun '95	837
E. coruscans	Nov '86-Dec '91	21,943
E. coruscans	Nov '93-Jun '95	5,500
E. carbunculus	Nov '86-Dec '91	8,170
E. carbunculus	Nov '93-Jun '95	924
E. morhua	Nov '86-Dec '91	4,039
E. morhua	Nov '93-Jun '95	254
E. septemfasciatus	Nov '86-Dec '91	3,200
E. septemfasciatus	Nov '93-Jun '95	1,189
L. chrysostomus	Nov '86-Dec '91	11,384
L. chrysostomus	Nov '93-Jun '95	404

Table 2.1 : Length frequency data available from the Tongan case study fishery.

2.2.2 Seychelles

Of the case studies, Seychelles demersal fishery is the most complex involving the greatest number of species, a number of different fishing vessel types and a number of different gear types to exploit the resource. Fishing on the Seychelles banks was limited to locations near to centres of population until 1974 when Schooners were introduced to develop the offshore demersal fishery. Presently, fishing activity occurs both near centres of population and on more distant banks resulting in a gradation of fishing pressure between locations. Some early data collection was undertaken by Government of Seychelles Statistics Office. Seychelles Fishing Authority (SFA) have operated a well established fisheries data collection programme since 1985. Given these details, Seychelles was considered a good location for the current study.

Catch and effort data collection are achieved through Seychelles Catch Assessment Survey (CAS) based on a stratified sampling programme by boat category and landing site. The amount of detail collected (location, depth, climate, current, fishing aids e.g. GPS, fishfinders etc.) differs by boat category. The number of species recorded by boat category also differs (Table 2.2). A large number of species occur. From the three families, lutjanidae, serranidae and lethrinidae, 20 genera and 87 species have been identified in Seychelles waters (Orts, 1993). Those recorded in Seychelles catch statistics are :

Lutjanidae : *Apharaeus rutilans, Aprion virescens, Lutjanus bohar, Lutjanus coccineus* (correctly, *L. sanguineus*), *Lutjanus gibbus, Lutjanus sebae, Pristipomoides filamentosus,* and a guild of all other lutjanids;

Serranidae : *Epinephelus chlorostigma, Epinephelus morhua, Epinephelus multinotatus, Variola loutii*, and a guild of all other serranids;

Lethrinidae : *Gymnocranius robinsonii* (correctly, *G. grandoculis*), *Lethrinus crocineus, Lethrinus elongatus* (correctly, *L. microdon*), *lethrinus mahsena, Lethrinus nebulosus*, and a guild of all other lethrinids.

Additionally statistics are kept on a number of other species relevant to other fisheries, e.g. rabbit fish for the trap fishery, pelagic species for the sport fishery, and so on.

BOAT TYPE	QUALITY OF DATA					
	PERIOD C / F	SP. COMP	LOCATION	DEPTH	OVERALL	
SMALL BOATS	Yes	Crude	Assume all inshore	No : Assume shallow	Fair / poor	from '85
WHALERS	Yes	Crude -90 Good 90+	Some data but poor reporting	No : Assume shallow	Fair	from '85
SCHOONERS	Yes	Good	Gross location reported	Since 1991	Good	from '83
MOTHERSHIP	Yes	Good	Lat / Long	Yes	Excellent	1991-93

Table 2.2 : The quality of data available for the different boat categories recorded in Seychelles Catch Assessment survey.

Length frequency and biological information was recorded for certain key species selected for abundance in the catch and economic importance. Length frequency data was principally collected for *A. virescens, L. sebae, P. filamentosus* and *E. chlorostigma* and biological data for *A. virescens* and *P. filamentosus*. Towards the end of the project data was available for some additional species (Table 2.3).

For length frequency, boxes of incompletely gutted fish were sampled at random. The fish were measured (fork length, to 1 mm) and where possible sexed. Biometric samples employed whole fish encompassing the extremes of the size range observed. They were weighed (to 1 g), measured (fork length to 1 mm) and sexed. Gonads were weighed (to 0.1 g). Maturity stage was assessed for females on a scale of 1 (immature) to 5 (spent) according to criteria established by micro and macroscopic examination. A Gonadosomatic Index (GSI) - the gonad weight in grammes / whole fish weight in kilograms - was calculated.

Species	Biological data		Length Frequency	
	Period	N	Period	N
Pristipomoides filamentosus	Dec '89-Dec '91	1769	Nov '89-Dec '94	38002
Aprion virescens	Apr '91-Apr '94	915	Jan ' 91-Dec '94	23083
Lutjanus sebae			Nov '89-Dec '94	28220
Epinephelus chlorostigma			May '90 - Dec '94	25112
Apharaeus rutilans	Sept - Dec '94	111	Sept-Dec '94	957
Lutjanus bohar	Sept - Dec '94	88		
Lethrinus mahsena	Oct-Dec '94	107		
Lethrinus variegatus	Sept-Dec '94	150	Sept-Dec '94	1114
Lethrinus nebulosus	Apr '92	50	Mar-Dec '92	860

Table 2.3 : Length frequency and biological data available from the Seychelles case study fishery.

2.2.3 Mauritius

The Mauritian banks fishery occurs on banks within its EEZ on the Mascarene Ridge in the Indian Ocean, and traditional fishing rights are also exercised on the Saya de Malha bank and within the EEZ of the British Indian Ocean Territory (BIOT, Chagos Archipelago). This case study is an example of a heavily exploited fishery with a data set extending back to 1977. However, with the exception of the Chagos Archipelago the fishery is essentially mono-specific.

The collaboration with Mauritius (Albion Fisheries Research Centre, AFRC) was not agreed until the second year of the project and only paper records of data were available. Catch and effort data for the period 1992-1994 was computerised in sufficient time to be included in the analysis. Subsequently, data extending back to 1988 was computerised, but not analysed. Summary catch and effort data only was collected between 1977 and 1987. In 1988 improved procedures were introduced. Details relating to species composition, however, are poor, and relate to guilds of ‘Red’ fish and ‘White’ fish. The principal white fish, *Lethrinus mahsena* makes up around 80% of the catch. In addition to the Mauritian data, BIOT inshore logbooks and observer data were available for fishing in the Chagos Archipelago between 1991 and 1995. These record greater species detail. Commonly recorded species from Chagos are :

> *Lethrinus mahsena, Lethrinus nebulosus, Lethrinus rubriopeculatus, Lethrinus variegatus, Aprion virescens, Lutjanus bohar, Lutjanus gibbus, Pristipomoides filamentosus, Epinephelus morhua, Plectropomus leopardus, Plectropomus maculatus, Variola albimarginata* and *Variola loutii.*

Length frequency data have been collected by AFRC for *L. mahsena*, but was not available in a suitable format for analysis by the time of writing. No biological data is available.

2.3 Modelling and simulation

This section describes the population dynamics model which was used for this study, and briefly describes the computer simulation which was developed to implement the model. This formed the major tool for the theoretical studies described in the following chapters.

2.3.1 Basic dynamics

For each stock, the number of fish of age a at time t is given by

$$N_{a,t} = \begin{cases} N_{a-\delta,\,t-\delta}\, e^{-JuvM(B(t-\delta)).\delta} & \text{for } a-\delta < A_r \\ \left(N_{a-\delta,\,t-\delta}\, e^{\frac{-M(a-\delta,\,B(t-\delta)).\delta}{2}} - C_{a,t}\right) e^{\frac{-M(a-\delta,\,B(t-\delta)).\delta}{2}} & \text{for } a-\delta \geq A_r \end{cases} \qquad (1)$$

where

$B(t)$ = vector of current minus unexploited equilibrium adult biomasses for all stocks

$M(a,B)$ = adult natural mortality (function of age , biomasses)

$JuvM(B)$ = juvenile natural mortality (function of biomasses)

δ = simulation delt

$C_{a,t}$ = catch of fish aged $a-\delta$ during the delt ending at time t

A_r = age at recruitment

For each stock and gear combination, the catch of fish aged a - δ to a, taken during the period t - δ to t, is

$$C_{a,t} = \begin{cases} 0 & \text{for } a-\delta < A_c \\ \dfrac{F(a-\delta,\,t-\delta)}{F(a-\delta,\,t-\delta)+M(a-\delta,\,B(t-\delta))} \,.\, N_{a-\delta,\,t-\delta}\left(1 - e^{-[F(a-\delta,\,t-\delta)+M(a-\delta,\,B(t-\delta))].\delta}\right) & \text{for } a-\delta \geq A_c \end{cases} \qquad (2)$$

where

A_c = age at first capture

$F(a,t)$ = fishing mortality

= $E(t).S(a).C(t)$ where

$E(t)$ = effort

$S(a)$ = selectivity

$C(t)$ = catchability

The catch for a gear is zero if the time falls within the closed season for that gear.

The weight of an individual at age a and time t is given by

$$W_{a,t} = A\ [L_{a-\delta} + (L_\infty - L_{a-\delta}).(1 - e^{-K(t-\delta,\, B(t-\delta)).\delta})]^{\alpha} \tag{3}$$

where

$L_a = [\frac{W_a}{A}]^{\frac{1}{\alpha}}$

$=$ length at age a

$A, \alpha =$ parameters relating length to weight

$L_\infty =$ asymptotic length

$K(t) =$ time (season) dependent growth rate

2.3.2 Biomass Indicators

Given that

$A =$ age group (integer)

$A_{\max} =$ maximum age group

$A_{Cmin} =$ minimum age at first capture

$O_M =$ maturity ogive (selection function)

$S_S, S_E =$ start, end of spawning season

The spawning stock biomass averaged over a year y is defined as

$$\mathrm{SSB}_y = \frac{1}{(S_E - S_S)} . \sum_{A=0}^{A_{\max}} \int_{t=S_S}^{S_E} N_{A+t,\, y+t} . W_{A+t,\, y+t} . O_{M_{A+t}}\, \mathrm{d}t \tag{4}$$

The yield over a year y is

$$\mathrm{Yield}_y = \sum_{A=0}^{A_{\max}} \int_{t=0}^{1} \sum_{\mathrm{Gears}} C_{A+t,\, y+t}\, \mathrm{d}t \tag{5}$$

The average exploitable biomass in year y is defined as

$$\mathrm{EXB}_y = \int_{t=\mathrm{frac}(A_{Cmin})}^{1} N_{A+t,\,y+t}\cdot W_{A+t,\,y+t}\,\mathrm{d}t + \sum_{A=\mathrm{ceil}(A_{Cmin})}^{A_{\max}} \int_{t=0}^{1} N_{A+t,\,y+t}\cdot W_{A+t,\,y+t}\,\mathrm{d}t \qquad (6)$$

The total exploitable biomass in year y is calculated as in equation (6) only using the age at recruitment, A_R, instead of the minimum age at first capture A_{Cmin}.

2.3.3 Stock Recruitment Relationship (SRR)

A number of functional forms for stock recruitment relationships have been defined in the literature. For the purpose of this study, the following four forms were considered. Each relationship is a function of the spawning stock biomass, and returns the number of fish born in the fishery in the next year.

I Constant recruitment

$$R = R_{const} \qquad (7)$$

ii Beverton-Holt

$$R = \frac{\mathrm{SSB}}{1-\theta\left(1-\frac{\mathrm{SSB}}{\gamma}\right)} \qquad (8)$$

iii Ricker

$$R = a.SSB.e^{-b.SSB} \qquad (9)$$

iv Shepherd

$$R = \frac{p.\mathrm{SSB}}{1+\left(\frac{q}{\mathrm{SSB}}\right)^{\beta}} \qquad (10)$$

The parameters p,q and β, a and b, or θ and γ are specific to the fish stock. The Shepherd SRR may be specified directly by providing the parameters p,q and β, or indirectly by specifying the unexploited equilibrium point and the shape of the SRR relative to that point. This means that the shape of the SRR is fixed, but not the absolute position or size. Given an equilibrium point at (R_0,S_0) (note that once R_0 is specified, S_0 can be calculated), the alternatives are:

a) Where β=1 (Beverton-Holt shape), specify the unexploited equilibrium recruitment (the corresponding spawning biomass is calculated by the program) and a parameter d (0 < d < 1) which indicates the degree of curvature (density dependence) of the SRR between the origin and the equilibrium point. The conversion equations are

$$\begin{aligned} \beta &= 1 \\ p &= \left(\frac{1+d}{1-d}\right)^2 . \frac{R_0}{S_0} \\ q &= \frac{R_0 . S_0}{pS_0 - R_0} \end{aligned} \qquad (11)$$

b) For a Ricker function shape, specify the unexploited equilibrium recruitment R_0, the ratio of the maximum to the unexploited recruitments R_m/R_0, and the ratio between the spawning biomass at which maximum recruitment occurs and the equilibrium spawning biomass Sm/S_0. Then,

$$\begin{aligned} 0 &= \beta\left(\frac{R_m}{R_0} - \frac{S_m}{S_0}\right) + \frac{S_m}{S_0}\left(1 - \frac{1}{(S_m/S_0)^\beta}\right) \qquad \text{(Solve for } \beta\text{)} \\ K &= \frac{(S_m/S_0).S_0}{\left(\frac{1}{\beta-1}\right)^{1/\beta}} \\ a &= \frac{(R_m/R_0).R_0}{K} \; \frac{\left(1 + \frac{1}{\beta-1}\right)}{\left(\frac{1}{\beta-1}\right)^{1/\beta}} \end{aligned} \qquad (12)$$

c) Specify β, the unexploited equilibrium recruitment and the slope G of the function at the equilibrium point given as a fraction of the slope R_0/S_0. The conversion is given by

$$\begin{aligned} q &= S_0 \sqrt[\beta]{\frac{-R_0\beta - (GS_0 - R_0)}{GS_0 - R_0}} \\ p &= \frac{R_0}{S_0}\left[1 + (\frac{S_0}{q})^\beta\right] \end{aligned} \qquad (13)$$

It should be emphasised that, in the form of the model used for this study, the age at recruitment has been set to zero, so the output from the stock recruitment relationship is a number of 'births'. Since larval stages and juvenile periods of high mortality are not being modelled, this is equivalent to the number of births that would be necessary to reach the correct population size at true recruitment, given constant mortality across all ages.

2.3.4 Interactions between stocks

The initial mortality of each species in unexploited equilibrium is assumed to contain a component which is due to predation by other species. This means that if the predator biomass declines from equilibrium levels, the prey mortality will increase. This increase is modelled as proportional to the percentage reduction in predator biomass. Density dependence in growth is modelled in an analogous manner.

Each species is modelled in the same way, so that the effects of fishing on both predator and prey can be evaluated. However, for this project, prey release was deemed to be the most significant multi-species effect.

Let the vector **B**(t) = (**B$_C$**(t) - **B$_0$**)/**B$_0$** denote the proportional difference between the current adult biomass and unexploited equilibrium biomassess for each stock. **B** has dimension N_s by 1, where N_s is the number of stocks. If, when all stocks are unexploited, the age dependent adult mortality for each stock is given by **M$_0$**(a), the juvenile mortality by **JuvM$_0$** and the seasonally variable growth rate by **K$_0$**(frac(t)), then the influence of the various stock biomasses can be modelled as

$$\begin{aligned} M(a,t) &= M_0(a) + P_M.B(t) \\ JuvM(t) &= JuvM_0 + P_J.B(t) \\ K(t) &= K_0(\mathrm{frac}(t)) - P_K.B(t) \end{aligned} \qquad (14)$$

where **P$_M$**, **P$_J$** and **P$_K$** are N_s by N_s arrays of constants. These constants are inputs to the model and will usually be positive. The biomass dependent mortalities and growth rate are subsequently limited so that they cannot vary by more than a given percentage around their biomass independent values.

2.3.5 MIDAS

The model described above was implemented, for this project, as a computer simulation which is known as the Multi-species Interactive Dynamic Age-structured Simulation (MIDAS). It is fully documented in Rossouw (1994).

To summarise, MIDAS simulates the dynamics of a number of stocks being fished by a number of gears. Basic biological parameters are required for each stock and the option exists to specify some as varying, e.g. mortality can vary with age and growth rate with season. Biological interaction between the stocks can be represented, whether based on competition or predation. An estimate of the relative recruitment or biomass levels of the stocks at unexploited equilibrium is required for initialisation. The model includes various stock recruitment relationships, with a maturity ogive and spawning taking place during a specified period. Instantaneous recruitment takes place once a year.

Interaction between each gear and each stock is specified in terms of selectivity functions and seasonal catchability. Management controls are exercised by means of overall effort levels, effort distributions amongst gears and open seasons for each gear.

MIDAS can be used to investigate both equilibrium states and transient dynamics of a biological system. It will:

- determine unexploited equilibrium recruitment for all the stocks such that the required adult biomass ratio between stocks is attained.

- find the equilibrium age-structure and other states of all the stocks for various overall effort levels.

- simulate the transient behaviour of the stock given:
  - changes in overall fishing effort with time;
  - lognormal variation in recruitment or catches.

The age structure is determined to a resolution of a year, but the simulation delt is a variable parameter. A delt of 0.02 (~ a week) has proved to be a good compromise between accuracy and speed.

3 CASE STUDY SITES AND FISHERIES

3.1 Characteristics of bank and deep slope fisheries

A number of different reef types have been defined of which bank reefs occur, like patch reefs, on irregularities on shallow parts of the sea bed. Unlike patch reefs, however, bank reefs tend to be deeper. They may occur both on the continental shelf and in oceanic waters (UNEP/IUCN, 1988). The banks are usually volcanic structures of various sizes, although in this respect, one case study differs - the Seychelles islands and Bank (Mahe Plateau). With St. Helena in the Atlantic, they form the only granitic islands in the world. Coraline structures tend to be associated with shallower parts of the banks -reef building corals are restricted to a maximum depth of 30 m. Deeper central parts of the banks may be composed of rock or coral rubble, sand or shell deposits. Outer bank slopes generally shelve steeply. Banks thus support a variety of habitats which in turn support a variety of fish species.

Fish distribution on banks is affected by substrate. That suitable for study species of lutjanids, serranids and lethrinids is patchy leading to isolated groups of fish with little lateral exchange or adult migration except where the patches are close together. These types of fishery may be regarded as consisting of meta-populations associated with specific features or habitats, interconnected through larval dispersal. Whilst genetically, fish in different patches may be the same population, practically at the adult stage they are discrete populations. Knowledge of fish behaviour, and genetic studies support this scenario :

- Serranids in particular, and some lutjanids are territorial whilst lethrinids tend not to be, e.g. *Epinephelus coicides, Epinephelus malabaricus* and *Lutjanus russeli* in NE Australia moved little and were recaptured within 40 m of release sites after 2.5 years (Sheaves, 1993); see also 4.2. However, serranids are known to form spawning aggregations, and the extent to which this may result in mixing of adult populations has not been established.

- Genetic studies in NW Australia (*L. sebae, L. nebulosus, Epinephelus multinotatus, Lethrinus choerorynchus*, Johnson *et al*., 1993) and the northern Gulf of Mexico (*Lutjanus campechenus*, Camper *et al*.,1993) are consistent with the hypothesis that single populations, or extensive connections of populations occur over large distances.

In contrast to this scenario, however, tagging indicated separate self sustaining stocks of snappers in different parts of Shark Bay in Australia (Moran, 1987). In the case of banks fisheries it is not known to what extent communication between banks may occur, or whether individual banks contain single or several self sustaining populations.

Seamounts are undersea mountains, mostly of volcanic origin, which rise steeply from the sea bottom to below sea level. They have been clearly defined (seamounts have an elevation of more than 1000m with limited extent across the summit), but in the literature on the biology, geology and oceanography of such features, frequently size has not been considered and the definitions have not been adhered to (for a review of the biology of seamounts, see Rogers, 1994). Seamounts close to the surface may be associated with coral growth. In the extreme, atolls, roughly circular reefs around a central lagoon probably correspond to the fringing reefs around long submerged islands. In this report it is deep-reef slope fisheries that are the focus, rather than seamounts in particular, although seamounts are a feature of the Tongan case study. Consequently discussion relates to fish taken at a particular depth band, which may relate to deep slopes of banks or to seamounts, but consideration is given to the characteristics of sea mounts relevant to the Tongan situation. Moffit (1993) provides a review of deep water demersal fish, and Dalzell and Preston (1992) summarise exploratory fishing information for deep slope resources in the Pacific.

Species composition is closely related to depth (see Section 4.2), and deep slope fisheries are broadly considered to be in the region 100-500 m. A rapid decrease in the number of species occurs from 200-400m depth, and most fish observed are associated with hard substrates, holes, ledges or caves (Chave and Mundy, 1994). Lethrinids are not generally caught in deep water. Territoriality is considered to be less important for deep water species of serranids, and snappers tend to form loose aggregations. Mixing of the adult population of deep water snappers and groupers may thus be greater than for shallow water species occurring on banks. However, in the context of seamounts, it is considered unlikely that adult migration will occur between isolated seamounts. Genetic studies on target fish species would be useful to establish if individual seamounts contain self sustaining populations of fish or whether it is likely that recruitment may occur from adjacent banks and seamounts.

Seamounts have complex effects on ocean circulation. One effect, known as the Taylor column, relates to eddies trapped over seamounts to form closed circulations. Such effects could result in larval retention around the seamount supporting self sustaining populations of fish. Although evidence for retention of larvae of seamount species over seamounts is sparse (Boehlert and Mundy, 1993), endemism has been reported for a number of fish and invertebrate species at seamounts (see Rogers, 1994). However, Wilson and Kaufman (1987) concluded that seamount species were dominated by those on nearby shelf areas, and that seamounts act as stepping stones for transoceanic dispersal of species. Snappers and groupers both produce pelagic eggs and larvae which tend to be most abundant over shelf waters. Larvae of eteline snappers, by contrast, are generally found in oceanic waters. It would seem likely that populations of snappers and groupers on seamounts rely to some extent on inputs of larvae from external sources. However, such recruitment may be highly variable and unpredictable, and thus recovery from overfishing would be uncertain.

The patchy nature of fish distribution, both on banks and between scattered sea mounts, has a number of implications :

- Fishing may occur sequentially between patches of habitat or seamounts. The signs that a resource is becoming depleted will be difficult to detect (catch rates will not decline, population demography, e.g. changes in length at capture may not be observed);

- For resource assessments based on yield per unit area accurate estimates of the areas of different habitat types will be required; the usefulness of production models may be limited due to sequential fishing;

- Changes in population parameters due to fishing will not be detected unless precise details of fishing location are known.

For these types of fishery it is clear that accurate assessments of the area of habitat are charted, and that in collection of fishery statistics, detailed information is obtained on fishing location. Information on habitat mapping is sparse and particularly in the context of developing countries, expensive to collect. Remote sensing techniques may be useful for shallow water habitats (to around 30 m) but their applicability to deeper banks and seamounts is limited. Seamounts are poorly charted, and even where their location is known, bathymetric assessments are frequently lacking.

Case study fisheries on banks and seamounts share the common characteristic that the principal fishing method is by hook and line which target large predators (lutjanids, lethrinids and serranids). Whilst this represents a multispecies fishery, information on non target species on banks and deep reef slopes is not available. The effects of fishing only part of the demersal resource on the ‘reef’ community as a whole cannot therefore be evaluated.

3.2 A description of the case study sites

3.2.1 Geography and physical characteristics

■ Tonga

The Kingdom of Tonga consists of a widely scattered group of 171 islands (land area, 699 km^2) lying between 15^{o}00'-23^{o}30' S and 173^{o}00'-179^{o}00' W in the Pacific Ocean. The 200nm exclusive economic zone (EEZ) encompasses 700,000 km^2 of ocean. There are three major island groups : the low coral islands of the Tongatapu and Eua Group in the south, and of most of the central Ha'apai Group, and the raised coral islands comprising most of the Vava'u Group in the North. Some islands are volcanic. The capital, Nuku'alofa, is situated on Tongatapu.

The climate is mild : average temperatures range from 23^{o}C in the south to 26^{o}C in the north. The south-east trade winds prevail but during the warm period (October - March) tropical cyclones may occur. Average rainfall in Nuku'alofa is 1733 mm (UNEP/IUCN, 1988).

A chain of volcanic seamounts occur on the Tonga Ridge running parallel to the Tonga Trench in a NNE-SSW line. The extent of the 200 m isobath (as an index of the fishable habitat for deep slope species) was estimated to be 960 nm for all banks and seamounts (Langi and Langi, 1987), and 294 nm for sea mounts alone (Langi *et al*. 1988). These are considered to be underestimates since many areas are poorly charted and new mounts are still being discovered. The current knowledge on the position of seamounts is indicated in Fig. 3.1. Certain sea mounts have been charted in detail (Tawara et al. 1987; 1989). Water temperatures recorded during December by Tawara et al. (1987; 1989) were similar for any given depth for the seamounts examined : sea surface, 26^{o}C; 50 m, 23^{o}C; 100 m, 22^{o}C; 150 m, 20^{o}C; 200 m, 19^{o}C; 250 m, 18^{o}C; 300 m, 16-17^{o}C; 400 m 12-14^{o}C; 500 m, 10^{o}C; 1000 m, 5^{o}C.

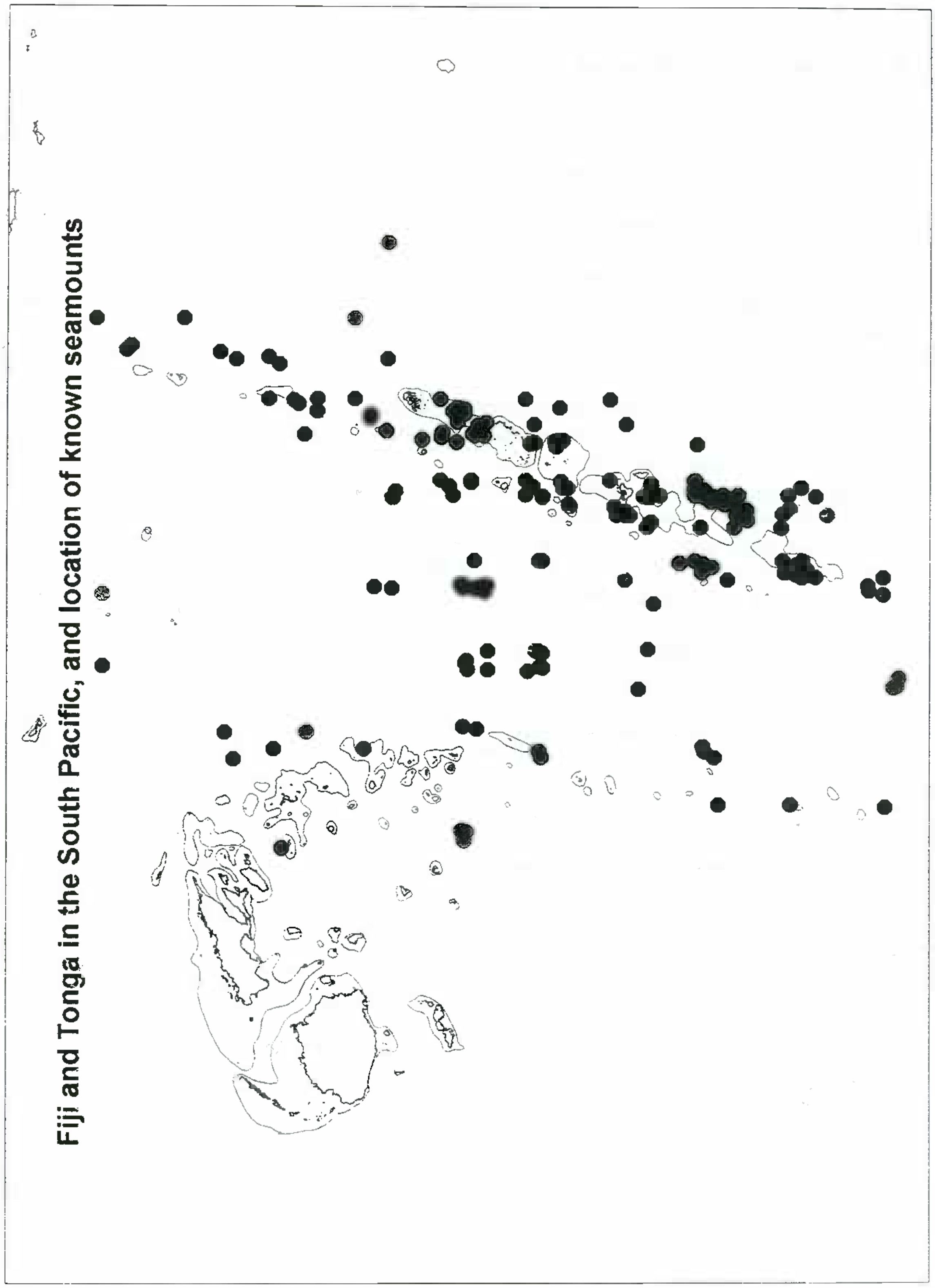


Figure 3.1 : The Tongan islands, and locations of all known seamounts (From various sources : Ministry of Fisheries, Royal Tongan Navy, TPM)

■ Seychelles and Mauritius

Seychelles and Mauritius occur as islands on the Mascarene Plateau, which extends from 4°S to 21°S in the western Indian Ocean. The extensive limestone Mascarene Plateau is of diverse origin, including the ancient granitic islands of Seychelles, younger volcanic islands such as Mauritius, sand cays and atolls, emerged and submerged coral reefs, an algal ridge, and sea mounts. The Plateau forms a shelf at a depth of 200 m on which there are several shallow banks (no more than a maximum of 100m deep, and on average substantially less) forming the basis of the banks fisheries in this region. In addition to the banks of the Mascarene Plateau Mauritius exercises traditional fishing rights in the predominantly coraline British Indian Ocean Territory (Chagos Archipelago), and the Republic of Seychelles includes coral islands and Plateaux to the west of the Mahe Plateau (Table 3.1; Fig. 3.2).

Pronounced bathymetric differences exist between the coraline banks of Seychelles and Chagos and the banks of the Mascarene Plateau. The southern islands lie in the west flowing South Equatorial Current whilst the northern islands are influenced by the east flowing Equatorial Counter Current. The northerly islands including all those in Seychelles and the Chagos are largely unaffected by cylones, whilst the Mauritian banks are subject to cyclonic depressions and severe sea states. These occur from mid November to mid March. South East Trade Winds affect the region from the end of May to October, and calmer periods of light variable winds occur in between. A detailed description of the climate, oceanography and productivity of the region occurs in project Technical Reports (MRAG, 1996a,b). Seasonal variation in climatic conditions affects fishing activities throughout the region.

Code	Bank / Plateau	Country	Bank Area (to 75m, km^2)	Yield (tonnes)
1.	Cosmoledo / Astove	Seychelles	398.1	40.1
2.	Providence / Farquahar	Seychelles	1621.1	163.4
3.	Amirantes Plateau	Seychelles	3999.0	403.1
4.	Mahe Plateau	Seychelles	41338.0	4166.9
5.	Banks South of the Mahe Plateau	Seychelles	2198.6	221.6
6.	Saya de Malha Bank	Mauritius / International waters	42116.0	2500.0
7.	Nazareth Bank	Mauritius	22814.0	2000.0
8.	St Brandon	including Albatross : Mauritius	9212.0	1200.0
9.	Chagos Archipelago	BIOT	8514.0	750.0

Table 3.1 : The banks and plateaux fished by Mauritius and Seychelles, and the codes assigned to them (1-9) in this and Technical reports, the estimated area of each bank and sustainable yield of handline caught demersal fish.

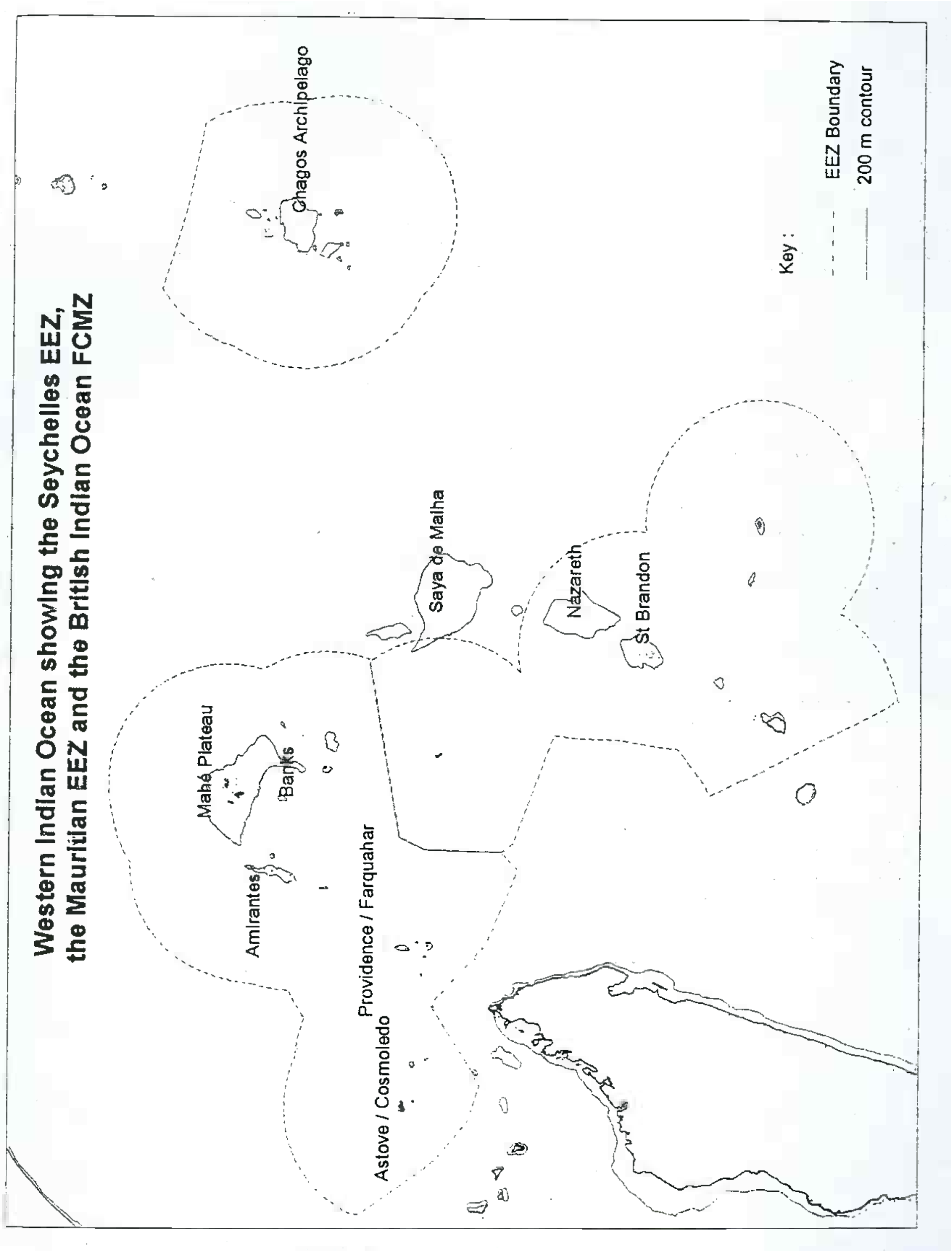


Figure 3.2 : The western Indian Ocean showing the location of banks exploited by fishing vessels from Seychelles and Mauritius.

3.2.2 Status of the fisheries, vessels and fishing methods

■ Tonga

Tonga was a net importer of fish at the end of the 1970's (UNEP/IUCN, 1988). Resources of coastal reefs and lagoons were considered to be under pressure. Following exploratory fishing by the South Pacific Commission Fisheries Programme (Mead, 1979; 1987; Dalzell and Preston, 1992) Government of Tonga began to promote exploitation of deep slope resources of snapper and grouper. Commercial exploitation began in 1980.

To provide suitable fishing vessels at reasonable terms to fishermen a boat building programme was established under the United Nations Capital Development Fund. FAO / UNDP provided technical assistance. This programme aimed to provide a fleet of 40 vessels designed for bottom fishing using FAO Samoan hand reels containing monofilament line terminating in wire trace with three baited tuna circle hooks (for a description of the fishing method see Mead, 1979). Earlier vessels were 20 feet in length, equipped with 2 reels and fished banks close to the main islands. Subsequently 28' and 32' wide and narrow bodied vessels fitted with 4 reels were constructed which fished further offshore. The 40 vessels were completed by 1988 and the addition of privately built boats brought the total fleet size to 44 that year. For a number of reasons the fleet had declined to 19 vessels by 1992. Certain export companies began to introduce larger vessels of different designs in 1992 and 1993 against which the 28' vessels compete poorly due to limited range and ice-hold capacity. This trend looks set to continue. Thus the structure and fishing power of the fleet will change significantly. In 1994 23 boats were actively fishing.

Fishing activity is directed at shallow banks surrounding the islands and deep sea mounts. Initially intended to relieve fishing pressure on inshore reefs the fishery soon became export oriented with a demand for deep slope snappers and groupers. Since 1980 the nature of the fishery has thus changed, shifting more towards deep water and the sea-mount fishery, and away from the shallow banks fishery. The two may be considered separate fisheries although some overlap of species occurs. Geographical distinctions may also be made in the fishery : that based at Tongatapu is highly export oriented, whilst boats on Vava'u target fish mainly for the local market. Ha'apai boats may be considered intermediate, often landing catches in Tongatapu. This distinction along commercial lines has become even more pronounced since the introduction of new vessels by companies in Tongatapu in the early 1990's. These vessels are well managed and operate as a semi-industrial fishery targeting deep water species for export in contrast to the artisanally operated FAO design vessels in the north which increasingly target banks species.

The greatest catch (563 tonnes) and effort (123,547 reel hours) occurred in 1987, and for years when data are available declined to 1990 (catch, 238 t; effort, 59,340 rh) increasing in 1991, and 1994 (catch, 358 t; effort, 74,210 rh). Most catch and effort related to seamount (and some bank) locations in the south (83% in 1987, 88% in 1990 and 80% in 1994). Catch rates were similar in the north and south in any one year during the period 1987-1991, but varied between years (3.6-5.3 $kg.rh^{-1}$). In 1994 (and for 2 months of 1993 and 8 months of 1995 for which data were available) catch rates in the north were less than those in the south (N, 4.18 $kg.rh^{-1}$; S, 5.95 $kg.rh^{-1}$ in 1994). This difference was even more pronounced considering catch per trip (N, 232 $kg.trip^{-1}$; S, 702 $kg.trip^{-1}$) and is believed to relate to the differences in the nature of the fleet, already described, although vessels from the south do fish in the north. Six target species dominate the catch (see Section 2.2.1). Species composition is a function of depth (see Section 4.2) and changes have been observed, principally from a predominance of *Pristipomoides* spp to *Etelis* spp., as increasingly deeper resources were exploited.

■ Seychelles

Immediately following colonisation of the Seychelles in the eighteenth Century, fishing for demersal species was confined to coastal areas near centres of population, and today is concentrated around the granitic islands. The coralline islands are sparsely inhabited and local exploitation will have had minimal impact. Exploitation occurs from a number of vessel categories including small boats with limited range (‘*Pirogues*’ and ‘*Outboard*’ powered craft). In 1974 ‘*Schooners*’ were introduced to allow fishing on the offshore banks and periphery of the Mahe and Amirantes Plateaux. Foreign ‘*Mothership-dory’* operations were licensed to fish in the Amirantes, Cosmoledo, Providence and Farquhar between 1974 - 1977 (Ratcliffe, 1974; Harris, 1977). ‘*Whalers*’ have increasingly targeted demersal species since the mid 1980's and developments in the Schooner fishery since that time have resulted in more efficient vessels with a greater range and better means of exploiting the demersal resource. Between 1991 and 1993 a Seychelles Mothership-dory venture operated. Thus fishing pressure has increased only relatively recently and for the outlying islands exploitation has been sporadic. Locations on the Mahe Plateau close to centres of population are considered to be heavily fished, whilst outlying banks will support increased exploitation. The analyses support this (See Technical Report, MRAG, 1996a, also for a discussion of changes in the fleet structure). However, there have been social and economic constraints to development of the more distant water fishery (Mees, 1990b).

The predominant fishing method employed for demersal species by all boat categories is hook and line. Pirogues and outboards may also deploy fish traps but the proportion of demersal species is negligible. Whalers may target either carangidae or demersal species using hook and line. Increasingly demersal species are being targeted although the carangid fishery is the traditional target of this category.

The schooners and mother-vessel target demersal species. Hooks and lines have traditionally been used, but electric fishing reels are employed by a number of schooners and the Mothership-dory operation. Catch rates are increased by up to 30 % (Bautil, 1988; Mees, 1990a). Drop-line fishing was tested by SFA and has recently (1994) been adopted commercially. Deep bottom set gill net fishing for demersal species was developed by SFA and was employed commercially by one of the larger schooners for a short period in 1993 but has since been abandoned in favour of drop lines.

The existence of a number of different vessel types and fishing methods has significant implications both for the analysis of the data (effort must be standardised) and management of the fishery.

Demersal fish represent 20-40% of all landings by Seychelles national vessels. Demersal catches have fluctuated over time reaching 2,285 tonnes in 1991 for a total effort of 77,815 mandays effort. By location, around 50% of the demersal fish catch derives from inshore locations on the Mahe Plateau exploited by small boats and whalers (between 43% and 60% of the catch during the period 1985-1994). This is the most heavily exploited part of the fishery. By comparison all other locations are lightly exploited and removals with time have been intermittent. For example, the Providence / Farquhar Group yielded 19% of all demersal landings in 1992 but had experienced no, or very light fishing since the previous mothership venture to that location in 1976-77. Whilst total removals from distant locations have been low, yield per unit area has been consistently high from the south of the Mahe Plateau, and relatively high intermittently in other locations, e.g. Providence / Farquhar Group. Available information indicates that locations distant from the centres of population are fished seldom by whalers, and most of the catch from these locations was derived from the schooner and mothership operations.

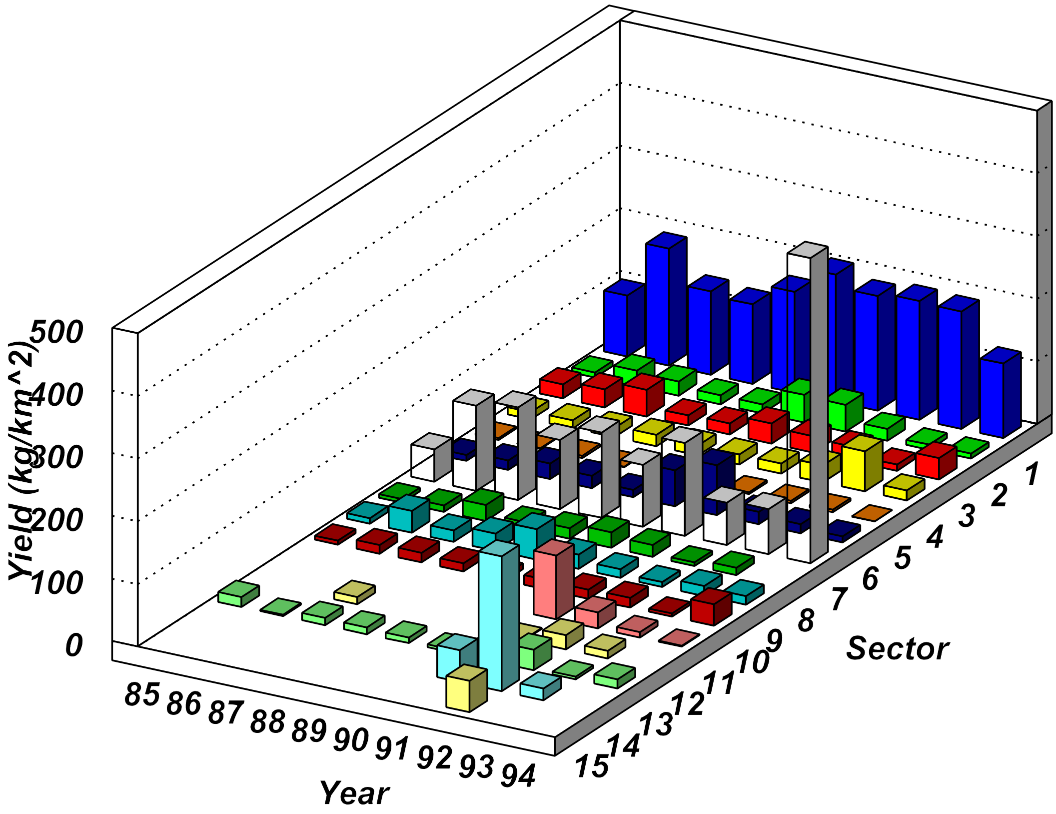


Figure 3.3 : Total yield per unit area (kg km^{-2}) for all boat categories, by Seychelles fishing location, 1985-1994 (Sectors 1-10 are on the Mahe Plateau, 11 is the banks to the south of the Mahe Plateau, 12 is Platte, 13 the Amirantes, 14 Providence/Farquhar and 15 is Astove/Cosmoledo).

Catch rates vary by boat-gear category and by location but are typically in the range 75-100 kg.manday^{-1}. Similarly, species composition varies by boat and location being a result of the combination of fishing method (usually hooks and lines), depth, substrate type and latitude. The rough coral and rubble grounds of the Mascarene Plateau support populations of lutjanids, lethrinids and serranids taken by handlines, and scarids, trigger fish, goat fish and unicorn fish not usually accessible to the line fishery. Species of these families, and additionally nemipterids and round scads are also reported from the shallow sandy areas of the banks accessible to trawlers. Although trawl surveys have documented a large number of demersal species in this region (Birkett, 1979; Kunzell, et al, 1983; see also, FAO/IOP, 1979; Tarbit, 1980), the variety of fish available to a handline fishery is limited and fish population densities are less on the sandy trawlable areas than rough grounds. Consequently, the handline fishery is only feasible over those parts of the total bank area which have suitable habitat. Trawling is not permitted in Seychelles.

Species composition varies with latitude in the Mascarene region. Birkett (1979) reported that lutjanids were the predominant type of demersal fishes taken in trawls on the Seychelles bank, but were of little importance except locally on the Saya de Malha bank, and virtually absent further south. Species composition in the Aldabra-Cosmoledo and Providence-Farquhar groups is similar but different from that in the Amirantes 320 km to the north (lutjanids are less important than the serranids and lethrinids). However, Birkett's observation were over greater distances and how much these differences relate to latitude, fishing history or substrate is not clear. Whilst the Amirantes and Mahe plateaux are at similar latitude, and lutjanids are predominant in catches from both, the coralline Amirantes support a different species composition to the granitic Mahe Plateau, and in commercial terms a greater proportion of high value species occur on the latter (see Lablache and Carrara, 1984). Species composition also differs on a fine scale between banks within the same general location (see Mees, 1991; 1992c), presumably related to substrate differences. Environmental variation is clearly important.

The predominant species from Seychelles demersal fishery were *L. sebae, A. virescens, P. filamentosus, E. chlorostigma* and *L. nebulosus*. By location, *L. sebae* was only important in catches from the Mahe Plateau and predominantly from inshore areas. The target of the deeper water fishery was *P. filamentosus*.

■ Mauritius

The Mauritian banks (Table 3.1) and Chagos Archipelago are exploited from refrigerated mother-vessels, 20-60m in length, carrying up to 20 6-8m pirogues or dories. Commonly the mother-vessels are converted second hand tuna long-line vessels. Each dory carries 3 men who fish using hand-lines rigged with 8-10 baited hooks. Other methods have been tried but found not to be commercially suitable (Ardil, 1986). Apart from the Saya de Malha bank, fishing has occurred from Mauritian or chartered vessels only (during the late 1970's in addition to Mauritian vessels, the fleet included two chartered Korean vessels). On Saya de Malha vessels from Reunion and occasionally Seychelles have also exploited the resources, and Russian trawling ventures have occurred in the past.

The annual catch from the banks only became substantial in the late 1960's. The number of vessels fell from 8 in 1977 to 3 in 1980, which included the two charter vessels. Catches fell from 3835 to 1686 tons. In 1982 steps were taken to revitalise the banks fishery (Wijkstrom and Kroepelein, 1979) including decontrolling the price of fish, reducing harbour dues, removing duty on fishing gears, equipment and spare parts, waiving import duty on vessels acquired by Mauritian companies, and providing adequate port facilities. Reinvestment in vessels occurred and the fleet has grown to 17 in 1995, although not all vessels have actively fished each year. Presently, the fishery employs up to 1000 fishermen, of which only about 500 are regularly active and spend about 150 days fishing per annum. 150 fishermen are engaged from the Republic of Madagascar. The catch in 1994 was estimated to be 5,444 tonnes of which 58% derived from Saya de Malha, 29% from Nazareth, 7% from St. Brandon, and 6% from Chagos.

Catch rates vary by location, and are significantly less at Chagos (around 50 kg.manday^{-1}) than the Mauritian banks (80-100 kg.manday^{-1}). The banks fisheries are considered to be fully exploited (except Chagos) and catch rates have declined in recent years. A decrease in the mean length of the principal species, *L. mahsena*, has also been reported.

It has been stated that species composition varies with latitude in the Mascarene region. Lutjanids, predominant on the Mahe Plateau, are of little importance except locally on the Saya de Malha bank, and are virtually absent further south where lethrinids predominate. However, populations of lutjanids and serranids do occur on the deep outer slopes of all banks in the Mascarene region. Despite good catch rates, these tend to be avoided in the south by Mauritian vessels due to the potential for ciguatera poisoning, and the Mauritian banks fishery operates chiefly in shallow water (<50m). The single species, *L. mahsena* is reported to constitute around 80-90% of the catch from shallow water (Ardil, 1986; Bertrand et al, 1986). However, during a 2 week period in April 1995 observers recorded that only 31% and 45% of the catch consisted of this species on St Brandon and Nazareth banks respectively (MRAG, 1996a;b). This suggests that the reported monospecific nature of the Mauritian banks fishery requires closer investigation.

In the Chagos, 36 demersal species from the families lethrinidae, lutjanidae and serranidae were identified during a BIOT inshore fishery observer programme in 1994, and lutjanids formed 44% of the catch, lethrinids 28%, serranids 16% and other fish 12%. Deeper water *Pristipomoides* species are becoming more important from Chagos, but normally shallow water species are the target. Species composition varies by location within the Archipelago, and *L mahsena* is important from the Great Chagos bank. The fishery is multi-species, however, and in 1994 its proportion in the catch did not exceed 25% (Maximum 24.5% at Eastern Chagos Bank, MRAG, 1995a).

The species composition of catches from the Chagos Archipelago is similar to that from the Seychelles. Thus, the more northerly Seychelles Banks and the Chagos Archipelago are predominantly multi-species in nature with snappers being the most commonly caught demersal fish. Further south on the Mauritian banks emperors are the most common. For the purposes of the multi-species tropical fisheries project, the Mauritian banks can thus be regarded as mono-specific.

3.3 Potential yield from banks and deep slope fish

3.3.1 Tonga

Biomass estimates for deep reef slope resources have been derived from depletion studies in the Pacific (eg. Ralston and Polovina, 1982; Polovina, 1986; Polovina *et al*. 1990). Both the short term Leslie model (eg. Polovina, 1986) and longer term Allen model (eg. Langi *et al*, 1992) have been applied. The convention of expressing biomass densities in tonnes per nautical mile of the 100 fathom (approximately 200m) isobath has been adopted in most reports from the Pacific from early work in Hawaii (Ralston and Polovina, 1982) and the Marianas (Polovina and Ralston, 1986). Biomass (B_0) estimates for a guild of all deep reef slope fish caught by lines from the Pacific (Polovina *et al,* 1990) are :

Reef / island fishing grounds, 0.2-2.3 t.nm^{-1}, median value 0.7 t.nm^{-1} (0.38 t.km^{-1})

Seamounts, 1.4-8.5 t.nm^{-1}, median value 2.7 t.nm^{-1} (1.46 t.km^{-1})

Yield (MSY) was determined using the methods of Gulland (1971), Pauly (1982) and Beddington and Cooke (1983). Depending on the method and value of natural mortality applied, yield was between 6% and 34% of biomass (B_0). For median values of (B_0) MSY was in the range :

Reef / island fishing grounds, 0.04-0.27 t.nm^{-1} (0.07-0.50 t.km^{-1})

Seamounts, 0.16-0.92 t.nm^{-1} (0.30-1.71 t.km^{-1})

Latu and Tulua (1989) and Langi *et al* (1992) applied the Allen model to data for Tongan seamounts and derived MSY estimates of 0.74 t.nm^{-1} and 0.22-0.67 t.nm^{-1} respectively. This study examined an extended data set for a number of seamounts including those examined by other authors but an adequate fit of model to data was only achieved for 2 of 15 examined. MSY was estimated to be in the range 0.22-0.34 t.nm^{-1} for a guild of the six principal species assuming that every trip to the seamount was recorded (as had Latu and Tulua (1989) and Langi *et al* (1992)). However, for the 1986-1991 Tongan data set, sampling frequency by individual seamount is not known. If it is assumed that the overall sampling frequency reported applies also to each seamount, the estimate of MSY becomes 0.50-0.77 t.nm^{-1}. Errors in the sampling frequency far outweigh any error in the estimate of natural mortality used in the depletion model (MRAG, 1994).

Langi and Langi (1987) estimated the total length of the 200m contour around banks and seamounts to be 960 nm, and 294 nm for seamounts alone. This almost certainly is an underestimate as incomplete bathymetric data is available for the region, and new mounts are still being discovered. Using this estimate, the surplus yield from Tongan seamount fisheries has been estimated as 217 t (Langi et al, 1992), 64-198 t (Latu and Tulua, 1989), and between 64 t and 225 t for a guild of the six main species from all depths, dependant upon whether data available for analysis is a sample or census of fishing trips to studied seamounts (MRAG, 1994). The banks, are likely to have a lower yield than the seamounts (Polovina *et al*, 1990). Dalzell and Preston (1992) applied a biomass (B_0) estimate of 0.7 tnm^{-1} to the banks and derived a total MSY estimate of 113-338 t for all banks and mounts. The estimated yield for banks only using this value is 112 t.

MRAG (1994) found that application of the Allen model in CEDA (MRAG, 1992a) fitted catch and effort data for a multispecies guild of the six principal species better than that of individual species alone. Extrapolating single species yield estimates to determine the yield of the whole complex resulted in larger estimates than those reported above (up to 768 t for seamounts). However, targeting and depth changes complicate this approach. Standardisation of catch rate data for seasonal and depth variations of individual species indicated that depletion was only significant at one of the seamounts studied (1403). A Fox depletion model was fitted to standardised data for *P. filamentosus* for this seamount. Surplus yield was estimated as 11.8 t, equivalent to 0.34 t nm^{-1} (Mees and Rossouw, 1995), which compares favourably with the estimates derived from the Allen model for that species (0.13-0.41 t nm^{-1}).

Latu and Tulua (1992) applied surplus production models to a guild of the principal species and a guild of *P. filamentosus* and *E. coruscans* for annually aggregated data from all locations, and estimated the sustainable yield from seamounts to be 350 t and 214 t respectively. MRAG (1994) repeated these analyses using biomass dynamic production models in CEDA (MRAG, 1992a). Whilst it was possible to derive biomass and yield estimates, data poorly fitted the models. Sequential fishing of seamounts, depth and target species changes over time all complicated the analysis. It is not considered appropriate to apply these models to aggregated seamount data (see Chapter 3).

Total catches from the Tongan banks and seamount fishery are close to, or exceed the estimated sustainable yield. To a certain extent, catches are maintained by the exploitation of new seamounts (see MRAG, 1994). Moffit (1993) also suggested that landings in excess of MSY may be expected due to a fishing down process. Catches will decline until an equilibrium yield is achieved. However, fishing mortality (F) is high in relation to natural mortality (M, Latu and Tulua, 1990). Recovery of depleted seamounts is unpredictable (see Chapter 3) and the number of new mounts is limited. Clearly there is a danger of overfishing and careful monitoring of the fishery is required.

3.3.2 Seychelles and Mauritius

Shallow sandy banks with coral outcrops and sea grass beds in general support fish densities of 1.5-2 tons/km^2. In deeper areas and on those substrates which are of dense coral, the density of fish may vary between 9.9 and 10.9 tons/km^2 (FAO/IOP, 1979). Early resource assessments on banks in the Mascarene Plateau related to demersal trawl surveys on suitable flat sandy substrates using the swept area method (Birkett, 1979; FAO/IOP, 1979). The estimates derived included all demersal species taken by the trawl (Mahe Plateau, 2.68 t.km^{-2}; Saya de Malha Bank, 1.8 t.km^{-2}; Nazareth Bank, 0.6 t.km^{-2}). Estimates for line caught fish only will suggest much lower figures. However, against this, coral substrates suitable for hand line caught species support greater densities of fish (2.6-4.8 t.km^{-2} on Seychelles coraline banks Tarbit reported in FAO/IOP, 1979). Biomass and yield estimates based on yield per unit area require accurate estimates of the fishable habitat (see Chapter 3).

Production models have not been successfully applied to long time-series data from Seychelles or Mauritius, although a Leslie depletion model was fitted to daily information from Seychelles, Mees (1992a;1993). Length and age based resource assessments have also been applied to key species from these fisheries (*L. mahsena*, Lebeau and Cueff, 1976; Bautil and Samboo, 1986; *L. sebae*, Lablache and Carrara, 1988; *P. filamentosus*, Mees, 1993; see also Mees, 1992b; Technical Reports, MRAG, 1996a; 1996b). Lablache and Carrara (1988) derived an estimate of the maximum sustainable yield of handline caught demersal species on the Mahe Plateau of 209 $kgkm^{-2}$. Mees (1992b) estimated the MSY of the shallow banks of the Mahe Plateau to be in the range 120-360 $kgkm^{-2}$ for fishable areas (approximately 60% of the total plateau area), and 1,300-1,500 $kgkm^{-2}$ for the intermediate depth band (75-150 m). Sanders (1988) assumed a yield of 220 $kgkm^{-2}$ for the Saya de Malha Bank, and current data suggests a yield of around 200 kg km^{-2} per annum for the St Brandon, Nazareth and Saya de Malha Banks is appropriate. Catch rate data for the Chagos Archipelago suggested that the abundance of demersal resources was between half and two thirds that on the Nazareth and Saya de Malha Banks. The current best estimates of the yield of a guild of handline caught demersal fish species (Table 3.1) employ the following values : Seychelles, a conservative value of 168 $kgkm^{-2}$ (MSY, Biomass = 1,400 $kgkm^{-2}$) for the fishable areas of the shallow depth strata (assumed to be 60% of total bank area), and 1,375 $kgkm^{-2}$ (MSY, Biomass=5,500 $kgkm^{-2}$) for the intermediate depth stratum; Mauritius, a shallow water yield of 200 kg km^{-2}, and 100 kg km^{-2} for the Chagos Archipelago for fishable areas of the banks (MRAG 1996b).

4 CHARACTERISTICS OF DEMERSAL FISH SPECIES

4.1 Background

The banks and deep reef slope fisheries target demersal species of lethrinidae, lutjanidae and serranidae. Members of these families are widely distributed throughout tropical and sub-tropical waters of the world. They may be characterised as long lived, slow growing bottom dwelling carnivores with relatively low rates of natural mortality. They have limited productive capacity and are vulnerable to overfishing. They are highly valued food fish fetching a high price on world markets.

With the exception of the Mauritian banks, case study fisheries are multispecies in nature. Catch and effort statistics are limited to principal species in the catch whilst key indicator species have been identified for biological assessment (Section 2.2). Details of the ecology of the principal species recorded in landing statistics from study sites are indicated in Table 4.1. and Sections 4.2-4.5. Estimates of population demographic parameters for these species have been derived from available data and the literature (Tables 4.2 and 4.3) and are described in Sections 4.6-4.10. Length based methods have been used in this study. From biological data, reproductive and length-weight parameters have been derived. Growth, mortality, and length at first capture were determined from length frequency data. Accurate parameter estimates are required for management simulations with MIDAS, and full details of analytical procedures adopted to derive the values indicated in Tables 4.2 and 4.3 occur in Technical Reports (MRAG 1994; 1996a; 1996b), which should be refered to for more detail and for information relating to any variation in these parameters over space and time.

References for Tables 4.2 and 4.3 are as follows. 1. Dalzell et al (1992); 2. Bautil and Samboo (1988); 3. Aldonov and Druzhinin (1979); 4.Loubens (1990); 5. Mees (1992b); 6. Lebeau and Cueff (1975); 7. Samboo (1987); 8. Moussac (1988); 9. Latu and Tulua (1992); 10. Lee and AlBaz (1989, Total Length); 11. Sanders and Morgan (1989, Total Length); 12. Moussac (1986); 13. Carpenter and Allen (1989); 14. Ralston and Williams (1988); 15. Brouard and Grandperrin (1985); 16. Munro (1983); 17. Polovina and Ralston, 1986); 18. Allen (1985)

Species	Synonym	Country	Substrate	Depth	Behaviour	Diet	Max TL	Ref
Gymnocranius euanus	*G. japonicus*	T	Sand/rubble/coral	15-50		Benthic gastropods	45	Carpenter and Allen (1989)
Gymnocranius grandoculis	*G. robinsonii*	S	Sand/coral	50-100		Carnivore, benthic macrofauna, fish	80	Carpenter and Allen (1989)
Lethrinus miniatus	*L. chrysostomus*	T		5-30	Schooling/nocturnal	Carnivore, benthic macrofauna, fish (3.1%)	90	Carpenter and Allen (1989)
Lethrinus crocineus		S	Sand/corals	to 150		?	45	Carpenter and Allen (1989)
Lethrinus microdon	*L. elongatus*	S	Sand/corals	to 80	Schooling	Carnivore , benthic macrofauna, fish	70	Carpenter and Allen (1989)
Lethrinus mahsena		SMC	Sand/Corals, associated sea grass	to 100	?	Carnivore, benthic macrofauna, fish	65	Carpenter and Allen (1989)
Lethrinus nebulosus		SC	Nearshore/off shore coral reefs, lagoons, seagrass, sand/rock	to 75	Adults solitary or small schools, juveniles in large schools	Carnivore, benthic macrofauna, fish (1.8%)	80	Carpenter and Allen (1989)
Lethrinus rubriopeculatus		TC	Sand/rubble on outer reef slopes	to 160	?	Carnivore, Benthic macrofauna and fish	50	Carpenter and Allen (1989)
Lethrinus variegatus		C	Reef associated		?	Carnivore, benthic invertebrates	20	Carpenter and Allen (1989)
Apharaeus rutilans		S	Reef/rocky bottom	to 100		Carnivore, fish, squid, crustaceans	80	Allen (1985)
Aprion virescens		SC	Inshore reefs/sand/coral	to 450	Rarely schools?/ occasionally migrates	Carnivore, macrofauna, fish	110	Allen (1985);Torres (1991)
Etelis carbunculus		T	Rocky bottoms	90-300	Occaisionally schools, diurnal	Carnivore, mainly crustaceans	127	Bagnis et al (1972) Smith and Kostlan (1991)
Etelis coruscans		T	Rocky bottoms	100-300	Rarely schools; diurnal	Carnivore, small fish, squid, crustaceans	96	Allen (1985); Langi and Langi (1987)
Lutjanus bohar		SC	Coral /rocky bottoms	10-70	Rarely schools	Carnivore, mainly fish and benthic macrofauna	75	Allen (1985)
Lutjanus sanguineus	*L. coccineus*	S	Coral and rock reefs	to 100	Occaisionally migrates	Carnivore	90	Allen (1985); Fischer et al (1990)Torres (1991)
Lutjanus gibbus		SC	Juveniles in seagrass, mixed sand and coral; Adults in deeper lagoon and seaward reefs	6-30	Occaisionally schools,	Carnivore, benthic macrofauna, fish rare (1-5%)	72	Bagnis et al (1972); Allen (1985)
Lutjanus sebae		S	Near reefs on adjacent sand flats; juveniles in mangrove areas	10-100	Probably schools? sometimes trawled	Carnivore, invertebrates and fish	100	Allen (1985); Torres (1991)
Pristipomoides filamentosus		TSC	Rock bottoms, outcrops and promontories	90-360	Schools	Carnivore, including fish (33.5%)	89	Allen (1985); Mees (1992)
Pristipomoides flavipinnis		T	Rock bottoms	90-360	?	Carnivore, mainly fish, crustaceans, squid.	60	Allen (1985)
Epinephelus chlorostigma		S	Outer reef slopes/seagrass/mud bottoms	4-280	?	Carnivore mainly stomatopods/crabs, some fish	75	Allen (1985) Fischer et al (1990) Randall and Heemstra (1991)
Epinephelus morhua		TSC		120-370			90	Gheno (1968) Bianci (1985)
Epinephelus multinotatus		S	Reef associated	1-100	?	Carnivore, small fish/crabs	100	Randall and Heemstra (1991)
Epinephelus septemfasciatus		T	Rocky bottoms					
Plectropomus leopardus		C	Reef associated					
Plectropomus maculatus		C	Reef associated					
Variola albimarginata		C	Reef associated					
Variola loutii		SC	Reef associated	3-240	?	Carnivore, fish/invertebrates	81	Heemstra and Randall (1993)

Table 4.1 : The ecology of principal species recorded in the catch from case study countries (T=Tonga, S= Seychelles, M=Mauritian banks, including C = Chagos (BIOT))

Lethrinidae					**Serranidae**		
Parameter	Lethrinus	Lethrinus	Lethrinus	Lethrinus	Epinephelus	Epinephelus	Epinephelus

Estimate	mahsena	mahsena	nebulosus	variegatus	chlorostigma	morhua	septemfasciatus
Country	Seychelles	Mauritius	Seychelles	Seychelles	Seychelles	Tonga	Tonga
Growth							
K - non seasonal	0.320 [1]	0.100 [2]	0.210 [3]	0.430 [4]	0.175 [5]	0.230	0.207
Loo (FL)	58.9 [1]	61.7 [2]	71.6 [3]	30.3 [4]	64.5 [5]	76.6	189.9
to		-0.710 [2]				-0.900	-0.330
Length/Weight							
Lmin	26.5		18.5	23.5	19.9	21.0	32.0
Lmax	43.3	48.9 [6]	79.0	39.0	80.7	102.0	172.0
a	0.00001749	0.00008060 [7]	0.00003355	0.00027200	0.00000612 [8]	0.00003167 [9]	0.00022713 [9]
b	3.06	2.74 [7]	2.84	2.22	3.25 [8]	2.83 [9]	2.45 [9]
Reproduction							
LengthMatMin-males	30.0 [6]	30.0 [6]	38?		39.0 [12]		
LengthMatMin-females	25.0 [6]	25.0 [6]	32.5 [10]		31.0 [12]		
Lm25-females							
Lm50-females	NA	NA	45.6 [11]	NA	NA	51.0	86.0
Lm75-females							
Spawning period			Mar & Oct-Nov				
Sex ratio M:F	1.33		0.62	1.63			
Mortality							
M	0.22 [2]	0.22 [2]	0.51 [13]	NA	0.43	0.40	0.25
K	0.10 [2]	0.10 [2]	0.23 [13]		0.18 [5]	0.23	0.15
Loo	61.7 [2]	61.7 [2]	80.0 [13]		64.5 [5]	76.6	154.4
Temp. (C)					24	18	18
Length at Capture							
Lc	NA	NA	53.0	25.0	35.0	59.0	97.5
Lc25%			43.1	22.5	27.8		
Lc50%	29.5 [2]	29.5 [2]	47.3	24.0	29.7	46.2	86.5
Lc75%	31.2 [2]	31.2 [2]	50.7	25.6	31.5	55.8	93.2

Table 4.2 : Population demographic parameters derived for principal lethrinid and serranid species from case study fishery data, where available, and from the literature (shaded).

Lutjanidae

Parameter Estimate Country	Apharaeus rutilans Seychelles	Aprion virescens Seychelles	Etelis carbunculus Tonga	Etelis coruscans Tonga	Lutjanus bohar Seychelles	Lutjanus sebae Seychelles	Pristipomoides filamentosus Seychelles	Pristipomoides filamentosus Tonga	Pristipomoides flavipinnis Tonga
Growth									
K - non seasonal	0.163[14]	0.321	0.129[15]	0.110	0.330[16]	0.180	0.244	0.220	0.268[17]
Loo (FL)	123.0[14]	94.7	127.0[15]	107.2	66.0[16]	99.1	75.8	77.6	49.0[17]
to	-0.360[14]	-0.190	1.410[15]	-0.636		-0.630	-0.260	-0.440	-1.010[17]
K - seasonal		0.321				0.193	0.240		
Loo		101.5				97.4	76.2		
to		-0.650				-0.350	-0.280		
C		0.19				0.64	0.42		
ts		0.23				-0.24	-0.26		
Length/Weight									
Lmin	38.2	21.0	13.0	17.0	25.4	14.1	21.5	18.0	20.0
Lmax	92.5	96.3	136.0	99.0	74.0	98.3	87.8	106.0	63.0
a	0.00004157	0.00002704	0.00001940[9]	0.00007485[9]	0.00001304	0.00001570[8]	0.00004283	0.00013914[9]	0.00007120[9]
b	2.723	2.831	2.980[9]	2.620[9]	3.127	3.020[8]	2.757	2.490[9]	2.670[9]
Reproduction									
LengthMatMin-males		34.0					31.0		
LengthMatMin-females		32.0				33.5	29.0		
Lm25-females		38 or 63				60.0	45.0		
Lm50-females	NA	42 or 69	68.0	49.5	50.0[18]	64.0	51.0	53.0	31.5
Lm75-females		74.0				70.0	59.0		
Spawning period		All year peak Feb-May & Oct-dec				all year	all year, peak Feb-May & Oct-Dec		
Sex ratio M:F	1.06	1.24			1.00	1.07	1.02		
Mortality									
M	NA	0.56	0.25	0.24	NA	0.41	0.48	0.38	0.50
K		0.32	0.13[15]	0.11		0.19	0.24	0.22	0.27[17]
Loo		101.5	127.0[15]	107.2		97.4	76.2	77.6	49.0[17]
Temp. (C)		24	18	18		24	22	18	18
Length at Capture									
Lc	41.0	69.0	76.5	59.0	NA	61.0	41.0	63.0	39.0
Lc25%	38.6	63.9				56.5	32.7		
Lc50%	39.4	67.0	67.3	56.8		59.3	37.1	60.2	37.4
Lc75%	40.9	69.2	73.2	59.3		61.4	39.8	62.5	39.3

Table 4.3 : Parameters derived for principal lutjanid species from case study data, where available, and from the literature (shaded).

4.2 Depth association and community structure

With the exception of the southern Indian Ocean banks fished by Mauritian vessels, the bank and deep slope fisheries studied are multi species in nature. Target species occur across a diversity of habitats and depths and in association with other teleost fish, elasmobranchs and invertebrates not recorded in catch statistics. Fish community structure in this report thus relates only to target demersal species assemblages. Such assemblages may show considerable overlap across habitat types but are clearly separated by depth, which will be the principal factor used to describe community structure.

Distinct depth associations are reported for certain species of emperors, snappers and groupers (Sundberg and Richards, 1984;Dalzell and Preston, 1992) and many snappers, but few groupers are restricted to feeding in deep water (Parrish, 1987). However, the same species have been observed to occur at different depths in the Pacific and Indian Oceans, and this is particularly noted for *Pristipomoides filamentosus*. For the Pacific, Moffit (1993) has classified the following depth strata : Shallow, to 120 m; Intermediate, 120-240 m; Deep > 240 m. For Seychelles, Mees (1992b) classified the equivalent strata as 0-75m; 75-150m; >150m, and it has subsequently been shown that these strata also apply in the Chagos Archipelago.

The emperor family (Lethrinidae) are bottom feeding carnivorous fish found usually in shallow coastal waters on or near reefs, although some species are reported at greater depths (eg *L. rubrioperculatus*). Lethrinids are not reported to be territorial and may be solitary or form schools. The snapper family (Lutjanidae) are largely confined to continental shelves and slopes and to corresponding depths around islands. Adults are usually associated with the bottom. The genus *Lutjanus* (Bloch,1790) is the largest of this family consisting primarily of inhabitants of shallow reefs. Species of the genus *Pristipomoides* (Bleeker, 1852) occur at intermediate depths, often schooling around rocky outcrops and promontories (Ralston *et al*, 1986) whilst Eteline snappers are deep water species. Groupers (Serranidae) are moderate to large size and mostly occur in shallow areas, although some are deep slope species. Groupers in general are more sedentary and territorial than the snappers or emperors and more dependant upon a hard substrate. However, of the species included in this study, *Epinephelus morhua* and *E. septemfasciatus* have been taken in trawls implying that they frequent soft bottom habitats and may form loose aggregations. In general, groupers may be less dependant upon hard bottom substrates at depth (Parrish, 1987). For each family, schooling behaviour is reported more frequently for juveniles than adults. Spawning aggregations may however, occur even for the solitary species at certain times of the year.

A commonly reported trend is that juveniles occur in shallow water whilst the adults are found in deeper water (eg. Parrish, 1987). Juveniles also tend to occur in different habitats than adults, possibly reflecting different feeding requirements, but also as a means of reducing predation (*L. gibbus, L. sebae*, Table 4.1). Nursery grounds for juvenile deep slope snappers and groupers are little known. In Hawaii, juveniles of *P. filamentosus* have been found on flat, featureless shallow banks avoiding high relief areas where the adults occur, and similarly juveniles of the deep slope grouper *Epinephelus quernus* are found in shallow water (Moffit, 1993). Ralston and Williams (1988), however, found that within deep slope species size was poorly correlated with depth. Similarly, for the Tongan study site no statistical significance could be attributed to an apparent trend for fork length to increase with depth, except in the case of *Etelis carbunculus* (Mees, 1994). By contrast, the size distribution of *Lethrinus nebulosus* is dependant upon depth (Egretaud, 1992).

4.2.1 Tonga

Depth distributions for the principal species caught in Tonga (Fig. 4.1) were examined by means of cluster analysis to determine depth associations between species (Fig. 4.2). The shallow water banks fishery species, *G. japonicus, L. chrysostomus,* and *L. rubrioperculatus* clustered separately from the predominantly sea-mount (and deep slope) species. The latter clustered into two groups representing the intermediate (*P. filamentosus, P. flavipinnis*, grouper Sp8) and deep (*E. coruscans, E. carbunculus, E. septemfasciatus*) water strata suggested by Moffit (1993). Within the intermediate stratum the *Pristipomoides* species were closely associated and species 8 showed a similar depth distribution. *Epinephelis morhua* clustered in the intermediate depth stratum but was separated from the other species. Similarly within the deep stratum, *E. septemfasciatus* clustered separately from the Eteline snappers which were closely associated. These grouper species were more evenly represented in the catch over a wide depth range, whilst the snappers tended to be predominant at certain depths.

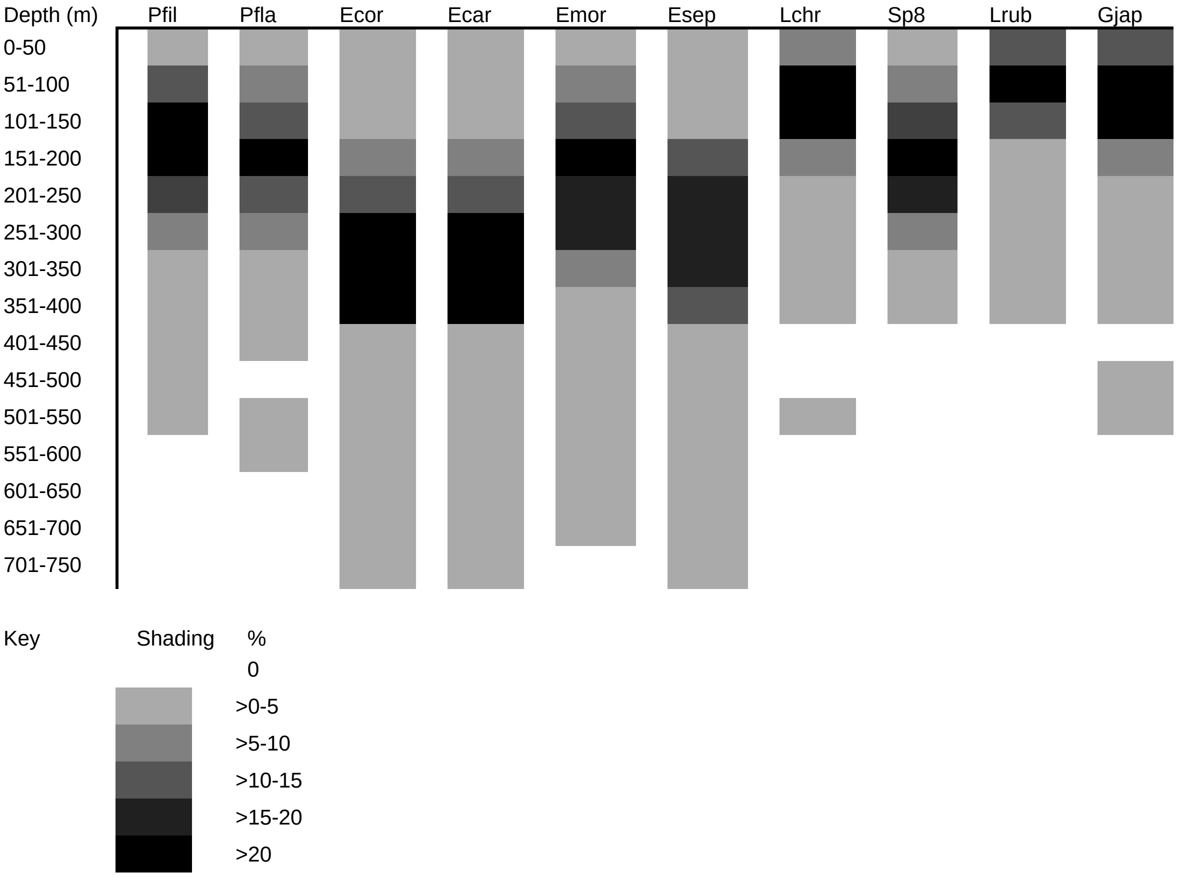


Figure 4.1 : The proportion of the total catch (1987-1994) of each species taken at depth from all locations of the Tongan banks and seamount fishery.

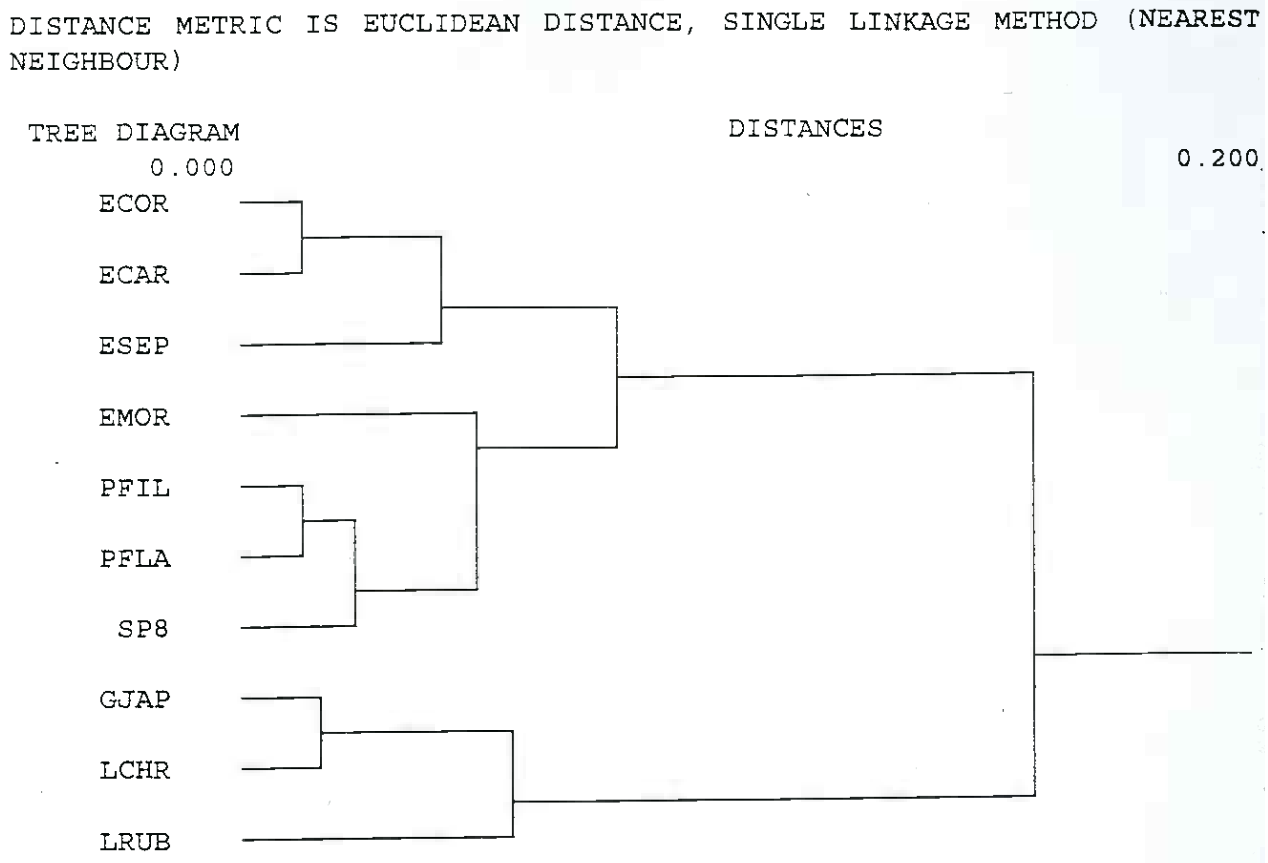


Figure 4.2 : Cluster analysis for Tongan banks and deep slope species based on the proportion of the catch of that species taken at depth to illustrate depth association and community structure.

4.2.2 Seychelles

Cluster analysis of the depth distributions of demersal species caught at the Mahe Plateau in Seychelles (Fig. 4.3; Fig. 4.4) provides a less clear picture of species associations than was the case for Tonga. There are more species and the depth range fished was considerably less. However, it is apparent that *A. rutilans, P. filamentosus* and *E. morhua* are distinct from the remaining species. These are known to be caught predominantly on the drop-off of the banks and plateaus in intermediate depths. The remaining species are caught over the banks and *L. gibbus* clusters separately being caught predominantly in shallow water. The rest are all taken at similar depths. *P. filamentosus* and *E. morhua* in Seychelles are predominant at significantly lesser depths than in Tonga. This is not considered to be a consequence of lack of fishing at greater depths. Certainly for *P. filamentosus* fishing trials have demonstrated that this species is predominant between 70-100 m.

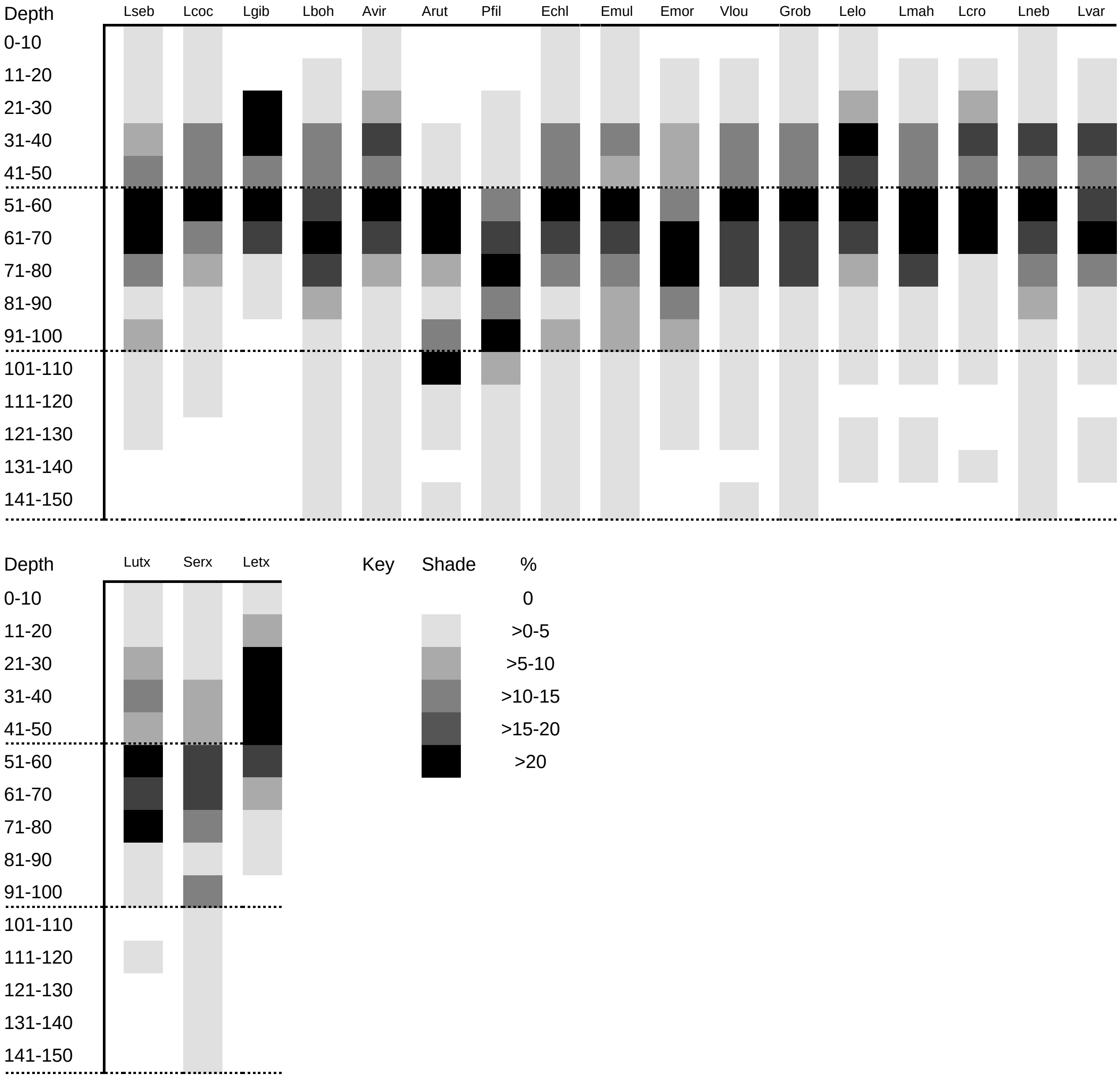
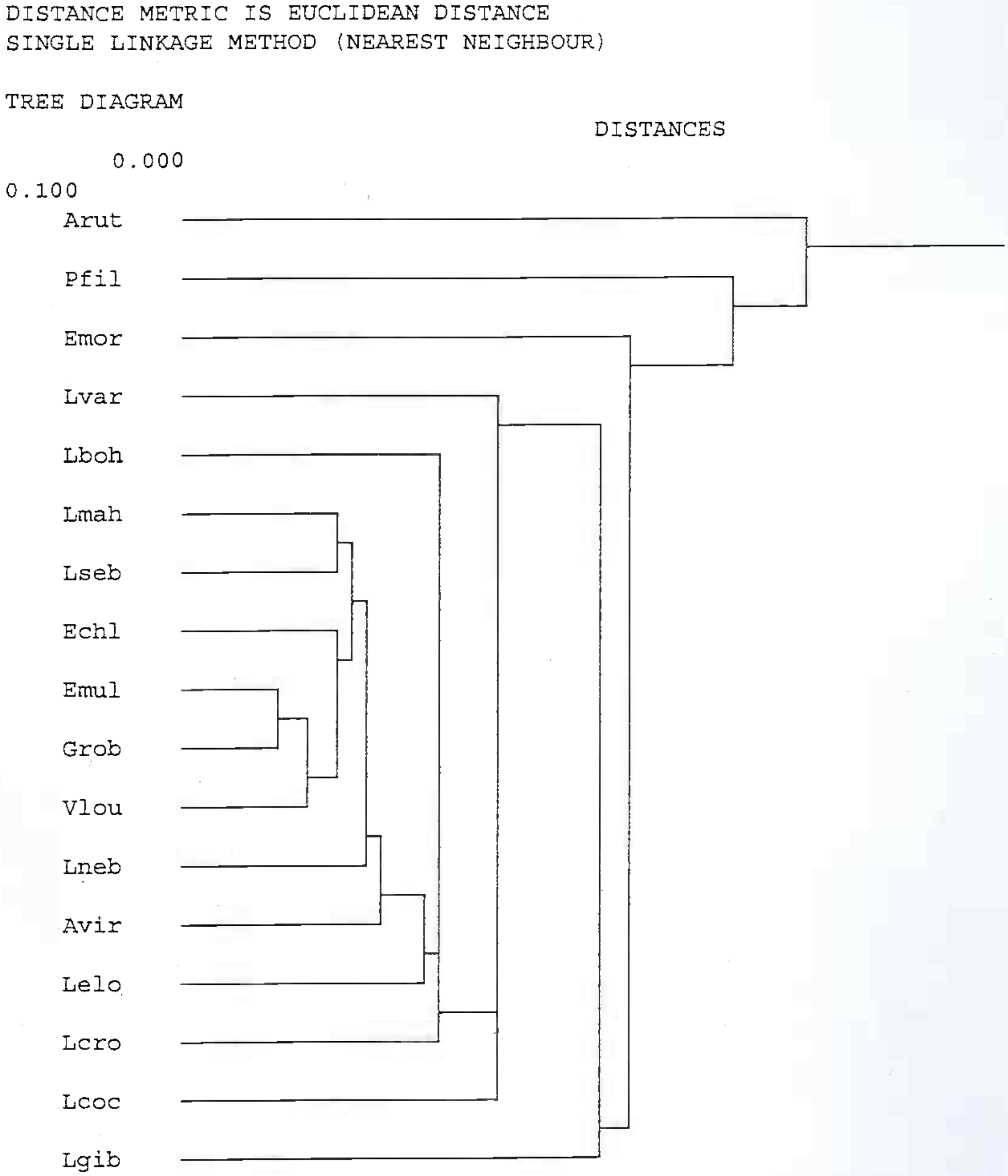


Figure 4.3 : The proportion of the total catch by species taken at depth by all vessel types fishing on the Mahe Plateau (1985-1994) where depth was recorded illustrating depth associations and community structure.



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DISTANCE METRIC IS EUCLIDEAN DISTANCE
SINGLE LINKAGE METHOD (NEAREST NEIGHBOUR)

TREE DIAGRAM
                                          DISTANCES
      0.000
0.100
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Figure 4.4 : Cluster analysis for Seychelles demersal fishery species based on the proportion of the catch of that species taken at depth (Mahe Plateau only)

4.2.3 Mauritius

Logbook data did not record depth, and only a limited amount of information is available from recent BIOT Inshore Fishery logbooks for periods when the Mauritian vessels fished in Chagos (Fig. 4.5, Fig. 4.6). *L. mahsena* from the Mauritian banks is taken around 50 m depth, and in Chagos most fishing also occurred in shallow water on the banks rather than the drop off. *L bohar* and *L. gibbus* have clustered separately because they were caught in discrete depths whilst all other species were caught over a more dispersed depth range. Of these *Pristipomoides* species and *L. microdon* are closely associated being caught in slightly deeper water. However, the limited amount of data, particularly from greater depths restricts the usefulness of this analysis. Observer reports however, confirm that *P. filamentosus* is usually taken in deeper water and at the edge of the banks.

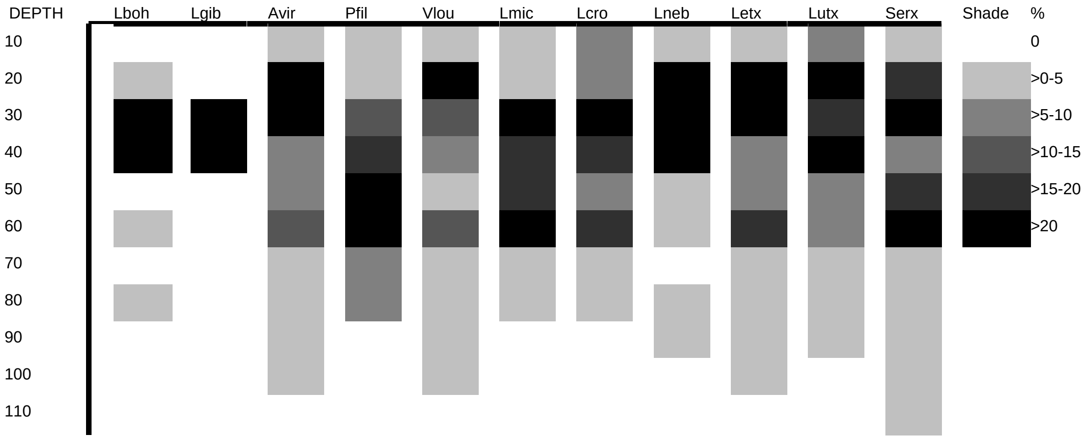

Figure 4.5 : Depth distribution and proportion of the catch taken at depth observed for certain species caught by Mauritian vessels fishing in the Chagos Archipelago (1993-1995)

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DISTANCE METRIC IS EUCLIDEAN DISTANCE
SINGLE LINKAGE METHOD (NEAREST NEIGHBOUR)

TREE DIAGRAM
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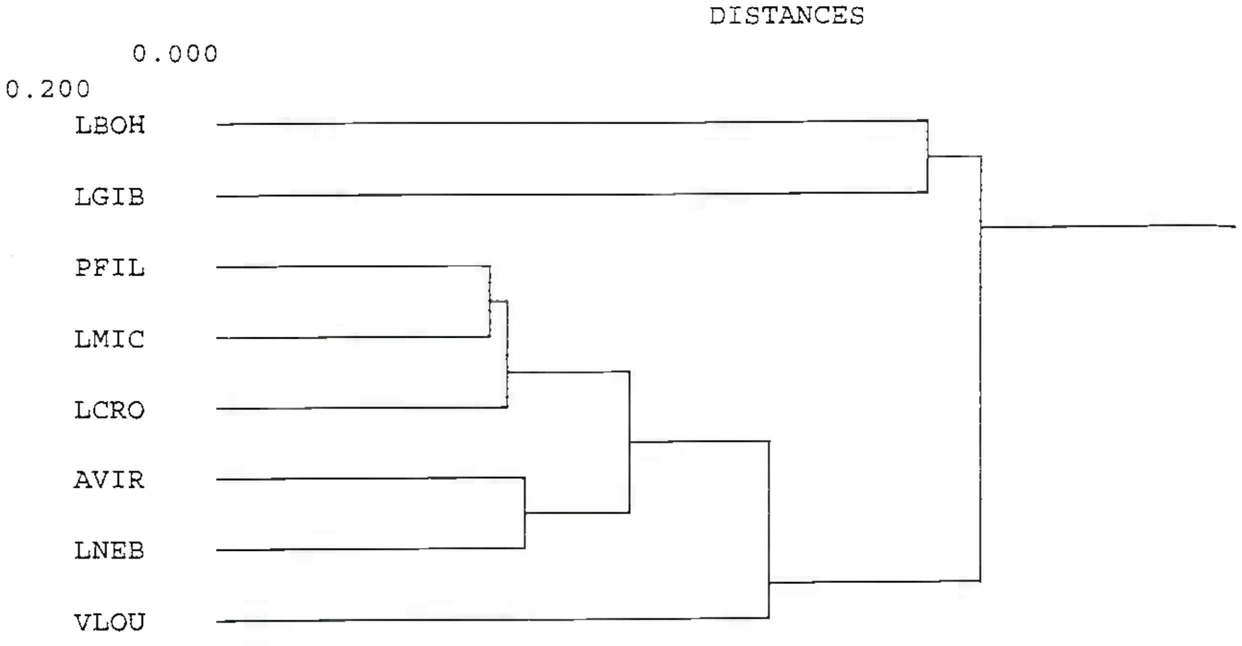

Figure 4.6 : Cluster analysis for Mauritian demersal fishery species based on the proportion of the catch of that species taken at depth (Chagos Archipelago only)

4.3 Habitat

In addition to depth, habitat and latitude affect species composition. Little direct information exists in relation to habitat. Within Tonga, the principal distinction relates to the predominantly shallow banks fisheries and deep sea mount fisheries. In the Indian ocean most fishing has occurred on the shallow banks and whilst the same families are represented, species differences occur between banks and plateaux. Furthermore, fishing history may also explain some of the differences in species assemblages, and this is investigated further in Chapter 5. Details of the proportions of each species represented in the catch are given elsewhere.

4.4 Trophic Relations

Gut contents and trophic relations were not examined for fish specimens from the study sites. Table 4.1 indicates the diet reported for principal species. The lethrinidae are carnivores feeding mostly on benthic macro fauna and predominantly invertebrates (Kulbicki, 1988). Small fish are also consumed, but where recorded they constitute only a small proportion of the diet (3.1% for *L. miniatus*; 1.8% for *L. nebulosus*, Table 4.1).

Lutjanids and serranids are characterised as generalised opportunistic carnivores. Their trophic biology is incompletely studied and especially little information exists for the deep slope species due to problems of regurgitation on hauling the fish and limited scientific collection. The diets of deep and shallow genera, however, include the same major prey groups, but the relative importance of prey items differs. Parrish (1987) has reviewed the available literature.

Fish formed more than 10% of the diet in 90% of lutjanid and 95% of serranid studies, were the dominant component in 56% and 72% of studies respectively, and were dominant in both deep and shallow water genera. A wide variety of fish were consumed from small schooling clupeids and engraulids to other predators such as muraenids and congrids. There is also evidence that members of these families eat other lutjanids and serranids (eg. Piedra, 1965; Rodrigues, 1974; Grimes, 1979; Parrish, 1987), mostly juveniles but predation on adults does occur. For deep water genera the prey tend to be primarily non fishery species such as monacanthidae and deepwater demersal cardinal fish (Parrish, 1987; Haight *et al*. 1993). However, other target species may be consumed. For example, *E. carbunculus* occurred in 2% of the guts studied of Hawaii's deep water grouper, *E. quernus* (Seki, 1984) and anecdotal evidence from fishermen in Tonga indicates that *E. coruscans* has been found in the guts of *E. septemfasciatus.* Zooplankton and benthic urochordates are also important components in the diets of deepwater genera and especially so for Pristipomoides species which Haight *et al* (1993) classified as zooplanktivores, whilst piscivores included *E. coruscans, E. carbunculus* and *A. virescens*.

Changes in composition of the diet have been reported with size and age for members of each of the families studied, e.g. for *L. nebulosus* (Egretaud, 1992); *P. leopardus* (Kingsford, 1992), but generally for snappers and groupers Parrish (1987) concluded that changes in diet with age are surprisingly small.

Feeding time is found to vary considerably within families but generally snappers tend to feed at night whilst groupers may feed at any time of day. Amongst the principal species at study sites the following are reported to feed at any time (*A. virescens, L. sebae*), at night (*L. miniatus, L. gibbus*) or during the day (*Etelis spp, L. bohar*)(Table 4.1, Parrish, 1987). Lethrinids tend to feed on the bottom, serranids hunt at the bottom, but have been observed to chase prey higher into the water column and lutjanids may feed on the bottom or higher in the water column. For example, *A. virescens* is not constrained by substrate association and may feed high in the water column (Parrish, 1987). In Seychelles, gill net trials indicated that *P. filamentosus* was taken at the bottom whilst, *A. rutilans* was caught higher in the water column (Bean, pers. comm) and is possibly a reason why the latter was less represented in demersal set hand line catches.

In conclusion, lethrinids, lutjanids and serranids appear to be generalist feeders with the ability to switch prey items, even across diverse prey groups, and this is particularly true for the lutjanids. Changes in the abundance of one or a few components of the diet would be unlikely to result in large changes of abundance of snappers and groupers (Parrish, 1987) and similarly these species are expected to be resilient to changes in the abundance of feeding competitors. Furthermore, information on the diet composition, depth and time of feeding suggests that considerable resource partitioning occurs amongst deep water snappers (Haight *et al*, 1993) and this is also likely to be true for shallow water genera, thus reducing intra specific competition.

4.5 Biological Interactions

Biological interactions relate to competition for food and / or space, and predator-prey responses within and between species. Knowledge of such interactions is a key feature in modelling these fisheries.

Species assemblages including lethrinids and lutjanids show considerable overlap across habitat types (Thollot and Kulbicki, 1988), and there is considerable overlap in prey species (particularly fish) taken by them, suggesting some level of competition for food. However, knowledge of the trophic relations of emperors, snappers and groupers suggests that whilst food competition may occur, strategies have developed to minimise intra specific competition. Owing to the generalist feeding nature of these genera, food is not a limiting factor suggesting that the demersal habitat dependency is more related to a need for shelter than food. Indeed Parrish (1987) indicated that there was little or no direct evidence concerning the importance of food as a limiting factor in growth, reproduction, or survival of wild populations of snappers and groupers.

With respect to space and habitat, whilst some species may form schools (see Table 4.1), solitary behaviour and territoriality is most common amongst serranids (eg Shpigel and Fishelson, 1991). Thus inter and intra specific competition for space is expected to limit grouper populations to a greater extent than those of the lethrinids and lutjanids. Smith (1961) suggested that whilst groupers are in food competition with other carnivores, particularly groupers, competition for space and shelter may be strongest. Habitat partitioning has been shown to occur for species of shallow water groupers (eg Shpiegel and Fishelson, 1989). This may reduce direct competition for food, but space is an important limiting factor and intra-specific dominance hierarchies may develop. Competition for habitat space is expected to be the most significant interaction, reflected through density dependant growth and mortality. Particularly in cases of territoriality, abundance will also be limited, but for the deep water groupers it is not clear to what extent territoriality occurs.

With respect to predation, it has been demonstrated that juveniles and sometimes adults may be eaten by other snappers and groupers. Lethrinids, however, consume primarily benthic invertebrates and so are more likely to be prey. Mechanisms to reduce inter-specific predation have developed (different depth and habitats for adults and juveniles). Intra-specific predation may occur, although all case study species may be classified loosely in the same trophic levels. Where predation is important, it tends to occur within the first year and predominantly immediately following settlement of the larvae (see Section 5.1).

4.5.1 Tonga

Based on knowledge of depth associations (Section 4.2) and trophic relations (Section 4.4), Figures 4.7 and 4.8 indicate the probable level of interaction occurring for principal Tongan species. Interactions between predominantly deep water seamount, and predominantly shallow water bank species will be low. However, deep water species exists around the banks and so some interaction will occur. This has not been fully reflected in Figures 4.7 and 4.8, except for the serranid, species 8. Based on the ecology of these fish, whilst competition for food may occur, its impact is expected to be low due to the generalist feeding behaviour exhibited. Similarly, predation by other snappers and groupers is expected to be low, and probably only occurs on juveniles of other species. Although evidence for different nursery habitats is weak it is assumed that predation on juveniles of the same species will be low. *E. septemfasciatus* is an exception, being a very large grouper assumed to prey also on adults of other target species, and there is evidence to support this (Section 4.4). Otherwise, the most significant interaction relates to competition for space. This will occur both within and between species.

Depth interactions:

	Pfil	Pfla	Ecor	Ecar	Emor	Esep	Lchr	Sp8	Lrub	Gjap
Pfil		+++	+	+	++	+	-	++	-	-
Pfla	+++		+	+	++	+	-	++	-	-
Ecor	+	+		+++	+	++	-	+	-	-
Ecar	+	+	++		+	++	-	+	-	-
Emor	+	+	+	+		+	-	++	-	-
Esep	+	+	+	+	++		-	+	-	-
Lchr	-	-	-	-	-	-		+'	++	+++
Sp8	+	+	+	+	++	+	+'		+'	+'
Lrub	-	-	-	-	-	-	+++	+'		++
Gjap	-	-	-	-	-	-	+++	+'	+++	

Competition for food

Figure 4.7 : The likely level of interaction between principal species in the catch from Tonga's banks and seamount fishery in relation to depth and competition for food (- indicates no competition, +; ++; +++ indicates low, moderate and high levels of interaction respectively, +' relates only to Sp8 which is regarded as a banks-caught species, although it clustered with the deep water species)

PREDATORS

	Zooplanktivores		Piscivores				Invertebrate feeders			
PREY	Pfil	Pfla	Ecor	Ecar	Emor	Esep	Sp8	Lchr	Lrub	Gjap
Pfil	J-	J	J	J	J+	A	J			
Pfla	J	J-	J	J	J+	A	J			
Ecor	J	J	J-	J	J	A+	J-			
Ecar	J	J	J+	J-	J	A+	J-			
Emor	J-	J-	J	J-	J-	J	J-			
Esep	J-	J-	J-	J-	J-	J-	J-			
Sp8	J-	J-	J-	J-	J-	A	J-	J-	J-	
Lchr							J+	J-	J	
Lrub							J+	J	J-	
Gjap							J+	J	J-	

Figure 4.8 : Proposed predator prey relationships between principal Tongan species (J indicates 'Very low level of predation on juveniles a likely possibility'; A indicates 'Very low level of predation on juveniles and/or adults a likely possibility'; J / A with : - indicates 'Predation possible (habitat / depths overlap) but concluded unlikely'; + indicates 'A low level of predation probably occurs')

4.5.2 Seychelles and Mauritius

At the Indian Ocean study sites of Seychelles and Chagos Archipelago, a large number of species are represented in the catch and insufficient information on their trophic relations is available to state with any certainty what species prey upon the others. Consequently, family groups of bank caught species have been pooled, except for *L. gibbus* (which was caught in shallow water and for which fish are rare in the diet), and the intermediate depth species. Figures 4.9 and 4.10 indicate potential interactions for space and food, and predator-prey relationships respectively. For the other banks fished by Mauritian vessels, the catch is predominantly mono-specific (80% *L. mahsena*) and inter specific competition (density dependance) will be the most important feature.

Depth / habitat competition

	Shallow / banks				Intermediate / drop off		
	Lgib	Lutx	Letx	Serx	Pfil	Arut	Emor
Lgib		++	+++	+++	+	+	+
Lutx	++	+++	+++	+++	+	+	+
Letx	+++	++	+++	+++	+	+	+
Serx	+	++	+	+++	+	+	+
Pfil	-	+	-	+		+++	+++
Arut	-	+	-	+	+		+++
Emor	-	+	-	++	++	+	

Food competition

Figure 4.9 : The likely level of space and food interaction between species, and species in the family groups lutjanidae (Lutx), lethrinidae (Letx) and serranidae (Serx) for fish caught predominantly on the shallow banks and those taken at intermediate depths on the drop off in Seychelles and by Mauritian vessels in Chagos. The same notation as Fig. 4.7 applies.

PREDATOR

	Shallow / banks				Intermediate / drop off		
PREY	Lgib	Lutx	Letx	Serx	Pfil	Arut	Emor
Lgib	J-	J	J-	J	J-	J-	J-
Lutx	J-	J	J-	J+	J	J-	J+
Letx	J-	J+	J-	J+	J	J-	J+
Serx	J-	J	J-	J	J	J-	J
Pfil					J-	J-	J+
Arut					J+	J-	J+
Emor					J	J-	J-

Figure 4.10 : Proposed predator - prey responses for Indian ocean study species. The notation of Fig. 4.8 applies.

4.6 Reproduction

The lutjanids, serranids and lethrinids have different reproductive strategies, and within families, differences occur between species. The reproductive biology of lutjanids has been reviewed by Grimes (1987), and that of serranids by Shapiro (1987). Lutjanids are gonochoristic (sex changes do not occur) whilst serranids are protogynous hermaphrodites (sex reversal is common). There are many examples in the literature (see Shapiro, 1987; *E. chlorostigma* : Ghorab *et al*. 1986; Moussac, 1986; *Plectropomus leopardus* : Ferreira, 1995). Protogynous hermaphroditism has also been demonstrated in several species of lethrinids (*L. elongatus*, Wassef and Bawazeer, 1992; FISHBASE, 1995) whilst others are gonochoristic. Sex reversal may occur at different ages, and the stimulus for change may differ between species. Generally, for protogynous species there are significantly more females in the population than males. Lutjanids may have a skewed population sex ratio or sex ratio at size, but this is a result of differential growth and mortality rates between the sexes.

For lutjanids, sexual maturity occurs at a slightly smaller size for males than females. Populations associated with islands mature at about 51% of maximum length compared to 43% for continental species and populations. Deep water species mature at a greater relative size (49% of maximum length) than shallow water species (43%, Grimes, 1987). Lutjanids are highly fecund (fecundity is positively correlated with size) and island populations tend to be batch spawners and reproduce all year round with pulses corresponding to local environmental conditions. The serranidae usually only spawn for about half the year (Shapiro, 1987) but within this time restricted spawning periods of about two weeks occur, frequently associated with aggregations. Lethrinids are reported to have a prolonged spawning period : e.g. *L. nebulosus* a single spawning period, September to February on the Australian NW shelf (Kuo and Sein-Sen; 1990); March to June in Okinawa (Ebisawa, 1990); *L. elongatus,* multiple spawning from May to August in the Red Sea (Wasef and Bawazeer, 1992).

Spawning events and settlement of larvae are frequently reported to occur in relation to full tides and around the new and full moons. Such strategies are thought to have developed to reduce predation pressure, but also to ensure that juveniles are returned to a suitable habitat for settlement. Exceptions have been reported for certain Lutjanids, including *L. sebae* and *L. malabaricus* which showed no apparent relationship between spawning activity and lunar cycle in Australia (McPherson et al, 1992). A number of spawning strategies and mating systems have developed. Amongst these, spawning aggregations have most commonly been reported for serranids (see Shapiro, 1987; Colin, 1992; Shapiro et al, 1993; Samoilys and Squire, 1994). For the Seychelles, spawning aggregations are known for *Epinephelus fuscoguttatus* at the main pass in Farquahar Island, and *Epinephelus caeruleopunctatus* at Victoria on Mahe (Nageon, pers. comm.) Lethrinids may also form large aggregations when spawning (see FISHBASE, 1994). They are less reported for lutjanids, but have been observed (eg Carter and Perrine, 1994). Fishing commonly occurs on spawning aggregations and the effects on the population structure are not yet fully understood (Shapiro, 1987; Shapiro et al, 1993).

4.6.1 Reproductive parameters for case study species

Biological information suitable for the assessment of reproductive parameters was available only for certain species from Seychelles (Tables 4.2 and 4.3), and principally for lutjanids. For the Tongan case study the length at which 50% of females reach sexual maturity (Lm) was assumed to be 50% of the maximum length. This was applied to all species, although Grimes' (1987) observation (Lm~51% Lmax for island species) related to lutjanids only. Langi and Langi (1987) used the asymptotic length (L_∞). No biological data was available for the Mauritian case study.

Available information from the literature relating to case study lethrinid, serranid (Table 4.4) and lutjanid (Table 4.5) species is presented for comparison and in order to define suitable parameter ranges for management simulations.

Genus	Species	Country	Sex	Age at 1st Maturity		tm	Length 1st Maturity		Lm	Type1	Spawning Period	Sex Ratio F:M	References
				From	To		From	To					
Lethrinus	miniatus	Australia	unsexed	2	3		31	35		SL			Carpenter and Allen (1989); Kailola et al (1993)
Lethrinus	miniatus	Tonga	female						38	50%L∞			Langi and Langi, 1987
Lethrinus	nebulosus	Egypt	female			5.9			45.6	TL			Sanders and Morgan (1989)
Lethrinus	nebulosus	Egypt	male			4.6			40	TL			Sanders and Morgan (1989)
Lethrinus	nebulosus	Kuwait	male			3							Carpenter and Allen (1989)
Lethrinus	nebulosus	Kuwait	female			4							Carpenter and Allen (1989)
Lethrinus	nebulosus	Kuwait	unsexed	4	4.5		32.5			TL			Lee and AlBaz (1989)
Epinephelus	chlorostigma	Red sea	female				28			TL		1:0.417	Ghorab et al. 1986
Epinephelus	chlorostigma	Seychelles	male				39			TL	May-Jul		Sanders et al (1988)
Epinephelus	chlorostigma	Seychelles	female				31			TL	May-Jul	1:0.417	Moussac, 1986
Epinephelus	chlorostigma	Seychelles	female				23	29		TL			Heemstra and Randall (1993)
Epinephelus	morhua	Tonga	female						37.1	50%LiL∞			Langi and Langi, 1987
Epinephelus	septemfasciatus	Tonga	female						86	50%LiL∞			Langi and Langi, 1987
Plectropomus	leopardus	Australia	male	3			30	54		SL			Heemstra and Randall (1993); Goeden (1978)
Plectropomus	leopardus	Australia	female	2	4		21	47		SL			Heemstra and Randall (1993); Goeden (1978)
Plectropomus	leopardus	Fiji	unsexed						30	FL			Lewis et al 1983

Table 4.4 : Reproductive parameters recorded in the literature for principal case study lethrinid and serranid fish species.

Genus	Species	Country	Sex	Age at 1st Maturity		tm	Length 1st Maturity		Lm	Type1	Spawning Period	Sex Ratio F:M	References
				From	To		From	To					
Aprion	virescens	East Africa	female				46.5			SL	Oct - Feb		Talbot, 1960
Aprion	virescens	Hawaii	female				42.9	50	44.9	FL			Everson et al (1989)
Aprion	virescens	Hawaii	female				42.9			FL	May - Oct Pk 6	1:1.05	Everson et al
Aprion	virescens	Seychelles	female				48.7		62-64	FL	Peak 9/11+3/4	1:1.8	Mees, 1992b
Etelis	carbunculus	Papua N Guin	unsexed						61				Lokani et al (1990)
Etelis	carbunculus	Tonga	female						57	50%L∞			Langi and Langi, 1987
Etelis	coruscans	Hawaii	female				52.2	82.4	66.3	FL			Everson et al (1989)
Etelis	coruscans	Tonga	female						49.6	50%LiL∞			Langi and Langi, 1987
Lutjanus	bohar						50	55		TL			Allen (1985)
Lutjanus	gibbus		unsexed						30	TL			Allen (1985)
Lutjanus	sanguineus		unsexed	6			50	60		TL			Allen (1985); Van der Elst (1981)
Lutjanus	sanguineus	Kuwait	unsexed	2.5	3				47.3	TL			Lee and AlBaz (1989)
Lutjanus	sebae	Australia	female				48.5		54.2	FL			McPherson et al (1992)
Lutjanus	sebae	East Africa	female				49			SL	Nov - Mar		Talbot, 1960
Lutjanus	sebae	Gulf Aden	female								March	1:1.04	Druzhinin and Filatova, 1981
Lutjanus	sebae	Seychelles	female							Trawl data		1:1.5	Tarbit in Moussac 1988
Lutjanus	sebae	Seychelles	female				34.6		61-63	FL	Peak 10/11+3/5	1:0.83	Mees, 1992b
Lutjanus	sebae	Seychelles	female				62			min at spawn	Dec-jaJanApr	1:0.66	Moussac, 1988
Pristipomoides	filamentosus		unsexed				35	50		TL			Allen (1985)
Pristipomoides	filamentosus	Papua N Guin	unsexed						34				Lokani et al (1990)
Pristipomoides	filamentosus	Seychelles	female				36.6		52	FL			Mees (1992b)
Pristipomoides	filamentosus	Seychelles	male				40			FL			Mees (1992b)
Pristipomoides	filamentosus	Andaman sea	female								March	1:0.76	Min et al., 1977
Pristipomoides	filamentosus	Guam	female									1:1.4	Kami, 1973
Pristipomoides	filamentosus	Hawaii	female								All yr pk 5-9	1:1.02	Ralston, 1981
Pristipomoides	filamentosus	Seychelles	female				33.2		51-53	FL, fem	All yr pk 10-4	1:1.07	Mees, 1992b; 1993
Pristipomoides	filamentosus	Seychelles	male				40-42			FL, male	All yr pk 10-4		Mees, 1992b; 1993
Pristipomoides	filamentosus	Tonga	female						38.6	50%L∞			Langi and Langi, 1987
Pristipomoides	flavipinnis	Tonga	female						28.8	50%LiL∞			Langi and Langi, 1987

Table 4.5 : Reproductive parameters recorded in the literature for principal lutjanid case study species.

4.7 Growth

Growth refers to the determination of body size as a function of age. The von Bertalanffy growth formula is the most commonly used model for expressing fish growth (see ?). A number of methods exist for age and growth determination : tagging, analysis of annual growth bands on hard parts such as otoliths and vertebrae, or daily bands on otoliths, and analysis of length frequency data, each with its own advantages and limitations.

Tagging of snapper and grouper has met with little success (Parker, 1990), often due to evisceration by deeper water species and high mortalities.

Otoliths and hard parts have been used principally for ageing temperate water species. In the tropics where strong seasonality is not evident, it is generally considered difficult to detect annuli on hard parts. However, this technique has been applied to annual banding in snappers, groupers and emperors from warmer waters, including important study species (Brothers and Mathews, 1987; Manooch & Drennon, 1987; Samuel et al., 1987; Seyama et al. 1991; Sadovy *et al.*, 1992; Rocha-Olivares & Gomez-Munoz, 1993; Hood, et al 1994; Johnson & Collins, 1994). This contrasts with other research (eg. Baillon and Morize, 1992; Milton *et al.*, 1995). Certainly knowledge of the life history of study species is required in order to interpret any rings on hard parts. Daily counting of rings has also been applied using Scanning Electron Microscopy (SEM). SEM has been recognised as a successful technique in ageing both larval and juvenile fish , and has potential for ageing older fish species (Ralston & Miyamoto, 1983; Radtke & Targett, 1984; Smith & Kostlan, 1991).

Methods allowing the conversion of length frequency data into age composition have been developed, and these are most frequently applied to tropical fish species in developing countries where resources for collection and reading of otoliths are limited. Length based methods have been reviewed by Gulland and Rosenberg (1992). They are generally considered to be unreliable for long lived slow growing species such as snappers and groupers (for Tonga, see Langi, 1990) and most useful for short lived fast growing species. Shepherd *et al.* (1987) defined four length frequency distribution types and discussed their applicability to growth and mortality estimation. The quality and characteristics of available length frequency greatly affect their applicability to length based methods. Simulations have been used to investigate this, for example the effects of truncation of the length frequency due to inadequate sampling of only part of a population (Shepherd *et al.* 1987).

Hampton and Majkowski (1987) examined the reliability of a length-based computer program (ELEFAN) for length-based stock assessment using a simulation study. The results of this analysis show that ELEFAN tends to overestimate $L\infty$ by 11-23%, and underestimates K by 16-36% in experiments where fishing is size selective. Where fishing is size-independent and growth variation is low these estimates are almost correct.

Manooch (1987) reviewed the growth of tropical snappers and groupers. The most commonly applied technique for aging has been the use of hard parts. These families of fish are characterised as long lived and slow growing with low rates of natural mortality. Growth coefficients (K) usually fall within the range 0.1-0.25 per year. An inverse relationship exists between the asymptotic length ($L\infty$) and K :

$$\log_{10} K = 1.098 - 0.658 \log_{10} L\infty$$

This relationship has been used in the absence of time series length frequency or other data to estimate K from an estimate of $L\infty$ derived by the method of Wetherall *et al*. (1987, e.g. Langi and Langi, 1987).

The range of growth parameters (K) reported for lethrinids in FISHBASE (1994) is 0.056-1.0. 60%, and 87% are in the ranges 0.1-0.3 and 0.1-0.5 respectively. Thus lethrinids are also relatively slow growing, although mostly faster than lutjanids or serranids.

4.7.1 Growth parameter estimates for case study species

Apart from data relating to otoliths for *P. filamentosus* from Seychelles which was inconclusive (see Technical Report, MRAG, 1996a), only length frequency data was available for Seychelles and Tongan study species for the assessment of growth. No data was available for the Mauritian case study species (Tables 4.2 and 4.3).

The von Bertalanffy growth formula for L_t, the predicted length at age t, takes the form :

$$L_t = L\infty\ (1\text{-}e\text{-}^{K(t\text{-}to)})$$

The von Bertalanffy growth parameters, L∞ (the asymptotic maximum length), K (the growth rate) and t_0 (the theoretical time at which length is zero) were estimated from length frequency data for each of the key species using LFDA (Tonga, MRAG, 1992b) and the upgraded LFDA4 (Seychelles, MRAG,1995b). This analytical tool allows non seasonal growth curves to be fitted to length frequency data using three methods : Shepherds Length Curve Analysis (Shepherd, 1987), a projection matrix method (Basson *et al*, 1988) and a version of the ELEFAN method (Pauly, 1987a). Seasonal growth may be fitted for the latter two methods using either one of two methods (LFDA4 only) :Hoenig and Choudry Hanumara (1982) to describe C (the relative amplitude of seasonal oscillations) and t_s (the phase of these oscillations, t_s+1/2 is referred to as the 'winter point'); Pauly *et al* (1992) to describe additionally a no growth period (NGT).

In order to estimate growth it is necessary to have one or more modes in the length frequency distribution which increase in length steadily with time (see Shepherd *et al*. 1987). For long lived slow growing species such as snappers, groupers and emperors, length classes of the older fish frequently overlap, making modal separation difficult. Sampling biases and inadequacies may also affect the length frequency distribution observed as discussed above. Nevertheless, in the absence of other means of assessing growth (eg. hard parts) length frequency data analysis may be applied to these species providing that analytical models are applied sensibly, and that the limitations of the outputs are understood.

Given the potential for sampling bias, Hilborn and Walters (1992) stress the need for representative sampling in the analysis of length frequency data and warn that often this is not achieved when commercial catches are sampled. Problems include gear selectivity, size related changes in fish distribution, and inadequate sub-sampling of catches. By contrast Gulland and Rosenberg (1992) suggest that only by sampling different locations and times will statistically adequate data be collected fully representing the population in question. In this study, disaggregated data was employed to estimate growth parameters and it was only aggregated where justified (see Technical Reports, MRAG 1994; 1996a).

The model was fitted to data for *P. filamentosus, E. coruscans, E. morhua* and tentatively to *E. septemfasciatus* from Tonga, whilst the data for *P. flavipinnis* and *E. carbunculus* were inadequate for growth assessment. Seychelles species studied were *P. filamentosus, L. sebae* and *A. virescens*. Data were inadequate for *E. chlorostigma*, and there was an insufficient time series for other Seychelles species. Generally in applying LFDA/LFDA4 it was found that a number of potential K-L∞ pairs would fit the data and it was necessary to choose the most appropriate fit from a combination of : details of the score function (a measure of the fit of model to data), the plotted fit of the growth curve to the data, and experience. The final estimates were therefore to some extent subjective, but nevertheless represent the best estimate for the data available. Spatial and temporal variation and inadequacies in data collection had the result that frequently insufficient data were available to permit growth assessment by location and year. Aggregation of the data was therefore necessary over space or time limiting the potential for spatial and temporal comparisons of growth rate. Where separate estimates of growth parameters could be derived by location and year, the confidence in the model outputs was not sufficient to permit such comparisons.

It may be seen that considerable uncertainty exits in relation to the confidence in growth parameters derived for study species from length based methods of assessment. For comparison, parameter estimates available in the literature derived for study species are reported (Tables 4.6-4.7). Considerable variation in growth parameter estimates occurs for the same species studied by different authors around the world. A number of factors may explain this including geographical variation, different levels of fishing pressure, sampling bias (eg, L_∞ for *A. virescens* from New Caledonia is reported as 66 cm, whilst in the Maldives and Seychelles, considerably larger fish have been observed, suggesting incomplete sampling of the population in New Caledonia), or uncertainties in parameter estimation due to the application of length based methods. Confidence intervals for parameter estimates are not reported and currently LFDA4 does not include this option.

Genus	Species	Country	K	L∞	to	Type	Method	References	
Gymnocranius	euanus	New Caledonia	0.240	35.1		SL		Munro 1983	Loubens 1980
Gymnocranius	euanus	New Caledonia	0.220	39.5		SL		Munro 1983	Loubens 1980
Gymnocranius	grandoculis	New Caledonia	0.230	46.4		SL		Dawson 1966	Loubens 1980
Lethrinus	mahsena	Yemen Dem	0.320	58.9		FL		Dalzell et al 1992	Aldonov&Druzhinin 1979
Lethrinus	microdon	Djibouti	0.200	82.0		TL		Carpenter and Allen (1989)	
Lethrinus	miniatus	Australia	0.170	58.5	-2.26	SL		Walker 1975	
Lethrinus	miniatus	New Caledonia	0.270	45.7		SL		Loubens 1980	
Lethrinus	miniatus	New Caledonia	0.260	48.9		SL		Loubens 1980	
Lethrinus	miniatus	Norfolk Is	0.270	47.3				Dalzell et al 1992	
Lethrinus	miniatus	Norfolk Is	0.100	70.0	-2.36	FL		Church 1989	
Lethrinus	miniatus	Norfolk Is	0.056	83.0	-4.60	FL		Church 1989	
Lethrinus	miniatus	PNG	0.250	75.0		FL		Munro and Williams 1985	
Lethrinus	miniatus	Yemen Dem	0.230	58.9				Dalzell et al 1992	Aldonov&Druzhinin 1979
Lethrinus	nebulosus	Australia	0.390	48.0		SL	Non-linear regression	McPherson et al 1985	
Lethrinus	nebulosus	Australia	0.430	32.6		NA		Erzini 1991	Morales Nin 1989
Lethrinus	nebulosus	Australia	0.106	61.1	-0.88	FL		Kuo and Lee 1986	
Lethrinus	nebulosus	Australia	0.127	52.7	-1.16	FL		Kuo and Lee 1986	
Lethrinus	nebulosus	Egypt	0.110	86.0		TL		Sanders and Morgan (1989)	Sanders et al 1984
Lethrinus	nebulosus	Fiji	0.230	80.0		FL		Carpenter and Allen (1989)	
Lethrinus	nebulosus	Fiji	0.340	61.4		FL	ELEFAN I	Sharma 1990	
Lethrinus	nebulosus	Fiji	0.360	61.5		FL	ELEFAN I	Sharma 1990	
Lethrinus	nebulosus	Fiji	0.160	64.6		SL	other method(s)	Dalzell et al 1992	
Lethrinus	nebulosus	Fiji	0.170	62.3		SL	other method(s)	Dalzell et al 1992	
Lethrinus	nebulosus	Fiji	0.210	50.2		SL	other method(s)	Dalzell et al 1992	
Lethrinus	nebulosus	Kuwait	0.190	62.7		TL		Baddar 1987	
Lethrinus	nebulosus	Kuwait	0.190	62.7		TL		Carpenter and Allen (1989)	
Lethrinus	nebulosus	New Caledonia	0.220	50.9		SL		Loubens 1980	
Lethrinus	nebulosus	New Caledonia	0.210	54.3		SL		Loubens 1980	
Lethrinus	nebulosus	Papua N Guin	0.410	54.7		FL	other method(s)	Sharma 1990	
Lethrinus	nebulosus	Papua N Guin	0.310	55.8		FL	ELEFAN I	Sharma 1990	
Lethrinus	nebulosus	Yemen Rep	0.101	85.9		FL	Ford/Walford plot	Aldonov&Druzhinin 1979	
Lethrinus	nebulosus	Yemen Rep	0.210	71.6		FL		Aldonov&Druzhinin 1979	
Lethrinus	nebulosus	Yemen Rep	0.090	99.9		FL		Carpenter and Allen (1989)	
Lethrinus	nebulosus	Yemen Rep	0.090	87.0	0.54	FL		Sanders and Morgan (1989)	Edwards et al 1985
Lethrinus	rubrioperculatus	N Marianas	0.220	30.8	-0.40	FL		Ralston and Williams 1988b	Ralston and Williams 1988
Lethrinus	variegatus	New Caledonia	0.430	30.3		SL		Loubens 1980	

Table 4.6 : Growth parameter estimates for principal lethrinid case study species

Genus	Species	Country	K	L∞	to	Type	Method	References	
Epinephelus	chlorostigma	Kuwait	0.195	64.8				Mathews and Samuel, 1987	
Epinephelus	chlorostigma	Seychelles	0.179	68.4			ELEFAN TL	Sanders et al., 1988	
Epinephelus	chlorostigma	Seychelles	0.175	64.5			ELEFAN TL	Mees, 1992b	
Epinephelus	chlorostigma	Seychelles	0.190	62.7		TL		Sanders et al 1988	
Epinephelus	chlorostigma	Seychelles	0.170	66.9		TL		Sanders et al 1988	
Epinephelus	morhua	Tonga	0.160	74.2				Langi and Langi 1987	
Epinephelus	morhua	Tonga		92.0				Latu and Tulua 1991	
Epinephelus	multinotatus	Kuwait	0.270	72.6	-1.21	TL	Other methods	Mathews and Samuel 1987	
Epinephelus	septemfasciatus	Tonga		198.0				Latu and Tulua 1991	
Plectropomus	leopardus	Australia	0.430	45.0		SL	Non-linear regression	McPherson et al 1985	
Plectropomus	leopardus	Australia	0.250	64.7		SL	ELEFAN I	Pauly and Ingles 1981	Goeden 1978
Plectropomus	leopardus	Australia	0.350	52.2	-0.77	FL	Non-linear regression	Ferreira and Russ 1993	
Plectropomus	leopardus	New Caledonia	0.160	50.0		SL		Munro 1983	Loubens 1980
Plectropomus	maculatus	Australia	0.210	60.0	-0.94	SL	v.Bertalanffy/Beverton plot	Ferreira and Russ 1992	
Variola	louti	PNG	0.180	64.0		FL	other methods	Munro and Williams 1985	

Table 4.7 : Growth parameter estimates for principal serranid case study species

Genus	Species	Country	K	L∞	to	Type	Method	References	
Aphareus	rutilans	N Marianas	0.163	123.0	-0.36	FL	Non-linear regression	Ralston and Williams 1988	
Aprion	virescens	Maldives	0.348	78.0			ELEFAN	Van der Knapp et al., 1988	
Aprion	virescens	New Caledonia	0.310	66.0		SL		Manooch 1987	Loubens 1980
Aprion	virescens	New Caledonia	0.310	65.6				Loubens, 1980	
Aprion	virescens	Seychelles	0.290	95.0			ELEFAN males	Mees, 1992b	
Aprion	virescens	Seychelles	0.260	104.0			ELEFAN all	Mees, 1992b	
Aprion	virescens	Seychelles	0.140	108.0			ELEFAN females	Mees, 1992b	
Etelis	carbunculus	Fr Polynesia	0.126	68.0	2.35		otoliths pd	Smith and Kostlan 1991	
Etelis	carbunculus	Hawaii	0.360	63.9	-0.60	FL	otoliths pd	Uchida 1986	
Etelis	carbunculus	Hawaii	0.163	71.8	3.32	FL	v.Bertalanffy/Beverton plot	Smith and Kostlan 1991	
Etelis	carbunculus	N Marianas	0.347	40.0	-1.06	FL	Non-linear regression	Ralston and Williams 1988	
Etelis	carbunculus	N Marianas	0.289	54.0	-0.23	FL	v.Bertalanffy/Beverton plot	Smith and Kostlan 1991	
Etelis	carbunculus	Polynesia Fr	0.126	68.0	2.35	FL	v.Bertalanffy/Beverton plot	Smith and Kostlan 1991	
Etelis	carbunculus	Tonga		120.0			for L∞	Latu and Tulua 1991	
Etelis	carbunculus	Vanuatu	0.129	127.0	1.41	FL	v.Bertalanffy/Beverton plot	Smith and Kostlan 1991	
Etelis	carbunculus	Vanuatu	0.070	94.0		FL		Brouard and Grandperrin	(1984)
Etelis	coruscans	N Marianas	0.123	109.0	-1.19	FL	Non-linear regression	Ralston and Williams 1988	
Etelis	coruscans	Tonga	0.130	99.3			Wetherall	Langi and Langi 1987	
Etelis	coruscans	Tonga		116.0			Wetherall	Latu and Tulua 1991	
Etelis	coruscans	Vanuatu	0.130	82.0		FL		Brouard and Grandperrin	(1984)
Lutjanus	bohar	Kenya	0.270	66.0		TL		Talbot 1957	
Lutjanus	bohar	New Caledonia	0.110	52.0		SL		Loubens 1980	
Lutjanus	bohar	Papua N Guin	0.270	81.7		FL	ELEFAN I	Wright et al 1986	
Lutjanus	bohar	Seychelles	0.330	66.0		TL	other method(s)	Munro 1983	Wheeler and Ommaney 1953
Lutjanus	gibbus	Palau	0.400	39.8		FL	ELEFAN I	Kitalong and Dalzell 1994	
Lutjanus	gibbus	PNG	0.310	44.2		FL		Munro and Williams 1985	
Lutjanus	sanguineus	Djibouti	0.236	89.0				Sanders and Morgan 1989	Kedidi and Bohlel 1985
Lutjanus	sanguineus	Kuwait	0.358	68.9	-0.76	TL	other method(s)	Mathews and Samuel 1991	Mathews and Samuel 1985
Lutjanus	sebae	Australia GBR	0.180	88.7	-0.32	FL	other method(s)	McPherson and Squire 1990	
Lutjanus	sebae	Australia GBR	0.210	90.0		TL		Lablache and Carrara 1988	McPherson et al 1985
Lutjanus	sebae	Australia GBR	0.210	72.0		SL	Non-linear regression	McPherson et al 1985	
Lutjanus	sebae	Australia GBR	0.130	81.7	-1.09	FL		Yeh et al 1986	
Lutjanus	sebae	Australia GBR	0.140	91.0		SL	Non-linear regression	McPherson et al 1985	
Lutjanus	sebae	Australia GBR	0.150	102.5	-0.32	FL	other method(s)	McPherson and Squire 1990	
Lutjanus	sebae	Australia NW	0.130	81.7	-1.09		vertebra pa	Yeh et al 1986	
Lutjanus	sebae	East Africa	0.157	85.1				Talbot, 1960	
Lutjanus	sebae	Seychelles	0.307	84.0			ELEFAN all	Mees, 1992b	
Lutjanus	sebae	Seychelles	0.380	95.1			ELEFAN males	Mees, 1992b	
Lutjanus	sebae	Seychelles	0.270	90.0			ELEFAN fem.	Mees, 1992b	
Lutjanus	sebae	Seychelles	0.220	98.0				Bach,1991	
Lutjanus	sebae	Seychelles	0.230	96.0		TL	Gulland & Holt plot	Lablache and Carrara 1988	
Lutjanus	sebae	Yemen Dem	0.160	85.1	-1.02	FL	scales pa	Druzhinin and Filatova 1980	
Pristipomoides	filamentosus	Hawaii	0.220					Ralston and Williams, 1983	
Pristipomoides	filamentosus	Hawaii	0.310	97.1	0.02	FL		Uchida 1986	
Pristipomoides	filamentosus	Hawaii	0.164	80.5	-0.84	FL	Non-linear regression	Ralston 1980	
Pristipomoides	filamentosus	Hawaii	0.146	78.0	-1.67	FL		Uchida 1986	Ralston and Miyamoto 1983
Pristipomoides	filamentosus	Hawaii	0.146	78.0	-1.67		otoliths pd	Uchida 1986	Ralston and Miyamoto 1983
Pristipomoides	filamentosus	Hawaii FFshl	0.164	80.5	-0.84		otoliths pd	Ralston 1980	
Pristipomoides	filamentosus	Hawaii NW	0.310	97.1	0.02		otoliths pd	Uchida 1986	
Pristipomoides	filamentosus	N Marianas	0.289	58.0	-0.54	FL	Non-linear regression	Ralston and Williams 1988	
Pristipomoides	filamentosus	N Marianas	0.289	58.0	-0.54		otoliths pa	Ralston and Williams 1988	
Pristipomoides	filamentosus	Seychelles	0.300	85.8		FL	ELEFAN males	Mees, 1993	
Pristipomoides	filamentosus	Seychelles	0.288	81.7			ELEFAN all	Mees, 1993	
Pristipomoides	filamentosus	Seychelles	0.275	77.6		FL	ELEFAN fem.	Mees, 1993	
Pristipomoides	filamentosus	Tonga	0.160	77.2			K = func of L∞	Langi and Langi 1987	
Pristipomoides	filamentosus	Tonga		80.0				Latu and Tulua 1991	
Pristipomoides	filamentosus	Vanuatu	0.290	60.0		FL		Brouard and Grandperrin	(1984)
Pristipomoides	flavipinnis	N Marianas	0.268	49.0	-1.01	FL	Non-linear regression	Ralston and Williams 1988	
Pristipomoides	flavipinnis	Tonga	0.190	57.5				Langi and Langi 1987	
Pristipomoides	flavipinnis	Tonga		57.0				Latu and Tulua 1991	
Pristipomoides	flavipinnis	Vanuatu	0.360	58.0		FL	otoliths pa	Brouard and Grandperrin	(1984)

Table 4.8 : Growth parameter estimates for principal lutjanid case study species

4.8 Mortality

The instantaneous total mortality (Z) rate of a fish population is described by the sum of natural mortality (M), and fishing mortality (F). It explains the loss of individuals from the population due to natural causes and fishing. In previously unfished populations, an estimate of the total mortality will be equivalent to the natural mortality. Gulland (1971) suggested that fishing effort / mortality is optimal when F=M. For snapper and grouper populations, Ralston (1987) reviewed mortality estimates, and compared the ratio of F/M for serranids (0.27-3.00) and lutjanids (1.25-3.30). Natural mortality is attributed to predation, parasitism, cold water shock and red tide poisoning.

Total mortality is usually measured through mark-recapture techniques (Ricker, 1975), but these have not successfully been applied to snappers and groupers. The survival of a cohort of fish is measured (Gulland, 1955), or, for a single sample the change in cohort sizes between modes in age-length data is measured (Gulland 1955). Length based methods have been developed for this cohort analysis (length converted catch curves, Jones 1984; see Sparre *et. al*, 1989). These require estimates of the growth parameters K and L_∞ as inputs. In addition to catch curve analysis a method developed by Powell (1979) and extended by Wetherall *et al*. (1987) may be used to determine growth and mortality parameters from length frequency data. Growth parameter estimates are not required inputs. The method provides estimates of the ratio Z/K and L_∞. Values of Z/K may be compared directly where no estimate of K exists in order to evaluate relative mortalities by location for example (eg. see Munro 1983). To separate Z it is possible to multiply by K, but strictly growth parameter estimates should be derived for each length frequency data set from which Z/K is estimated.

Natural mortality may be determined directly for unfished locations (M=Z). Pauly (1980; 1984) derived an empirical model for estimating natural mortality from mean water temperature, and the growth parameters K and L_∞. Trenkel (1993) used an enlarged data set for fish species occurring at water temperatures above 5°C to modify Pauly's original model. FISAT, an analytical tool for length based data (Gayanilo et al, 1994), also uses an extended data set to improve the estimate of M. This model, and that of Trenkel lead to slightly lower estimates of M than the original Pauly (1980) estimator. For snappers and groupers Ralston (1987) derived two models for the estimation of natural mortality from growth parameters: one based on linear regression and one based on functional regression. He concluded that for these fish the Pauly (1980) method underestimates M for fast growing fish, and overestimates it for slow growing fish.

Catch curves to determine Z, Z/K ratio estimates, and Pauly's empirical formula for M are the most frequently employed techniques to determine mortality in snappers and groupers (Ralston 1987). Length based methods were also employed in this study (Tables 4.2 and 4.3; MRAG 1994; 1996a). Few estimates of M exist for snappers and groupers in the literature (Table 4.9).

GENUS	SPECIES	COUNTRY	M	K	L∞	TYPE	References	
Lethrinus	nebulosus	Fiji	0.51	0.230	80	FL	Carpenter and Allen (1989)	
Lethrinus	nebulosus	Kuwait	0.36	0.190	62.7	TL	Carpenter and Allen (1989)	
Plectropomus	leopardus	Australia	0.46	0.250	64.7	SL	Pauly and Ingles 1981	
Lutjanus	sebae	Seychelles	0.48				Lablache and Carrara, 1984	
Aprion	virescens	Maldives	0.49				Van der Knap et al, 1988	
Etelis	carbunculus	Vanuatu	0.08	0.070	94	FL	Brouard and Grandperrin 1984	Ralston 1987
Etelis	coruscans	N Marianas	0.36	0.123	109	FL	Ralston and Williams 1988	
Etelis	coruscans	Vanuatu	0.12	0.130	82	FL	Brouard and Grandperrin 1984	Ralston 1987
Pristipomoides	filamentosus	Hawaii	0.25	0.146	78	FL	Uchida 1986	Ralston 1987
Pristipomoides	filamentosus	N Marianas	0.52	0.289	58	FL	Ralston and Williams 1988	
Pristipomoides	filamentosus	Vanuatu	0.53	0.290	60	FL	Brouard and Grandperrin 1984	Ralston 1987
Pristipomoides	flavipinnis	Vanuatu	0.83	0.360	58	FL	Brouard and Grandperrin 1984	Ralston 1987

Table 4.9 : Natural mortality estimates reported in the literature for case study species

4.9 Length-weight relationships

The weight of a fish increases approximately in proportion to the length cubed and the relationship is expressed :

$$W = a.L^b$$

where W is the weight (in kg in this report) and L is the fork length (in cm). The parameters‘a’ and ‘b’ are constants derived by regression of log transformed data.

Length and weight data were only available for certain species from Seychelles (Tables 4.2 and 4.3). Details for Mauritian and Tongan species were obtained from the literature.

4.10 Length at capture

In order to determine the length at first capture, gear specific selection parameters were derived from length frequency data using probability of capture at length information derived through catch curve analysis (Gayanillo *et al*., 1994). A moving average was used:

$$P_{L,i(new)} = (P_{L,i-1} + P_{L,i(old)} + P_{L,i+1}) / 3$$

where P_L is the probability of capture for length L, a smoothed series of probabilities is produced from which Lc_{25} (probability of 25 % of all fish at that length being caught), Lc_{50} and Lc_{75} are estimated through linear interpolation. Lc, the first fully exploited length class was taken as the cut off point of the catch curve analysis. Growth parameters are required inputs for catch curve analysis.

To estimate length at capture, data was stratified by gear type (Seychelles only), but otherwise, unless specified, was aggregated over all locations. Tables 4.2 and 4.3 indicate length at capture for handline caught species only (see MRAG, 1996a for other gears). That for *E. carbunculus*, which was found to have two predominant modes in its length frequency distribution from Tonga, relates to the larger mode (see MRAG, 1994).

4.11 Implications of uncertainty in growth parameter estimates

Growth parameters estimates are key inputs to other models used in fisheries assessment. These include those used to establish total mortality or length at capture for example (eg, length converted catch curve, Jones, 1984); natural mortality if Pauly's (1980; 1984) empirical formula is used; the yield per recruit analysis of Beverton and Holt (1957); MIDAS (Rossouw, 1994). As a result, uncertainty in the growth parameter estimates is translated into uncertainty in other parameter estimates, and hence the need to perform sensitivity analyses for these parameters in order to establish expected ranges for them. In models such as MIDAS which requires a large number of inter-dependent parameter estimates as inputs it is appropriate for other parameters dependent on K and L∞ to vary when sensitivity of model outputs to changes in growth parameters are assessed, rather than have them as fixed values. MIDAS has this facility.

For population demographic parameters derived for case study species, the sensitivity of outputs from catch curve analysis (Z, Fig. 4.11; Lc_{50} Fig. 4.12) and from Pauly's empirical estimator for M (Fig. 4.13) were investigated using the example of *P. filamentosus* from the Seychelles. Sensitivity of outputs from MIDAS to variation in growth parameters are investigated in Section 5.2.3.

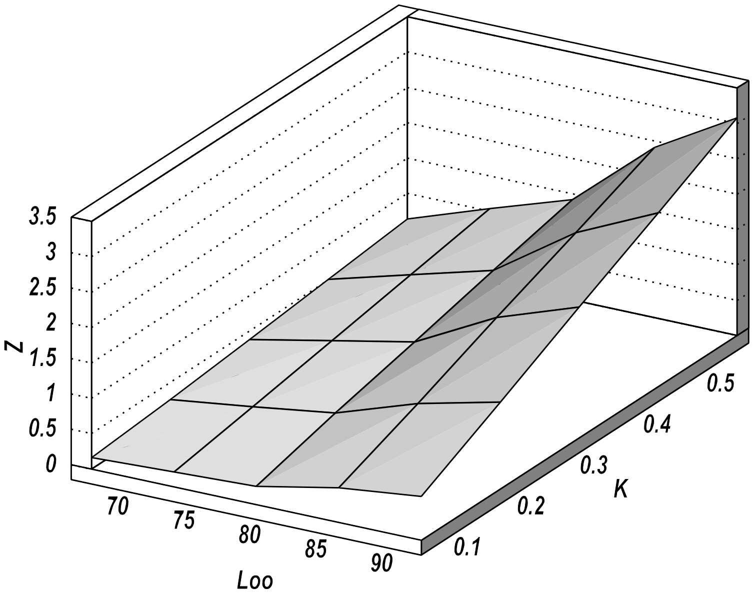

Figure 4.11 : Sensitivity of the estimate of total mortality, Z, to variation in growth parameter inputs to catch curve analysis for length frequency data for *P. filamentosus* from the Mahe Plateau (Lmax = 78cm)

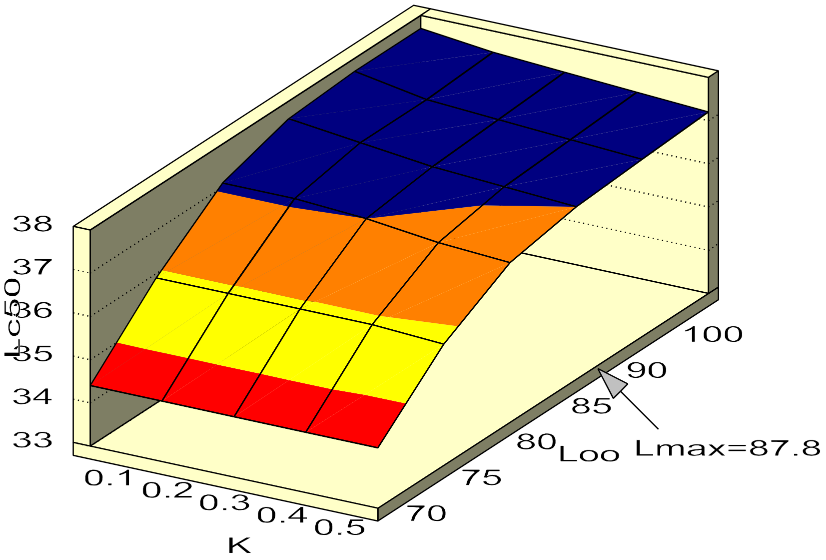

Figure 4.12 : Sensitivity of the estimate of Lc_{50} to variation in growth parameter estimates applied to probability of capture in catch curve analysis for *P. filamentosus* caught by handlines from all locations (Lmax=87.8 cm)

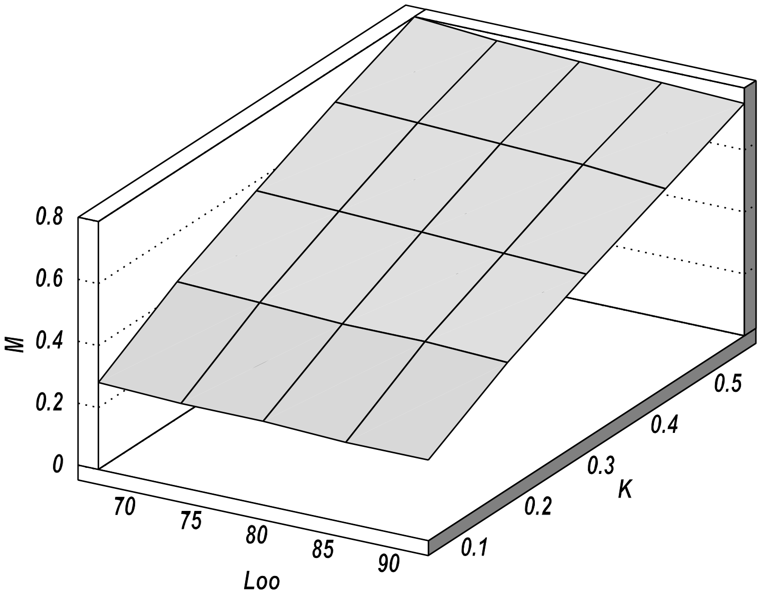

Figure 4.13 : Sensitivity of the empirical estimate (Pauly, 1984) of natural mortality to variation in growth parameter inputs (T^oC=22)

These sensitivity analyses indicated that :

- The estimate of Z derived from catch curve analysis is sensitive to variation in both K and L∞. At any given length, Z increases linearly with K. At any given value of K, Z increases with increasing L∞, and there is an inflexion point about Lmax after which the rate of change increases. A considerable range of estimates for Z is suggested. However, in practice, as L∞ increases, the estimate of K tends to decrease (K and L∞ are negatively correlated) so Z is unlikely to vary to the extent suggested (Fig. 4.11). Natural mortality is sensitive to variation in values of K but not L∞ (Fig 4.13). Fishing mortality follows the pattern for Z, being the remainder of total mortality less natural mortality. MRAG (1996a) showed that for *P. filamentosus*, where seasonal parameters were included in the growth function, a higher estimate of Z was derived from catch curve analysis than when they were not included. Where Pauly's estimator for M is used, Ralston (1987) indicated that M was overestimated for slow growing fish. As a result, fishing mortality may tend to be underestimated

- The estimate of Lc_{50} was little affected by changes in K. For values of L∞ below Lmax, Lc_{50} tended to be underestimated, but Lc_{50} was insensitive to changes in L∞ when this parameter was around, or greater than Lmax (Fig. 4.12). This suggests that providing the value of L∞ used is close to Lmax, we may be confident in the value of Lc_{50} derived, providing that the sample size was sufficiently large.

These analyses indicate that for the derivation of population demographic parameters using length based methods of assessment such as the catch-curve, growth parameter estimates where L∞ approximates Lmax are more likely to provide reasonable estimates. This is particularly important where growth parameter estimates may not be available for study species, and it is necessary to apply those from the literature. Variation in K is less important, except for the estimation of natural mortality, suggesting that alternative means of assessment should be applied where possible. Furthermore, the sensitivity to variations in M should be investigated where fisheries models employ both K and M.

In addition to the above parameters, in Tonga, for example, length at maturity, Lm was derived from 0.5(L∞) (Langi and Langi, 1987). Where L∞ is close to Lmax this relates more closely to Grimes' (1987) original observation. The estimate of length at maturity will reflect inaccuracies in estimation of growth parameters and indicates that where possible, independent assessment of this parameter is most appropriate.

Growth might be expected to vary with fishing pressure (see Chapter 5). However it was found that, for study species growth could not reliably be estimated from the data available using length based models and it was not possible to detect the expected changes in these parameters with fishing pressure. Estimates of growth parameters (K, L∞) are key inputs to MIDAS, and may be allowed to vary with time in order to simulate changes resulting from, for example, fishing. However, if these changes cannot be accurately detected, it raises the question of to what extent do changes in growth rates in fished populations affect our ability to manage fish resources. This is beyond the scope of the current project.

5 EFFECTS OF FISHING

5.1 Current knowledge of the effects of fishing

5.1.1 Background

A comprehensive review of the effects of fishing on coral reef fishes is provided by Russ (1991). Population (single species) and community (multispecies) effects can be identified which act directly on the fish resources, or indirectly through mechanisms such as habitat modification (e.g. Sainsbury *et al.*, in press). Multispecies responses are the focus of the present study, but single species effects also need to be understood in explaining effects at the community level. Indirect effects of fishing are not important for case study fisheries where the predominant fishing method is by hook and line, a method which has minimal impact on the habitat. Technical interactions and gear selectivity (ie catchability) have an important influence on species composition of multispecies resources. Those species with higher catchabilities to a particular gear will be subject to greater fishing mortality and will be removed first leading to a greater proportion of other species in the catch (but not necessarily a greater abundance of other species). This is an effect of fishing on fish populations related to the choice of gear. However, care must also be taken not to interpret a change in target species or technological change as reflecting changes in fish abundance.

For multispecies fisheries 'prey release' is a common theme in the literature (see 5.1.3). Grigg *et al*. (1984) and Munro and Williams (1985) advocated fishing of predators in order to enhance catches of prey. However, frequently it is the predators which are the target being the most highly valued, as is the case for line fisheries for banks and deep slope reef fish. Any management strategy needs to be aimed at sustainable exploitation of these resources. In reviewing the trophic relations of snappers and groupers Parrish (1987) proposed that changes in abundance of prey items would have little impact on the abundance of fishery target species due to their generalist feeding nature. He suggested that major reductions in the populations of some snappers and groupers due to fishing could be expected to reduce predation pressure and feeding competition resulting in increased populations and potential catches of other carnivorous demersal fish, including other snappers and groupers. Additionally he postulated increases in the abundance of non target prey species.

Hooks and lines represent a selective method of fishing and the 'prey' species of the lutjanids, serranids and lethrinids sought are not captured or targeted. The present study therefore is confined to examining interactions between target species (including predator-prey responses between them or between age classes of the same species) and any effects on other prey species will not be detected. For one of the case study locations, Seychelles, the inshore reefs are exploited by small boats using a variety of methods including traps, nets and hook and line. In a fishery independent survey, Jennings *et al*. (1995) indicated that fishing depleted the top predators (lutjanids, serranids and lethrinids) but there was no evidence for prey release or an increase in abundance of fish at other trophic levels related to fishing following their removal. It may be assumed that this observation will also apply to the offshore banks of the present study, and suggests that the lack of information on non-target species is not important.

5.1.2 Single species effects of fishing

The effects of fishing on a single species may be summarised as a reduction of density and biomass, reflected as catch rate changes, an increase in total mortality and growth rate, a reduction in the mean length and modification of the size and social structure of the population (eg sex ratio at length and in total), and reduction of the reproductive capacity of the resource and its recruitment.

By removing fish from an area, fishing results in a reduction of the density (per unit area) and biomass of individual species. Biomass and density differences attributed to fishing are well reported, particularly for large predatory fish such as lutjanids, lethrinids and serranids (eg. Bohnsak, 1982; Munro, 1983; see Russ, 1991), and dramatic reductions in biomass can occur over a short period of time with hook and line fisheries. For example, reporting fishing at Boult Reef on the Great Barrier Reef, Beinssen (1988) showed that 25% of *P. leopardus* were removed in 14 days of fishing; Polovina (1986) indicated a 48% reduction in the biomass of deep water lutjanids following 13 days fishing on a seamount in the Marianas; Mees (1991; 1992c; in press) indicated depletion following intensive fishing on isolated banks in Seychelles over a few days.

If the catchability of a fishing gear remains constant, the catch rate, or catch per unit (of fishing) effort (CPUE), is an index of abundance of the resource. CPUE is therefore used to infer changes in biomass of fished resources. This index is the basis of fisheries production models which determine the initial (unfished) biomass (B_0) and sustainable yield. CPUE declines with fishing and is expected to decline linearly with the abundance of fish. However, over time as a fishery develops experience and improvements in fishing gear, for example may mean that the relative level of effort changes, and effort and catchability need to be scaled or standardised. Furthermore, a non-linear relationship between catch rates and abundance is predicted (Beddington, 1984; Cooke and Beddington, 1984) with implications for the estimation of abundance. Thus whilst the precise nature of the relationship between catch rate and abundance is uncertain, a measurable effect of fishing is catch rate change.

Catch rates are normally reported for a time series of data from a single location subject to fishing pressure. Catch rate changes may be observed in the short term (days, e.g. Polovina, 1986; Mees, 1991; 1992c) or over several years. Spatial comparisons of CPUE have also been made for a number of locations subject to different levels of fishing pressure (eg. Gulland, 1979; Munro, 1983; Koslow *et al.* 1994). Spatial comparisons introduce another dimension : not only must catchability remain constant, in order to make inferences about abundance utilising spatial data, the ecological and productive characteristics of each location must be similar. Koslow *et al.* (1994) recorded significant differences in catch rates by location, related to fishing pressure, including declines in lutjanids and serranids. However, examples also exist of spatial comparisons where little difference in catch rate was observed despite different levels of fishing effort (see Russ, 1991).

For a fish stock, total mortality increases with fishing effort (by definition, $Z=F+M$, see 4.8, and $F=qf$ where q is catchability and f is effort). However, mortality rates, and in particular, independent assessments of natural mortality, are difficult to obtain. Little information exists to illustrate the relationship between mortality and fishing pressure for reef fish. A spatial comparison of Jamaican fishing sites subject to different levels of fishing pressure demonstrated for a number of species including lutjanids and serranids that Z and F were higher at locations with higher effort, and similarly Z/K was greater at fished reefs than M/K from an unexploited bank (Munro, 1983; note M=Z at unfished locations, see 4.8).

The age structure and growth rate of fished and unfished populations of fish may be expected to vary. Gulland (1983) described how both K and L_∞ would vary with stock density, and therefore with fishing pressure. As a stock is fished the density decreases giving rise to an increase in growth rate, and to an increase in the size of individuals of a given age, thus altering the age structure of the population. For example, it has been shown by several authors that depending on the state of the fishery (unfished / new or equilibrium), the level of fishing pressure can significantly affect the mean size of fish and their overall growth rate (Gulland, 1983; Schaap & Green, 1988; Filipsson, 1989; Klein, 1992; Ross and Nelson, 1992). Most of this work relates to temperate water species. Whilst mean size has been reported to decrease due to fishing for demersal study species (see below) this has not been related to age or growth and the effects of fishing on growth of snappers, groupers and emperors is little reported. Carneiro-Ximenes and Fonteles-Filho (1988) reported a greater growth rate for *Lutjanus purpurus* in Brazil which they related to fishing. A model developed by Huntsman and Schaaf (1994) to investigate reproduction in groupers also investigated the effect of various compensatory mechanisms on maintaining fertilized-egg production despite increased fishing mortality. One of these was highlighted as ‘accelerated growth’.

Fishing tends to target larger individuals in a population leading to a reduction in mean length and change in length frequency distribution of fished and unfished populations. Examples exist for demersal banks and deep reef slope species, and a considerable number of fishery independent spatial surveys indicate size reductions related to fishing (see 5.1.3). The age at entry into the Hawaiian fishery for *P. filamentosus* decreased with increased fishing mortality (Ralston and Kawamoto, 1985). In the Gulf of Mexico, following large catches of old fish, total catch was reduced and the fish were smaller (Hunstman and Waters, 1987). For the Tongan seamounts, Langi and Langi (1989) inferred size reductions related to fishing pressure from spatial comparisons. However, Latu and Tulua (1992) and MRAG (1994) found no evidence for a change in mean length over time, but there is some evidence that spatial differences in size distribution may occur naturally (see ?). This suggests caution is required when comparing population demography between locations.

Where differential growth rates occur between the sexes, and fishing targets larger fish, fishing could conceivably result in changes to the sex ratio of the population in total and at any given length. For hermaphroditic fish, the effects of fishing may be complex. If sex reversal is related to age or size for example, then removal of large fish could dramatically affect the sex ratio and structure of the population. The effects of exploitation on gonochoristic and protogynous fish populations have been modelled (Bannerot *et al*. 1987). Protogynous species will only be more affected by fishing than gonochoristic species where fishing occurs to the extent that the availability of males is limiting.

The spawning stock biomass is that part of the total biomass represented by reproductively active fish. By reducing the total biomass, fishing decreases the spawning stock biomass. Since individual fecundity is considered to increase exponentially with size (although evidence for this is poor for snappers due to variable methodology in determining fecundity, Grimes, 1987), and fishing usually removes larger individuals first, the effect of fishing on population fecundity is greater than might be predicted simply from a reduction in biomass. Depending upon the fishing gear, length at capture may be less than that at which fish mature, directly affecting reproductive capacity. For fish which form spawning aggregations (eg. serranids), catchability and fishing mortality are increased. The effect of fishing is therefore expected to be a reduction of the recruitment of fish from larval settlement. However, the relationship between stock size and recruitment may be highly variable, and environmental variations may have a greater impact than fishing. By reducing the number of reproductive age classes in a stock Pauly (1987b) argues that fishing may reduce the buffering of any natural variation in recruitment, leading to increased variability in egg and larval production and ultimately biomass variation. At the extreme, where fishing reduces the spawning stock biomass below sustainable levels, recruitment overfishing is said to have occurred, and a stock may collapse. Recovery of depleted resources is complicated by considerations of larval dispersal and the source of potential new recruits (see 3)

5.1.3 Multispecies effects of fishing

Ecosystem overfishing occurs when intensive fishing on a multispecies resource leads to changes in the community structure (Pauly, 1979; 1988). Examples exist in the Philippines (Russ and Alcala, 1989), in the Caribbean (Koslow *et al.* 1994). Changes in the relative abundance (species composition) of species in a multispecies stock are related to fishing intensity, the relative catchabilities of the species and the level of interaction between them. Removal of competition, predators or prey are postulated to lead to species composition changes. Competition and predation are considered the most likely forms of interaction in models of multispecies fisheries and may result in changes in recruitment, natural mortality (eg. of prey), and growth (eg, May et al, 1979, in an example of a temperate water fishery; Polovina, 1984 in an example of a coral reef ecosystem).

Pauly (1979) described the effects of the Gulf of Thailand trawl fishery. Prey species declined more rapidly than predators such as lutjanids. He postulated that prey are exploited close to their maximum sustainable yield by the predators, and were therefore more sensitive to increased mortality due to fishing. However, no analogous situations have been reported for reef fisheries (Russ, 1991).

In general, fishing tends to remove top predators. They are targeted as highly valued food fish (although many examples of where prey fish are the target also exist). Furthermore, aspects of their biology tend to make them more vulnerable to fishing: they tend to be aggressive predators with high catchability to gear such as baited hooks and lines; they are slow growing so even if the number of fish remains constant, the biomass will be reduced and the relative proportion of predators to fast growing species in the catch will change; they have relatively lower reproductive capacity and take longer to reach maturity so higher total fishing mortality will occur before spawning resulting in a lower spawning biomass - recruitment may be reduced. Within demersal bank and deep reef slope fisheries, there is some expectation that serranids may be depleted to a greater extent than the lutjanids or lethrinids due to their ecology and population demography.

From fishery independent underwater visual census (UVC) work in fished and unfished locations, and from catch and biological data, there exists much evidence of single species (abundance and size differences, including members of the lutjanidae, lethrinidae and serranidae) and multispecies (changes in catch composition) responses to fishing (Craik, 1981a; 1981b; Bohnsak, 1982; Russ, 1984; 1985; Ayling and Ayling, 1986; Lock, 1986; Koslow *et al*. 1988; Samoilys, 1988; Russ and Alcala, 1989;Ayling *et al*. 1992; Polunin and Roberts, 1993; Koslow *et al*.,1994; Watson and Ormond, 1994; Jennings *et al.* 1995). These studies indicate that fishing results in a reduction in abundance of the top predators and reduction in average size of fish.

Removal of predators is postulated to lead to increase in prey (eg. Beddington and Cooke, 1982; Beddington, 1984; Munro and Smith, 1984; Koslow et al, 1988). However, little evidence exists for this for fish in coral reef systems and observations contradict these predictions. For example, prey release was not found to occur in Seychelles (Jennings *et al*, 1995), and following experimental removal of adult groupers from reefs in the Red Sea, no change in the abundance or species composition of prey occurred (Shpigel and Fishelson, 1991), although examples of invertebrate prey release are known (eg. Mclanahan and Muthiga, 1988, see also Jennings and Lock, in press). Similarly, little evidence exists to indicate that removal of prey results in measurable changes of fish, individually or in total, at other trophic levels (such as a reduction in predator populations, see Russ, 1991). These examples relate to local overfishing of predators and it is likely that due to spatial connectivity of the resources, stocks can be maintained despite fishing. By contrast the example of the Caribbean (Koslow *et al*, 1994) relates to gross overfishing of the whole system. Lacking sources of replenishment, community responses become detectable.

Natural variations in recruitment are also offered as an explanation of the failure to detect community effects. Larval supply is considered to be a more important control on abundance than predation (Doherty and Williams, 1988). Where predation effects are most likely to be significant is on survival of newly settled fish. High rates of mortality of coral reef fish are experienced in the first year, and much of this occurs within the first week after settlement (Shulman *et al*., 1983; Doherty and Sale, 1986; Doherty and Williams, 1988), although strategies to reduce this such as different 'nursery grounds' and adult feeding areas have developed (see Section 4.5). However, natural variability in recruitment means that changes in the abundance of prey fishes due to reduction in predators through fishing will be difficult to detect. For example, Kulbicki and Wantiez (1990) reported up to a 13 fold decline in biomass and density estimates from trawl surveys in New Caledonia which were unrelated to fishing. The largest changes observed related to faster growing species including lethrinidae, rather than slow growing species.

MRAG (1993) argue that if populations are below their carrying capacity due to factors such as fishing, predation, disease, environmental and recruitment variability, then significant competition will not occur. Indeed, evidence for resource limitation due to competition is lacking (Doherty and Williams, 1988). For snappers and groupers on the Cuban shelf depletion of *L. Synagris* has resulted in replacement by 3 sp of Haemulidae. Despite protection from fishing for 10 years the Lutjanid resource has not recovered and competition with the Haemulidae is proposed as the reason (Claro, 1991). This is an extreme case of heavy fishing. Between tropical case study demersal species, which may be grouped at the same trophic level, the ability to switch prey items in the diet, and the separation of species by depth, habitat, feeding times and locations mean that both intra- and inter-specific competition is likely to be low. Competition for space rather than food, however, may be important for serranid species, especially at the carrying capacity of the local environment (See Section 4.5).

It is concluded that environmental influences, particularly in relation to egg and larval production and settlement, are likely to have a greater impact than biological interactions. Similarly, in fished populations, technical interactions are likely to have a greater measurable effect than biological. Russ (1991) proposed that future development of more realistic models to investigate effects of fishing on multispecies resources will occur if the fish community is regarded as existing in an open non resource limiting, non equilibrium system. They should not necessarily be characterised by strong interactions between species, but should be strongly influenced by variability in recruitment.

5.2 Predicted effects of fishing

The introductory section of this chapter described qualitatively the effects that fishing may have on single stocks and on multi-species communities. In this section, modelling and simulation techniques are used to derive more quantitative predictions for these effects, within the context of tropical reef and bank fisheries. Section 5.3 will discuss trends and effects that were actually observed in the data, and highlight the correlation between the observed and predicted events.

5.2.1 Methodology

The mathematical model that was used for this analysis is described fully in Section 2.3. The simulation program MIDAS, which implements that model, is also discussed.

In order to investigate the effects of fishing, it is necessary to define a generic ‘species’ for each case study area to serve as a baseline for comparison. These ‘baseline species’ must have parameters representative of the commercially important species in each region, and sensitivity studies around this baseline reflect the range of parameters found amongst these species.

The biological parameters derived for each of the major species are listed in tables 4.2 and 4.3 on page 49. Accordingly, the two baseline species were defined as follows:

Parameter	Indian ocean banks species		Deep slope species	
	Baseline	Parameter range	Baseline	Parameter range
M	0.5	0.4 - 0.6	0.25	0.2 - 0.4
K	0.25	0.15 - 0.3	0.15	0.1 - 0.2
Lm	0.6	0.3 - 0.7	0.5	0.3 - 0.7
Lc	0.45	0.3 - 0.7	0.45	0.3 - 0.7

Table 5.1 : Definition of parameters for the banks and deep slope baseline species, and the parameter ranges found within the two study sites.

In the table, M is the natural mortality rate and K is the Von Bertalanffy growth rate. Lm and Lc are the length of maturity and length at first capture respectively, expressed as a fraction of L_∞. Lm values for Tongan stocks are unknown, so the value 0.5 has been used, as was discussed in Chapter 4. Estimates of Lm and Lc were very variable and subject to additional uncertainty because of uncertainty about the L_∞ estimates. Generally, the lowest measured Lc’s were used for the baseline species, while wide ranges of Lc and Lm were considered for sensitivity studies. For the baseline species, knife edge selectivity was used for both Lm and Lc, but selection ogives were also considered and are discussed later.

These are the most significant parameters necessary to define a species for the purpose of this analysis. A number of other parameters need to be specified for the model, but these either have no or little impact on the results of this section, or will be discussed individually.

- *Parameters having little impact on the model outputs*. Initial single species sensitivity studies showed that some parameters do not have a major effect on the model outputs when applied to the baseline species. The relative importance of the various parameters is discussed on page 86. Examples of these are *b*, the power in the length-weight relationship (nominally b=3), and the length of the period over which adult biomass is averaged to calculate spawning stock biomass.

- *Parameters having only a scaling effect on the model outputs*. Certain parameters such as initial stock biomass affect the magnitude of the outputs, but do not affect relative trends, or ratios between states such as maximum sustainable yield (MSY) and unexploited population biomass. L_∞ and the length-weight parameter a are other examples of such parameters. These were merely set to convenient values for this study. Of course, in later work where specific species are examined, appropriate values for these parameters have been used.

- *Parameters which cannot be estimated reliably*. The most important parameter in this category is density dependence of recruitment. For the case study sites, data from which to estimate a stock recruitment relationship does not exist. There was only one related species for which published data could be found, and that was the silk snapper *Lutjanus synagris* from Cuba (Carillo de Albornoz, 1979, courtesy of Myers' database, 1994). However, there were insufficient data at low biomass levels for any fit to be conclusive. Density dependence in the model is specified by means of the parameter d, which represents the degree of density dependence, with d assuming values between 0 and 1 with d=1 corresponding to constant recruitment (see Section 2.3). It was felt that constant recruitment is inadequate to represent stock effects at low biomass, so a degree of density dependence d=0.8 was selected for the baseline. This implies that when the spawning biomass (SSB) has been reduced to 20% of initial levels, recruitment will have dropped by 5%. Where appropriate to the following discussion, comparisons with near-constant recruitment (d=0.95: 20%SSBo ⇒ 99.7%Ro) and lower density dependence (d=0.6: 20%SSBo ⇒ 80%R) will be made.

The commercially important banks species in Seychelles typically have mortality rates of about M=0.5 and are consequently rather shorter lived that the majority of Tongan deep slope species which have mortalities around M=0.25. Similarly, the banks species' growth rate K=0.25 is higher than the deep slope ones, at K=0.15. Only one deep slope species, *Epinephelus morhua*, approximates the banks parameters, having M=0.4 and K=0.2, but currently makes up a very small proportion of the catch.

A significant similarity between these parameter sets is that M/K is approximately equal to 2 in both cases. Indeed, referring back to the species parameters in tables 4.2 and 4.2, one can see that on average M/K=2.00 with a standard deviation of 0.23. Polovina (1987) demonstrated the influence of the ratio M/K on spawning stock biomass levels, and Kirkwood *et al* (1994) have shown that the ratio M/K has a significant influence on theoretical maximum yield and optimal effort levels. The fact that all the commercially important species in the case study areas appear to have a similar M/K ratio suggests that there will be important similarities in their response to fishing.

5.2.2 Single species effects

Figures 5.1 and 5.2 show how the yield and spawning stock biomass (SSB) of the deep slope baseline species change with time in response to high, medium and low changes in effort.

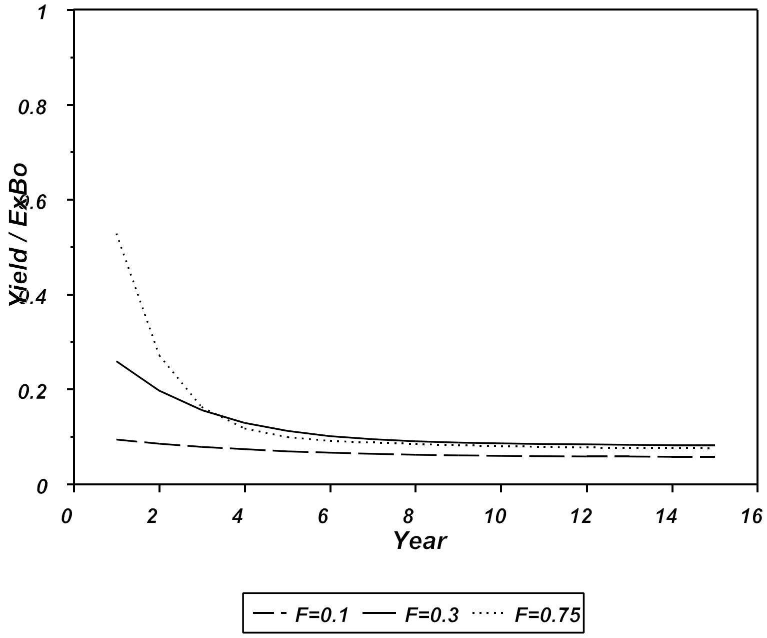

Figure 5.1 : Response of yield biomass to various levels of effort (baseline deep slope species).

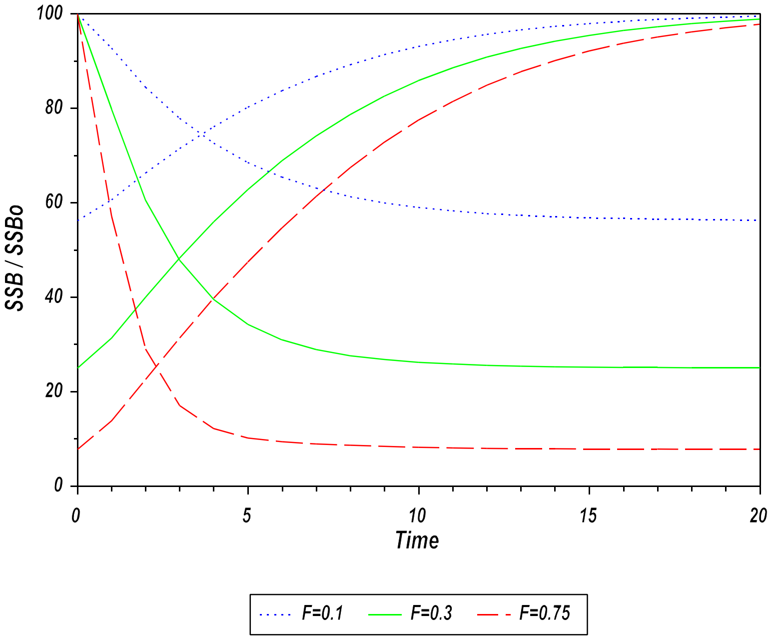

Figure 5.2 : Decline and recovery of spawning stock biomass for three different levels of effort. Each pair shows (i) biomass decline from the unexploited state under effort, and (ii) recovery of biomass initially in equilibrium at that effort, after removal of effort (baseline deep slope species).

Yield is expressed as a proportion of the virgin exploitable biomass (ExBo), while SSB is given as a proportion of the virgin SSB (SSBo). Exploitable biomass is the annual average of the biomass of fish with length greater than Lc50. At time 0, the species is in unexploited equilibrium, then the effort jumps to its new level and stays constant at that level for the duration of the simulation. With constant effort, the first year's yield is always the highest, and both yield and biomass show a smooth, monotonic decline from the initial value to a new equilibrium.

Figure 5.2 also illustrates the recovery of biomass under effort change, for the same three levels of fishing mortality. Each pair of curves indicates (i) biomass declining from virgin levels under constant effort, and (ii) biomass initially in equilibrium at that effort, recovering after the removal of effort.

As will be shown later, the solid line corresponds to F=0.3=1.2M, which is approximately the optimal level of effort for this scenario. The higher effort level (F=0.75) will provide greater yields in the first two years but lower ones thereafter, and although the long-term loss in yield seems small, the spawning biomass at this effort has been reduced to less than 10% of its unexploited level. This would leave the stock far more vulnerable to natural variability than the maximal case, F=0.3, which leaves the spawning biomass at around 25% of unexploited levels. The lowest effort shown provides a more uniform yield but does not fully exploit the available resource.

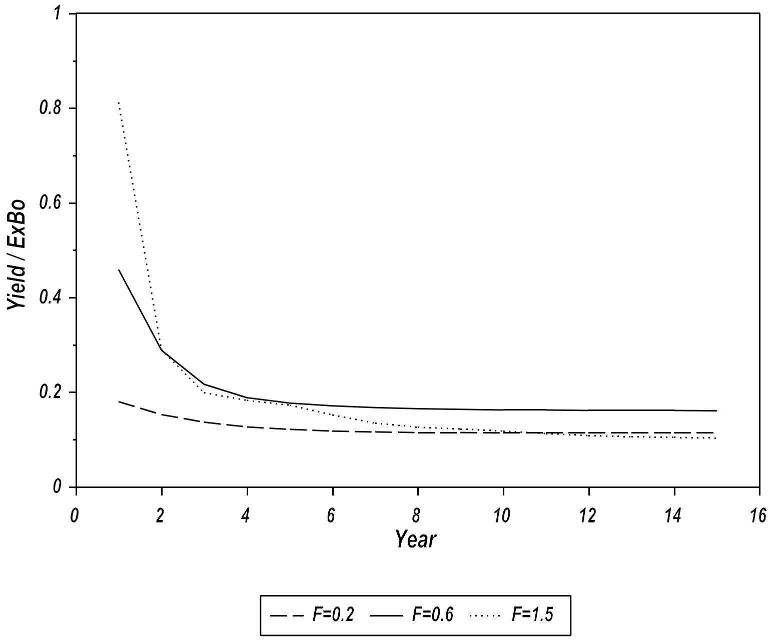

Figure 5.3 : Response of yield biomass to various levels of effort (baseline banks species).

The baseline Indian Ocean banks species displays a similar trend in yield and SSB over time, although there are differences in scale due to the higher mortality of these species. The analogous graphs are shown in figures 5.3 and 5.4. In general, the species can tolerate higher fishing effort, providing greater yields but declining faster to lower biomass levels. Again, the solid line indicates approximately the MSY at F=0.6=1.2M, so this fishing mortality is the one that is used for comparative purposes in this section. Whether or not this is a suitable target for management of the fishery will be discussed in the next chapter. Given that the spawning biomass ends up at around 15% of its virgin level, a cautious manager would probably not wish to aim too close to this target. However, it is suitable for the analysis described in this section.

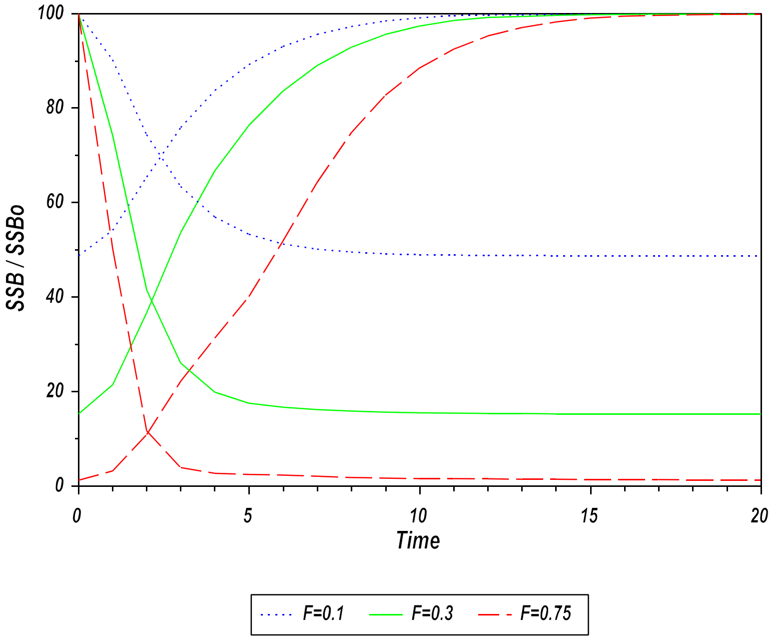

Figure 5.4 : Decline and recovery of spawning stock biomass for three different levels of effort. Each pair shows (i) biomass decline from the unexploited state under effort, and (ii) recovery of biomass initially in equilibrium at that effort, after removal of effort (baseline banks species).

It is interesting to note that, for MSY levels of effort, the baseline deep slope species takes three times as long to recover as to decline. It will take 5 years for a half of the lost biomass to be recovered, and a further 10 years for the biomass to reach 90% of virgin levels. For the Seychelles species with their shorter lifespans and faster responses to fishing, the corresponding times are 3 and a further 5 years. In both case study regions, recovery after low effort is almost as fast as the decline, but recovery from high effort will take five times as long as the initial decline. Recovery rates from low biomass are approximately 8% per year for deep slope and 12% per year for banks species.

The differences between the two case study baseline species are most clearly illustrated by the next two figures, which show the equilibrium (steady state) yield that would result from different levels of fishing mortality. Once again, yield is expressed as a proportion of the virgin exploitable biomass. These graphs also show the influence of the stock recruitment relationship.

The choice of stock recruitment relationship (SRR) for the baseline species (d=0.8) was discussed in the previous section. Figures 5.5 and 5.6 show the yield-effort curves for the baseline deep slope and banks species, and illustrates how dependent these curves are on the degree of density dependence assumed. Accepting a SRR with density dependence d=0.8 implies that the maximum yield can be obtained with a fishing mortality of about F=0.6 for banks and F=0.3 for deep slope species. This represents a fishing mortality 20% higher than the natural mortality, for both species. More accurate estimates of MSY for the various species will be discussed in the next chapter. Generally, underestimating the density dependence of a species means that higher yields are obtained than were expected, whereas overestimating it can lead to a false sense of security about the stock's tolerance to high effort.

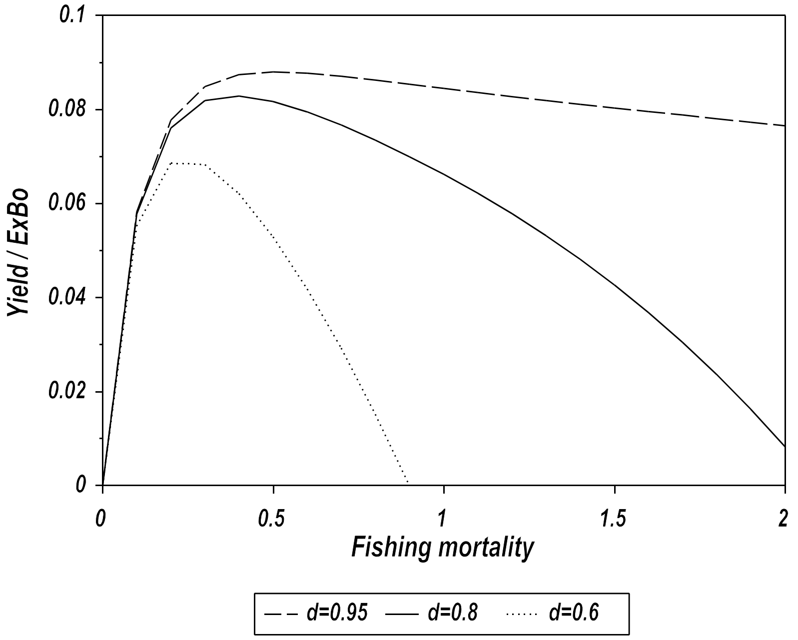

Figure 5.5 : Yield-effort curves for the baseline deep slope species showing the influence of the stock recruitment relationship.

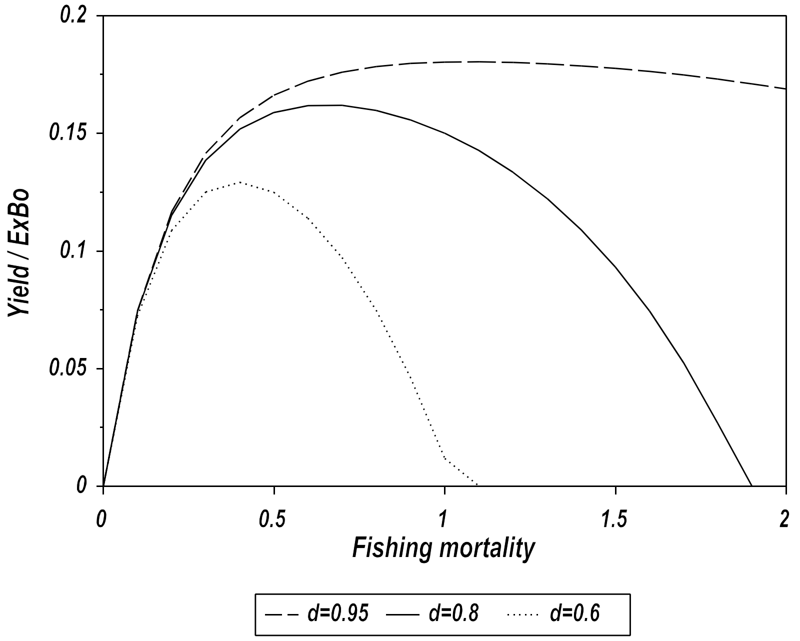

Figure 5.6 : Yield-effort curves for the baseline banks species showing the influence of the stock recruitment relationship.

Looking at figures 5.5 and 5.6, it is clear that the parameter differences have resulted in a number of significant changes to the shape of the yield-effort curves. The difference in height of the curves is a result of the difference in mortality: the deep slope species have half the mortality rate and so produce approximately half the MSY. This relationship holds largely because the M/K values are similar. The F at which MSY occurs has also halved in the deep slope case, although the maximum F (F at which the stock becomes extinct) is similar - the right hand side of the curve slopes far more gradually than it does for the banks species. This is a result of the fact that Lc and Lm are taken to be so close to each other in Tonga, i.e. the fish are caught at about the age at which they mature. This is largely an assumption, since data by which Lm could be estimated for the Tongan deep slope fishery are not available. In Seychelles, however, it is known that the banks species mature later, with Lm approximately 0.6. This means that a larger proportion of the catch is immature, increasing the susceptibility of the stock to overfishing.

The influence of biological parameters on the equilibria under exploitation may be summarised as follows:

- the lower the mortality of a species, the lower the potential yield and the lower the allowable effort.
- lower growth rates tend to lead to higher potential yield but this is reached at lower effort levels.
- a high length at maturity acts like a lower growth rate in that it results in increased yield but lower MSY effort and maximum effort levels.
- if maturity is better represented by an ogive, rather than knife-edge selectivity, the MSY yield and effort are not greatly affected, although the tolerance of the species to overfishing is enhanced.
- increasing the catchability of a species does not change the potential yield, but does ensure that yield is reached with lower effort.

Of particular interest in this study has been the transient response of the species' states to fishing, rather than just the equilibria that ensue. A single stock's yield may decline gradually until a new equilibrium is reached, but when the single species response of a number of different stocks is superimposed, the catch composition may appear to vary in a far more complex fashion. This type of effect will be discussed in the next section.

● MULTI-SPECIES EFFECTS

When a number of different species of fish are exploited simultaneously, it is common for the species composition of the catch to vary with time. There are three major reasons why this may occur. Firstly, species with different biological parameters respond differently to fishing, e.g. their abundance may decline at different rates leading to a real shift in species composition of the resource. Secondly, biological interaction such as prey release may cause changes in biological parameters such as mortality, which can once again lead to a change in species composition. Thirdly, changes in fishermen's behaviour may influence relative catchability of species which can lead to changes in the catch composition which are not a real reflection of the resource abundance. This section describes simulations that were performed to assess the importance of these effects in tropical deep slope and banks fisheries.

5.2.3 Multi-species effects due to biological differences

It hardly needs to be said that species with different biological characteristics will react differently to fishing effort, and so the species composition of a multi-species resource will change with time. Of interest here, however, is the degree to which various types of biological differences influence species composition, because this will give an indication of the relative importance of these biological differences to fisheries managers.

Species that differ only in biological parameters of lesser importance may be treated as a guild. Also, quantifying the effect of uncertainty in the parameters can be useful for prioritising research.

The method used for this investigation was to define two species to be identical in all respects except for the parameter to be investigated. MIDAS adjusts unexploited recruitment of the two species until their initial adult biomasses (SSB) are the same, and then fits stock recruitment relationships of the same relative shape through the equilibrium points thus obtained. The two species were then subjected to the same level of simulated effort, and the ratio between the final yield biomasses of the species (YR) and a similar ratio between the final adult biomasses (SSBR) were determined. The way in which these ratios changed under different circumstances was then investigated.

- MORTALITY AND GROWTH

In looking at variability of yield and biomass with biological parameters, mortality and growth must be the most significant. It has also long been clear that these parameters are closely linked, so much so that the importance of the ratio M/K on MSY has been investigated by many authors (e.g. Polovina, 1987, and Kirkwood *et al*, 1994).

This section investigates the influence mortality and growth differences between species, by considering such variation as a change in M/K. Where M and K vary proportionally, i.e. M/K is the same for all/both species, species composition will not change under effort. This appears largely to be the case for the case study sites, where the M/K estimate is close to 2 for all species. Possible variation or uncertainty in the parameters estimates could result in deviation of M/K to between 1.5 and 3.5, and it is over this range that the effect of such differences were investigated.

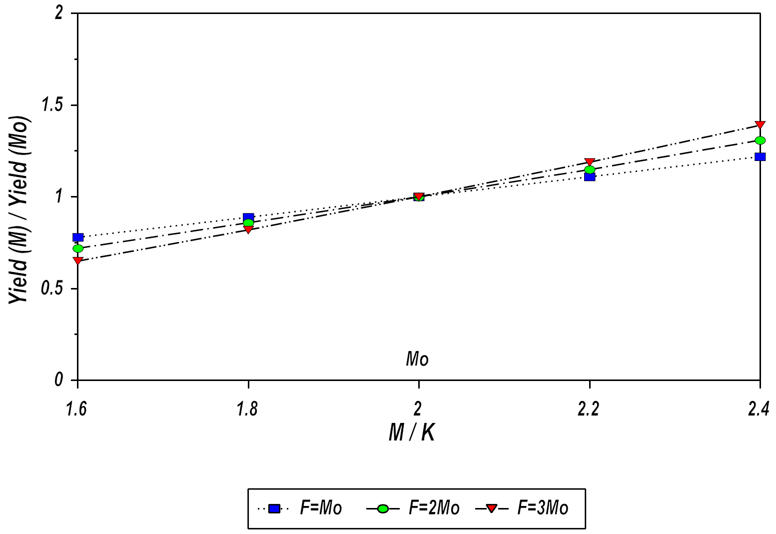

Figure 5.7 : Sensitivity of catch composition to relative mortality

Figure 5.7 shows the effect of differences in mortality on two otherwise identical species. One of the species had Mo=K, while the other, with the same K, could have M with values between 1.6K and 2.4K. Mortality has a linear effect on catch composition and, under MSY effort, could cause a potential shift of up to 25%. The symmetry of the figure indicates that increases and decreases in M/K have analogous effects. A figure showing change in species composition was nearly identical, indicating that yield and biomass respond to mortality in the same way, and to the same degree.

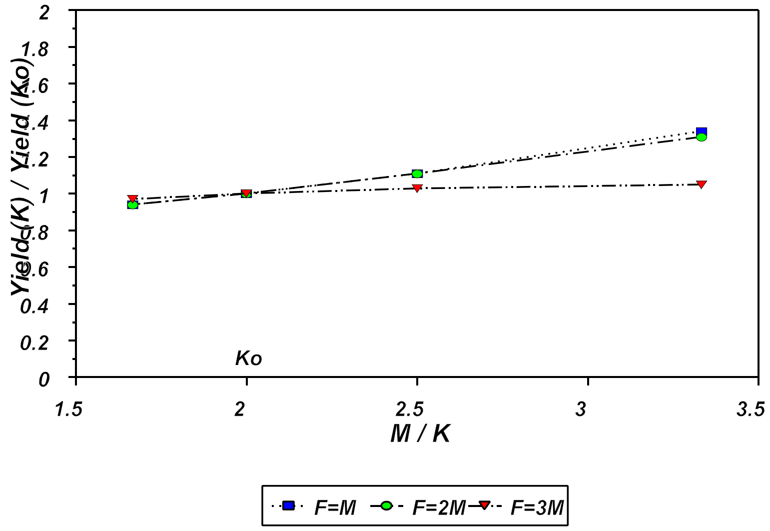

Figure 5.8 : Sensitivity of catch composition to relative growth rate

Similar curves for differences in growth rate are given in figure 5.8. M/K varies over the same range as before, but this time the difference in M/K is brought about by differences in K rather than in M. K does not seems to have as great an effect on yield as does mortality, however the effect on biomass ratio is far greater. Besides magnitude, the variation is also in the opposite direction, with higher M/K producing bigger yields but smaller adult biomass (figure 5.9).

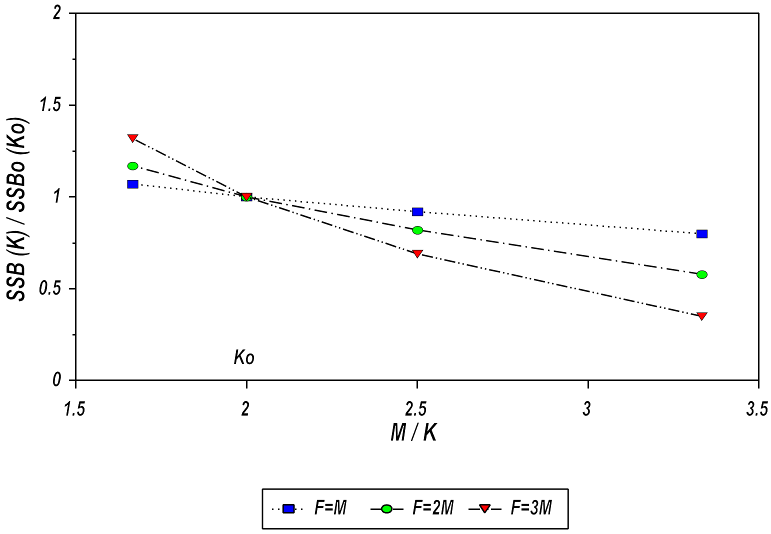

Figure 5.9 : Sensitivity of adult biomass composition to relative growth rate

Comparing the effect of M and K differences on stock biomass, it seems that M has a greater influence when effort levels and low. However, under high effort, differences in K are more significant.

The fact that the mortality effect is linear means that such effects will be more predictable. Growth is more sensitive to differences (or uncertainty in estimates) when K is smaller, i.e. M/K is higher. If K is overestimated, results could differ more widely from expectation than otherwise. Yields are better than anticipated, but biomasses are far worse. In general, if forced to choose between two equally plausible estimates for K, the lower one will be more conservative and result in less risk to the stock. This will result in an small overestimate of yield, but a larger underestimate of biomass.

- SPAWNING SEASON

Simulations were generally not very sensitive to the length of the spawning season, except at very high effort levels. The spawning season in the model is the period over which adult biomass is averaged to calculate an annual SSB to input into the stock recruitment relationship. The standard parameterisation used the average of the whole year, but even if this period was reduced to the last third of the year, the change in expected yield was less than 10%, for all F<1.5M. This is because even with the SRR parameter d=0.8, recruitment is very density dependent and so is fairly insensitive to adult biomass, except where effort is high. Both baseline species were robust in this way.

- MATURITY OGIVE

In most of the simulations done for this report, knife-edge maturation has been assumed. In this case, a symmetric maturity ogive was added to one species, with Lm75 up to 1.1*Lm50 (the maximum estimated for the study species). Adding a maturity ogive to a species causes no discernable difference in yield, but can cause up to 20% increase in the adult biomass. This change increases with effort, but has the same magnitude for both banks and deep slope species.

Having the ogive tended to reduce the sensitivity of the outputs to changes in other parameters. This would suggest that where the maturity ogive is symmetric but hard to quantify, the knife edge model would provide a satisfactory conservative alternative.

- GEAR SELECTION

Gear selection ogives for the case study species are typically given by an Lc75 of 5-10% above Lc50. Such an ogive makes virtually no difference to yield or biomass, at all effort levels considered. This result is independent of M/K, within appropriate ranges.

- POWER IN LENGTH-WEIGHT RELATIONSHIP

Varying the parameter b between 2.5 and 3.5 had less than 15% effect on species composition for F<2M.

- ASYMPTOTIC LENGTH AND LENGTH-WEIGHT CONSTANT

As was mentioned earlier, these parameters have only a scaling effect on the model, changing population numbers to biomass. This is the case because of the way that the stock recruitment relationship is defined relative to the unexploited equilibrium, as described in Section 2.3.3.

5.2.4 Biological interaction

Section 2.3.4 describes the model for biological interaction (prey release, competition) that was included in MIDAS. It is impossible, from the data that is available, to try to parameterise this model for the case study areas. However, it is useful to consider how different levels of interaction change the effect that fishing has on a multi-species system. It is also possible to choose interaction levels which represent the maximum likely to be observed in the case study fisheries and which can thus be used to estimate the maximum effects that could reasonably be expected.

As was discussed in the introduction to this chapter, prey release is the biological interaction effect most likely to be encountered in tropical reef and bank fisheries. The modelling for this section has thus centred around prey release. Two identical species were defined, each having the same (baseline) set of parameters and having the same unexploited adult biomass. One was designated predator and the other prey, so that as the biomass of the predator declines (as a result of fishing) the mortality of the prey decreases. Fishing effort was applied to both species equally. The predator is not affected by changes in the prey biomass, so reacts to fishing in the same way as a baseline species would without interaction.

An important difference between the two case study areas is that of the age at which target species may be exposed to predation. On the Seychelles banks, there is no evidence for significant predation on adults of target species. Accordingly, only juveniles of less than a year old are affected in the model. However, there is some evidence that adults of Tongan deep slope species may be exposed to predation. In the deep slope model, prey release can thus affect fish up to 60% of $L\infty$. It is relevant to reiterate here that larval stages of the life-cycle are not modelled - a reduced number of 'juvenile-equivalent' fish represent the number of births that would be necessary to arrive at the correct population size at true recruitment, given constant mortality across all ages (see section 2.3.3).

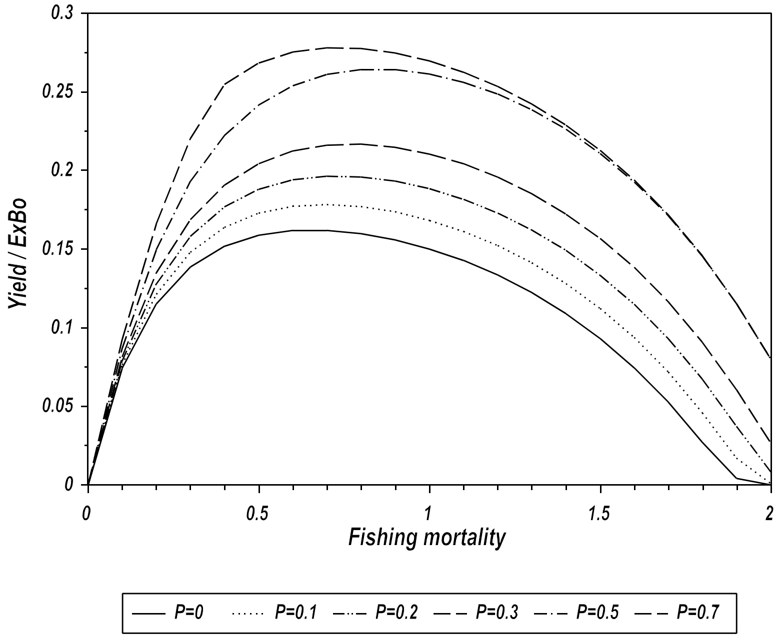

Figure 5.10 : Yield-effort curves for various levels of interaction, for the baseline banks species

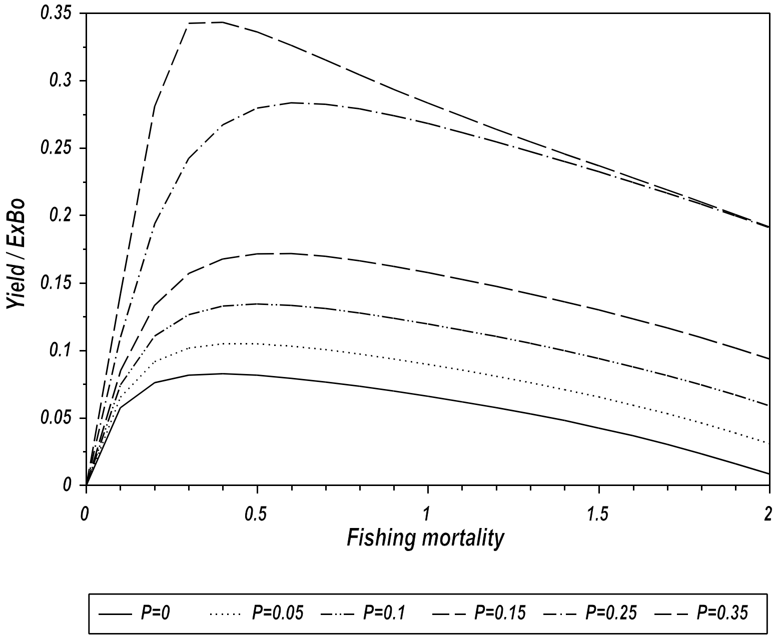

Figure 5.11 : Yield-effort curves for various levels of interaction, for the baseline deep slope species

Figures 5.10 and 5.11 show how the yield-effort curves change in shape and size for various degrees of interaction as determined by the interaction parameter P. Both species are subject to the same level of fishing effort. The solid lines indicate the predator response, equivalent to the baseline curve without interaction, and thus identical to the solid lines of figures 5.5 and 5.6. The highest P values illustrated in each case are sufficient to reduce the natural mortality of the released portion of the stock to zero under higher fishing mortality (hence the discontinuity in the graphs, as M cannot fall below zero). This would correspond to a situation where all the juvenile mortality is due to predation by other target species. This is clearly far too high a level of predation for the case study species but it does provide a useful indication of the maximum possible predation effect.

As expected, in both cases the potential yield of the prey species increases as the predator is removed. The curves suggest that prey release could have a far greater benefit in terms of increased yield for the longer lived deep slope species than for the shorter lived banks ones. The effort at which MSY occurs is also increased, meaning that greater levels of effort are theoretically possible. However, the yield-effort curves are fairly flat around this MSY region. This indicates that if effort was set to reflect the MSY of the stock without taking interaction into account, then even if there is prey release taking place the yield obtained will not be smaller than predicted with the simpler model. In fact, it will not be much less than the maximum theoretically possible for the stock even when interaction is taken into account.

Figures 5.12 and 5.13 show how the yield of the two species changes with time in response to a sudden step in effort applied to both species, given varying degrees of interaction. The species start from unexploited equilibrium, effort jumps to the baseline level (approximately MSY, as discussed in the previous section) and stays at this level while the system settles into a new equilibrium. The solid lines represent the states of the prey when no predation is taking place, which is equivalent to those of the predator in all cases.

It is clear that when the interaction is small, the difference between the predator and prey response is so small as to be undetectable. As the predation level increases, the characteristic increase in yield and biomass of the prey species becomes more visible. Even so, it is interesting to note that the predator and prey responses only begin deviating significantly from one another after 4-5 years, in both case study species. This implies that one cannot expect to see the effects of prey release in these fisheries until 5 or more years after the onset of fishing. This time delay is largely independent of the size of the step in effort that is driving the response. The effort used in these figures is close to MSY levels; smaller or more gradual increases in effort will have a less noticeable effect.

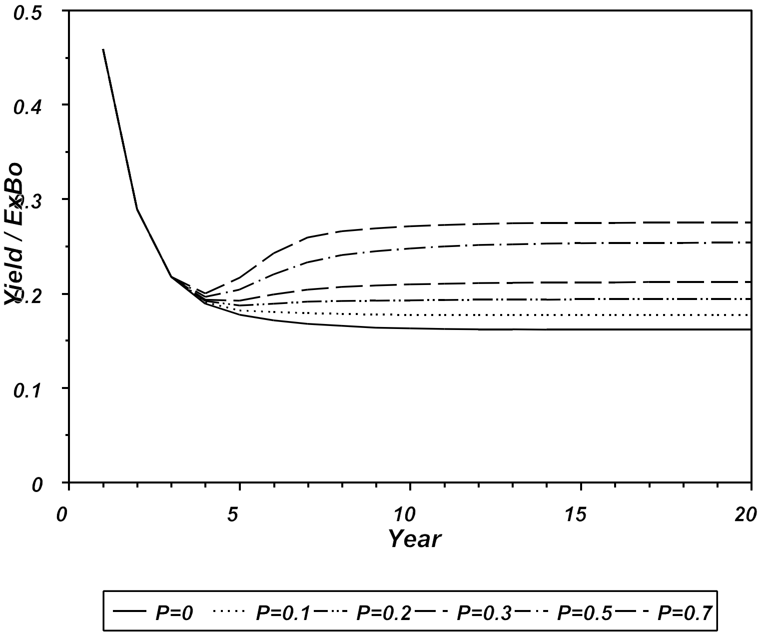

Figure 5.12 : Yield biomass for the baseline predator-prey system with various values of the interaction parameter P, for the baseline banks species with F=0.6

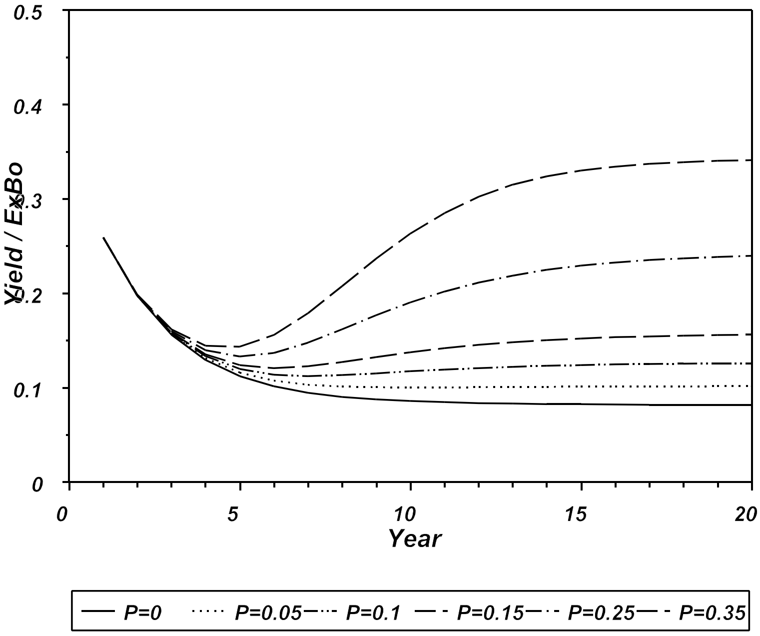

Figure 5.13 : Yield biomass for the baseline predator-prey system with various values of the interaction parameter P, for the baseline deep slope species with F=0.3

As mentioned before, the higher levels of interaction are totally unrealistic for the species in question. Unfortunately there is no way of knowing just what value for P would be appropriate. It remains then to select appropriate values for P to use for subsequent simulations, to indicate the effect of these interactions while bearing in mind the uncertainty in their magnitude.

Under a specific level of fishing effort, the biomass of the predator will decline until a new equilibrium is reached. At the new equilibrium, the vulnerable age groups of the prey species thus experience a lower mortality rate, due to the reduction in predation. This final mortality rate can be calculated for different levels of predation, and this then gives an indication of how much of the equilibrium mortality on these age groups is due to predation. The final values for M given the baseline effort levels (F=0.6 and F=0.3 for Seychelles and Tonga respectively) are tabulated in table ? below.

Degree of predation P	**Banks species** Final M	% of M_0	Degree of predation P	**Deep slope species** Final M	% of M_0
0.0	0.500	100%	0.00	0.250	100%
0.1	0.415	85%	0.05	0.212	85%
0.2	0.331	65%	0.10	0.174	70%
0.3	0.246	50%	0.15	0.136	55%
0.5	0.077	15%	0.25	0.061	25%
0.7	0.000	0%	0.35	0.000	0%

Table 5.2 : Final value of mortality rate after MSY level removal of predator and prey

For the purposes of the rest of this study, the baseline degree of predation was chosen such that the natural mortality is reduced to about 70% of its natural level. This implies that 30% of the equilibrium natural mortality is due to predation by species which are themselves targets of the fishery. It is anticipated that the real level of predation may well be much lower, but it is unlikely to exceed this figure. Accordingly, in subsequent simulations, predation levels of P=0.2 and P=0.1 have been used for banks and deep slope species respectively. With the effort approximating MSY for both regions (F=0.6 and F=0.3 respectively as described above), this is known as the baseline interaction scenario.

Figures 5.15 and 5.14 show the transient response of the baseline species to the baseline interaction scenario for the two case study areas. These figures show the most important trends which result from biological interaction: both yield and biomass show an initial decline, then a subsequent recovery. The proportion of predator to prey in the catch in also seen to decline over time.

Given the level of natural variability in fisheries data, as well as the difficulties in data collection and analysis discussed in previous chapters, it is highly unlikely that these small effects will be detectable. Biological interaction of this level amongst deep slope species results in larger effects than is the case with banks species, and could lead to a significant change in catch composition in the long term. However, the increase in prey yield caused by prey release is gradual and the transient responses of the prey and predator do not differ sufficiently to be distinguished within typical fisheries data.

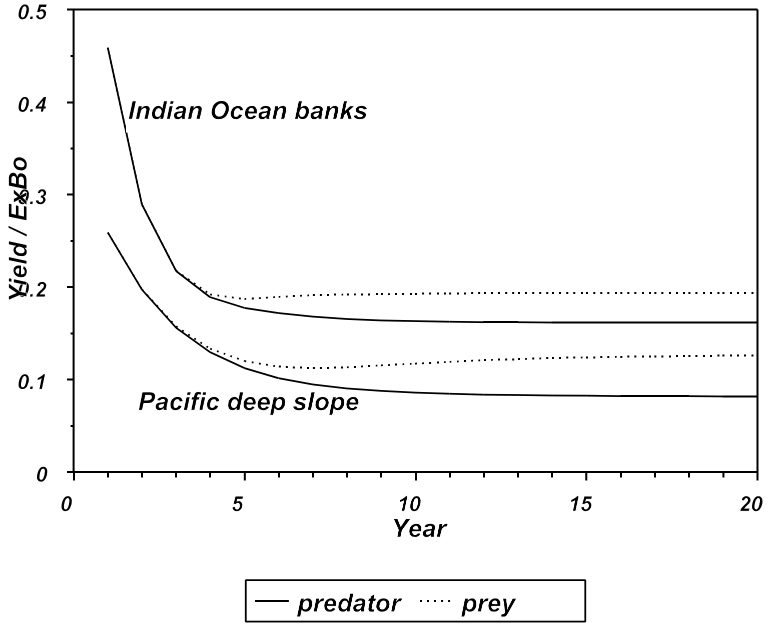

Figure 5.14 : Response of the two baseline species' yield biomass over time, given the baseline scenario levels of effort and predation.

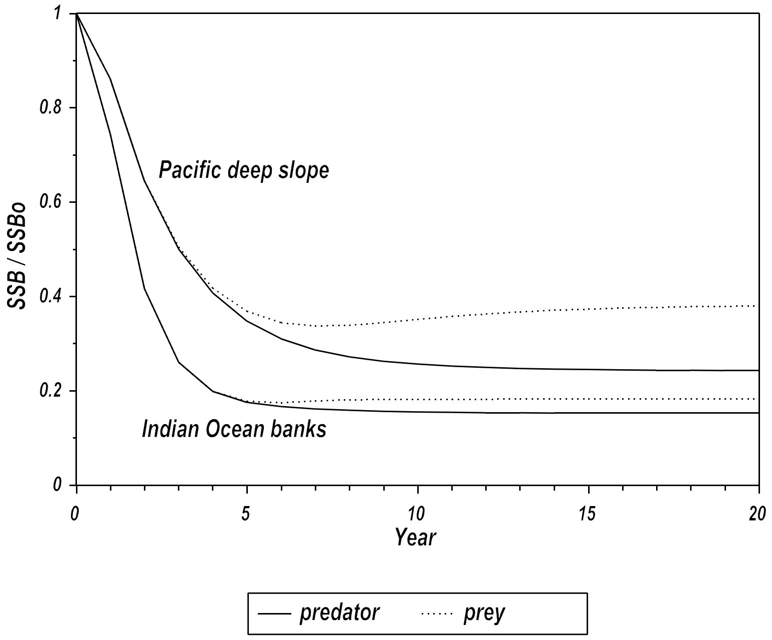

Figure 5.15 : Response of the two baseline species' spawning stock biomass over time, given the baseline scenario levels of effort and predation.

The conclusion to be drawn from this is that, from the point of view of biological interaction, single species models are accurate enough for the assessment of these tropical demersal reef and bank fisheries. The levels of predation that could be encountered between target species are unlikely to be sufficient to significantly influence management decisions.

The yield-effort curve of a prey species experiencing release from predation is similar to that of a non-interacting species with a lower overall mortality. This means that the predation effect can be exacerbated if the prey has a lower M than the predator, or compensated for if the prey M is higher. A more gradual maturity ogive can have the same effect, as can a higher Lc.

5.2.5 Technical interaction

When thinking of multi-species effects, biological interaction is the first thing that usually comes to mind. The previous section considered this aspect of multi-species systems, specifically looking at the effect of prey release on the fishery.

However, factors relating to gear deployment, i.e. technical interactions, can have as significant an effect on the fishery, and on our perception of the state of the fishery. In particular, technical interactions can cause trends in the data which might easily be misinterpreted as being the result of biological interaction. In such a case, the dynamics of the fishery are essentially single-species, but multi-species effects come into play because different species are being caught at the same time, and with the same gear.

Although this section has primarily to do with technical interactions, the effect of biological interaction on these results will also be investigated. This will give an indication of the relative importance of the various effects.

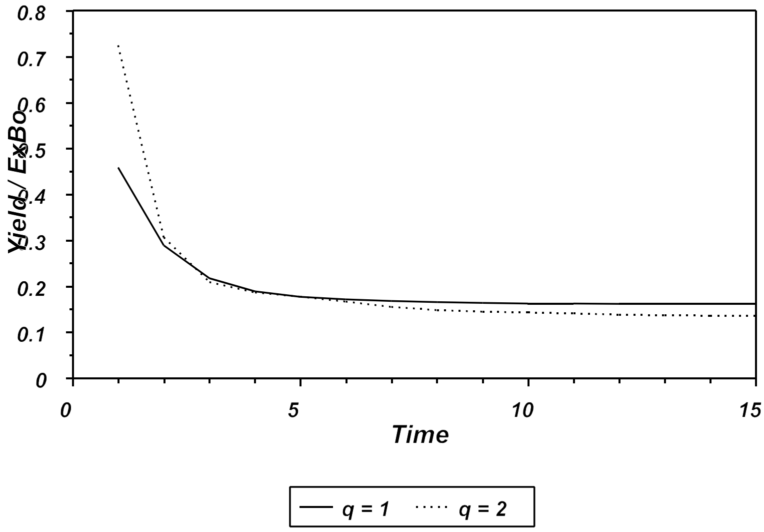

Figure 5.16 : Yield over time of two identical, non-interacting species with different catchabilities

An important factor in the case study fisheries is that of targeting. Because the different study species tend to congregate at different depths, fishermen can target certain species by setting their hooks at the appropriate depth. In fact, as was discussed in section 3.2.2?, some of the Tongan fishermen can target species with a high degree of accuracy. This is essentially a modification of the catchability of the various species. Another way in which catchability could be influenced is through the nature of the fish themselves. Fish that are more aggressive could be more likely to take bait, and may thus have a higher catchability. The effect of differences in catchability will be examined in this section.

Figure 5.16 illustrates the case of two identical banks species with different catchabilities: one twice as high as the other. There is no predation between the two species. The species with higher catchability is being caught at above MSY level, so after high initial yields, the yield drops to below that of the other species in the long term. The visible effect of this will be a switch in the catch composition over the lifetime of the fishery. The more efficient the targeting, the more pronounced this effect could be. If the less catchable species is preyed upon by the more catchable one, this species composition change could also be far more marked. As expected from the single species investigations, the magnitude of this trend is also a function of the biological parameters. If the length at first capture Lc is actually lower than the 0.45 baseline, or if there is less density dependence than assumed, then this effect can become much more pronounced, even with a small difference in catchability.

It is interesting to compare the long term adult biomasses that will result when two species that are otherwise the same are fished with different combinations of catchability coefficient. Figure 5.17 shows the steady state biomass ratio for two identical non-interacting species with various degrees of relative catchability. These species have the deep slope baseline parameters. The species are identical, have the same equilibrium population size and do not interact, so where the catchabilities are the same, the resultant biomasses are also the same.

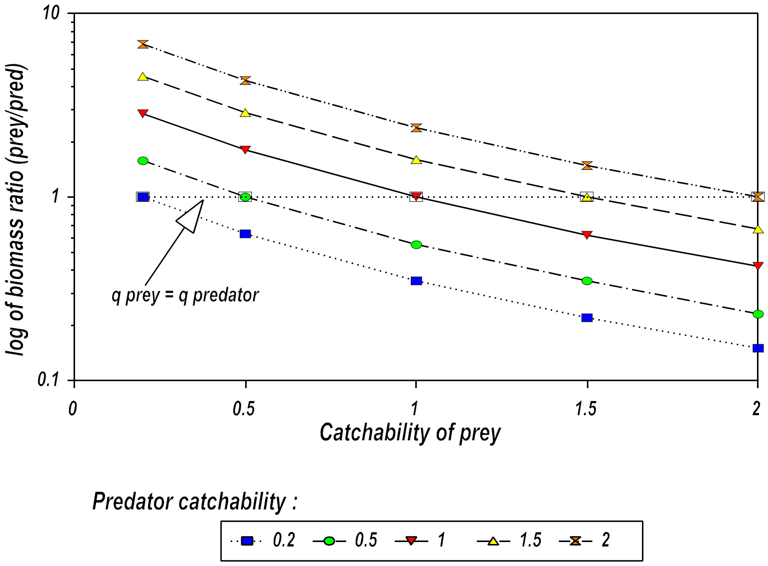

Figure 5.17 : Ratio of adult biomass for two identical, non-interacting baseline deep slope species with various relative catchability coefficients

Now allow predation between the species, of the baseline level and form. Figure 5.18 shows the biomass ratios as before, only this time the ratios have increased, indicating that the prey forms a greater proportion of the population. The dotted line indicating those points where the two species have identical catchabilities has now moved away from 1. This illustrates the effect of the biological interaction, and how that effect increases to an asymptote as catchability increases.

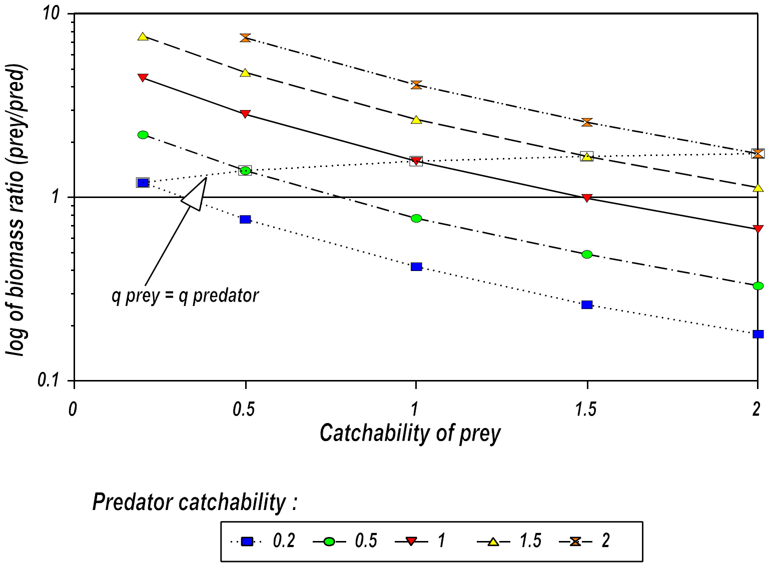

Figure 5.18 : Ratio of adult biomass for two identical, interacting baseline deep slope species with various relative catchability coefficients

By dividing the ratios in figure 5.18 with those of figure 5.18, it is possible to extract those changes in ratio which are due to biological interaction alone. Since the predator biomass is the same in both cases, this quotient is equivalent to the ratio of the prey biomass under predation to the prey biomass as a single species. Figures 5.19, 5.20 and ? depict this ratio for all the previous combinations of catchability, this time for both case study areas.

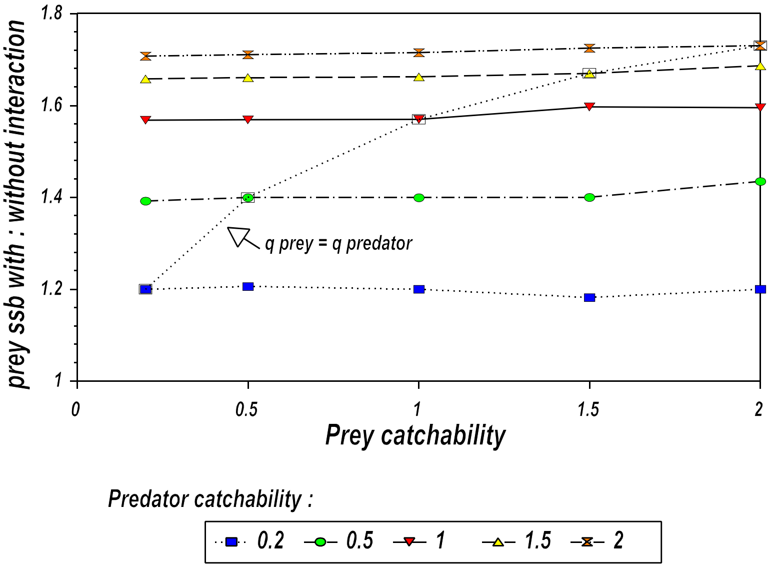

Figure 5.19 : Species ratio with a predation effect as a fraction of the species ratio without a predation effect, for identical deep slope baseline species having various relative catchability coefficients

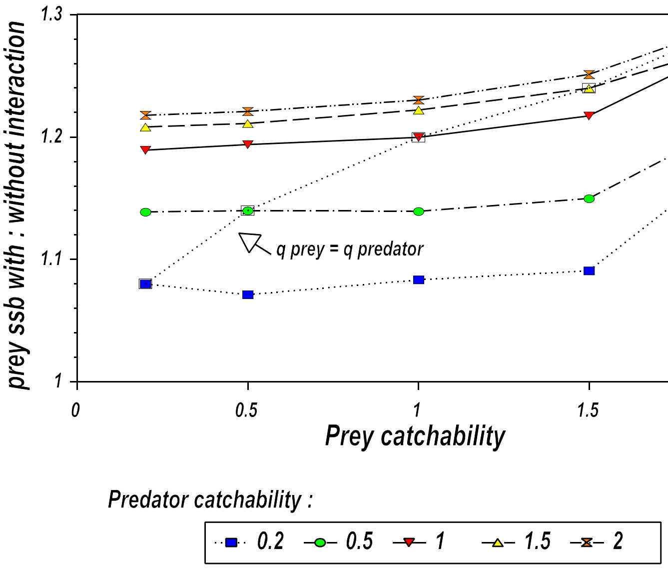

Figure 5.20 : Species ratio with a predation effect as a fraction of the species ratio without a predation effect, for identical banks baseline species having various relative catchability coefficients

Once again, the dotted line indicates the points where the catchability of the two species is the same. At these particular points, the species are truly identical, except for the fact that one preys upon the 'juveniles' of the other. The points show the effect of predation on species ratio, and the line shows how this effect changes as the species become more catchable (this last is also equivalent to increasing overall effort). Considering the rest of the graph, it is not surprising that the effect of predation is much greater when the predator is more catchable but is not influenced by the catchability of the prey. The ratios also do not change greatly at very high predator catchabilities, but these are states where the predator is fished nearly to extinction, and so would not generally be considered desirable.

The axis of figure 5.17is a logarithmic one; the influence of catchability ratio on species composition can be very large. In contrast, the ratios in figure 5.19, 5.20 are relatively small. When the predator catchability is 1 in both cases, the prey catchability must be within 25% of that value for the effect of predation to exceed that of the catchability difference. Where predator catchability deviates from 1, the catchability effect exceeds the pure predation effect in most instances. It is clear that, where targeting is efficient, it will have a far greater influence on the fishery than predation between target species could have.

The previous paragraphs have concentrated on equilibria, i.e. the state that the system will settle to in the longer term. At the beginning of this section, figure 5.16 showed that differences in catchability can cause short term effects that could manifest as, for example, a change in catch composition over the first 10 years of the fishery's existence. In all of the previous calculations, catchability was assumed, for simplicity, to remain constant over the period simulated. However, surprising results can, in principle, be obtained when catchability changes over time.

In Tonga, the depth at which fish were caught and the species that were targeted have changed over the years. Initially, most effort was applied in relatively shallow water, but fishermen have gradually fished at lower depths until now they are largely targeting the deep water species. Also, the favoured geographical locations have changed over the years, and it is possible that species may be more catchable in some locations than in others.

Figures 5.21 and 5.22 show a caricature of such a situation, to serve as an illustration of the type of effect that can be seen. At time=0, an MSY level of effort is being targeted on one species, which has a catchability of 1, while the other species is not caught at all. Over the next 10 years, the catchability of the first species declines linearly to zero, while that of the other increases linearly to 1. The effort remains at MSY level (E=0.3) over the whole period. From year 10 to year 15, the catchabilities remain constant.

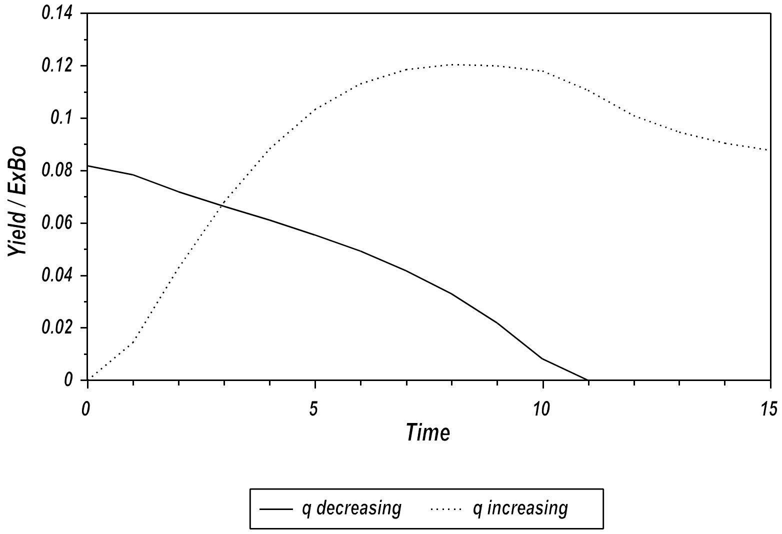

Figure 5.21 : Yield response of identical baseline species to constant fishing effort where the catchabilities vary with time (see text)

The patterns in the figures look remarkably like those one would expect of a predator-prey situation. It would be tempting to assume that, since the effort has remained constant over the whole period, the declining yield of the first species reflects declining abundance. One might then easily assume that the increasing yield (abundance) of the second species is due to prey release resulting from the lower biomass of the first species. This shows that such an assumption would be erroneous. In fact, for these tropical reef and bank fisheries, patterns like this in the data are far more likely to indicate technical interaction than biological.

It is interesting to note that the recovery of the first species is fairly slow. At MSY equilibrium (year 0), the spawning biomass is about 25% of initial levels. By year 10 the species is not being caught at all, the catchability having declined linearly since year 0, yet the adult biomass has only recovered to 50% of virgin levels. It takes another 10 years for the virgin biomass to be restored. Recovery is about twice as fast with banks species, because of their higher mortality and growth rate. It must be borne in mind, though, that these are deterministic simulations. Without knowing the natural recruitment variability of the stock, it is impossible to be certain that it can recover from such high levels of fishing mortality.

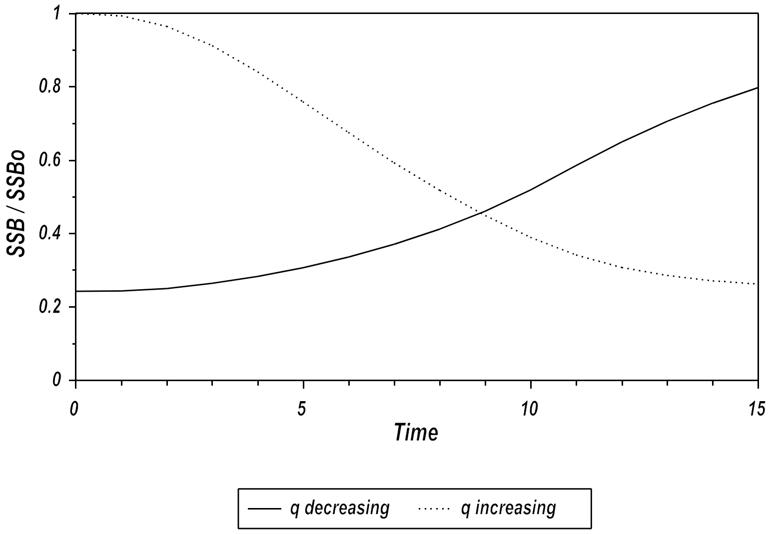

Figure 5.22 : Biomass response of identical baseline species to constant fishing effort where the catchabilities vary with time

As mentioned above, figure 5.21 and 5.22 represent a somewhat exaggerated scenario. To illustrate its applicability to Tongan deep slope species, figure 5.23 shows a pattern of effort and catchability which is roughly similar to that seen in this region. This is not meant to represent the case study scenario, which will be discussed in the next chapter, merely to illustrate possible trends.

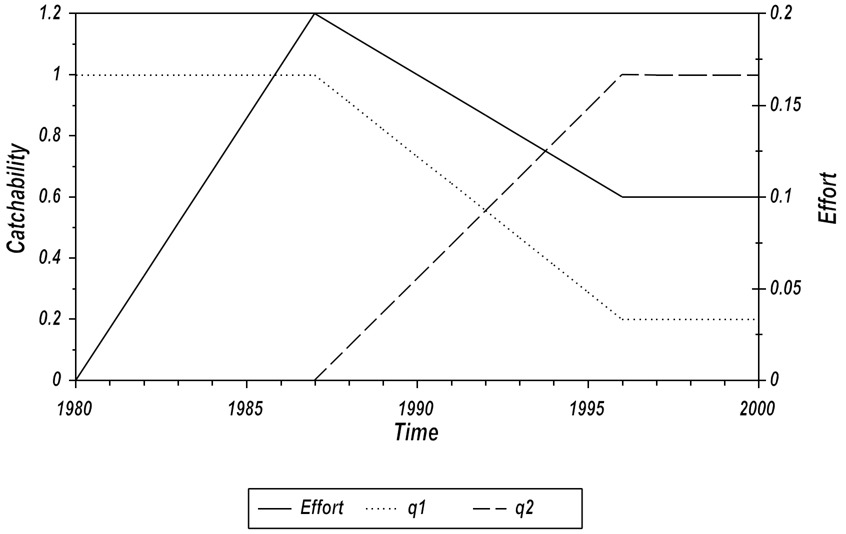

Figure 5.23 : Two identical baseline deep slope species with more realistically varying catchability and effort: effort and catchability inputs

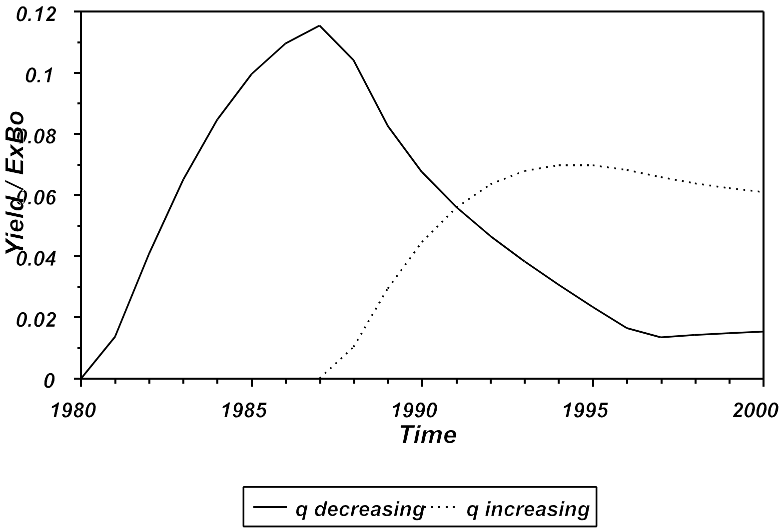

Figure 5.24 : Two identical baseline deep slope species with more realistically varying catchability and effort: yield as a function of time

Figure 5.24 shows the effort and catchabilities applied to two identical deep slope baseline species. Effort increases linearly from zero in 1980 to two thirds of MSY level in 1987, thereafter it declines to one third of MSY by 1996, subsequently remaining at that level (MSY effort corresponds to a catchability of q=1; all other catchabilities are set relative to this amount). For the first seven years, one species has q1=1 and the other q2=0. From 1987 to 1996 the catchabilities respectively increase and decrease linearly to q1=0.2 and q2=1. From 1996 onwards both catchabilities remain constant. Figure 5.23 shows the resulting yield of the two species over time. It is worth bearing this picture in mind while reading the next section on the trends observed in the data.

Banks species can demonstrate similar effects, although they tend to respond faster to changes. Species states also tend to move to greater extremes, for instance at MSY the adult biomass is only 15% of virgin levels.

A last form of technical interaction is due to the fact that gears tend to target certain sizes of fish, without reference to whether the fish reaches that size early or late in its life cycle. If a number of species of different sizes co-exist, then a gear which takes fish of e.g. 40cm may be catching one species at Lc=0.3 and another at Lc=0.6 (as before, Lc is the length at first capture Lc50 expressed as a fraction of L∞). The effect of using a single gear on a multi-species fishery is discussed, with recommendations for size limits for such fisheries, in the latter half of Section 6.3.2.

5.2.6 Conclusions

The major conclusion of this section is that prey release is unlikely to be a significant factor in the fishery. However, technical interactions can have a major impact, and so it is important to have an idea of relative catchability of the species, and how these have changed in time and with geographical area or depth.

A number of biological parameters are not particularly important:
- ogives for maturity and gear selectivity
- length-weight parameters

Mortality and growth can have significant effects, but with these species where M/K is approximately 2, and given the potential variability in M/K, the influence of these parameters is not as great as would otherwise be the case.

Density dependence in recruitment is a significant parameter which is difficult to quantify. In general it is not very important where fishing pressure is moderate, but should be taken into consideration where F exceeds M.

On the whole, technical issues seem to have more significance than biological ones.

In this section, the effects of fishing and of species differences have been characterised by their effects on MSY, F_{msy}, species ratios and transient response to changes in effort. Ideally, it would be more relevant to assess the impact of these changes on management strategy and the results of assessment. New techniques have emerged to do this, and these are the focus of a new FMSP funded project (R6437) for which Tonga will be a case study site.

5.3 Observed effects of fishing at case study locations

5.3.1 Single species effects :

- **Density, biomass and catch rate changes**

Catch rate was used as an index of abundance to assess biomass changes related to fishing for individual species and guilds of species within the multi-species complex of fish caught. Since the focus of this report relates to multispecies fisheries biomass changes of single species and the possibility of biological interactions with other species are discussed fully under multispecies effects (see below). Time series and spatial data were employed.

- **Population demography**

Length based methods were used to derive population demographic parameters (see Chapter 4). Sampling bias and inadequacies affect the characteristics of the data. The length frequency distribution observed is determined by the gear type employed, may vary spatially (by location and depth) and seasonally due to fish movements at any one location. Recruitment variability may also affect the distribution observed. Thus, in order to examine the effects of fishing on population demography it is necessary to eliminate variation due to these factors. However, stratification of available length frequency data by gear type, location (and depth), month and/or year reveals sampling inadequacies and the patchy nature of the data (for example, see data for *P. filamentosus* caught by handlines only by location per month from 1990-1994 in Annex 10, MRAG, 1996a). The value of the data collected from commercial fishing operations is thus limited with respect to addressing the question of fishing effects, and fishery independent data collected according to a well defined sampling strategy is required. Improvements in the sampling strategy for data from the commercial fleet may not resolve the problem, since the patchy nature of the data frequently reflects fishing patterns rather than a lack of appropriate sampling.

Length frequency data was available for certain species from Seychelles and Tonga, but not Mauritius. Biological data suitable for assessment of sex ratio and reproductive characteristics was only available for Seychelles fish (see Section 2.2). Assessments of population parameters have been made using these data (see Chapter 4). However, for the reasons discussed, only limited evaluation of the effects of fishing was possible for mean length and mortality estimates. It was not possible to evaluate the effect of fishing on sex ratio and social structure, reproductive characteristics or growth rate. For the latter, in addition to patchiness of sampled data over space and time, low confidence in parameter estimates meant that comparison of growth rates derived by location and year was not appropriate (see Section 4.7).

Mortality

Total fishing mortality was determined by means of length converted catch curve analysis for which growth parameter estimates are required. Additionally the ratio of Z/K was determined using the Powell-Wetherall method (see Section 4.8). The latter is not dependant upon growth parameter estimates, for which a degree of uncertainty exists. Total mortality tended to increase with fishing effort for both Tongan and Seychelles species, but the relationship was weak and mostly insignificant. Z/K indicated no clear relationship to fishing effort.

For the six principal Tongan species, aggregated, and depth and location stratified data were examined. Estimates of population parameters were derived for the most suitable data for length frequency analysis according to the criteria of Shepherd *et al* (1987. See MRAG, 1994). Regression analyses indicated that total (and fishing) mortality increased significantly with increasing fishing effort for *P. filamentosus* both when growth (K) was allowed to vary (ie. annual estimates of K derived; P=0.025) and when K was fixed over time (P=0.044; Fig. 5.25). Note, however, that K derived by length based methods did not vary significantly with fishing effort

(P=0.404). Similarly, total mortality increased significantly for *P. flavipinnis* (P=0.032, Fig. 5.26), but did not for any other Tongan species studied by depth or location (P>0.05), although for *E. coruscans* at depths greater than 300m (P=0.275, Fig. 5.27) and for *E. morhua* at all depths (P=0.092, Fig. ?) an increasing trend was apparent. No correlation of Z/K was found for Tongan species.

For Seychelles estimates of Z and Z/K were derived for data stratified by fishing bank each year, and for annually aggregated data for *P. filamentosus, A. virescens, L. sebae* and *E. chlorostigma* (MRAG, 1996a). Total fishing effort by all boat categories was standardised per unit area of bank in order to allow comparison of spatial and time series data. No significant increase in total mortality or the ratio Z/K occurred with fishing effort for any species (P>0.10) although an increasing trend was apparent for shallow water species (*Aprion virescens,* Fig. 5.29; *L. sebae*, Fig. 5.30; *E. chlorostigma*, Fig. 5.31). Owing to the difficulty in separating effort directed only at the deeper water species, *P. filamentosus*, mortality estimates for this species were additionally examined in relation to the catch per unit area per annum (Fig. 5.32), but no significant correlation occurred (P>0.10).

Mean Length

Mean length data were computed stratified by year and location for Tonga and Seychelles and additionally by depth for Tonga. Analysis of variance was used to examine statistical differences by location and year and location and depth for Tongan species : *P. filamentosus, P. flavipinnis, E. coruscans, E. carbunculus, E. morhua, E. septemfasciatus* (see MRAG, 1994). No significant change in mean length occurred over time for any species (P>0.10) except for *E. carbunculus* (P=0.03). Significant differences in mean length occurred between locations for *P. filamentosus* (P<0.01), *P. flavipinnis* (P<0.01), *E. coruscans* (P=0.01) and *E. carbunculus* (P=0.04), but not for either of the serranid species (P>0.10). No significant increase in mean length occurred with depth (P>0.05) except for *E. carbunculus* (P<0.01), and regression of fork length on depth indicated no significant change with depth for any species (P>0.05). Bimodal length frequency distributions occurred for *E. carbunculus* and *E. septemfasciatus*. Mean length was greater for fish caught south of 21^{o}S than those caught further north for *P. filamentosus* and *P. flavipinnis*. By contrast, *E. coruscans* caught south of 20^{o}S and *E. carbunculus* south of 21^{o}S were smaller than those from northerly latitudes. Fishing began in the north and moved south over time. However, mean length within locations does not change over time, and north-south differences in mean length by species are not consistent suggesting that the spatial differences observed are not related to fishing pressure. Indeed, fishermen report not only differences in fish length with latitude but also colour and quality of the flesh. These differences are concluded to be unrelated to fishing, and further, for the lutjanid species, suggest the existence of different (overlapping?) populations from north to south.

For the four Seychelles species studied (*P. filamentosus, A. virescens, L. sebae* and *E. chlorostigma*) no consistent relationship between mean length (for handline caught fish only) and fishing effort occurred at any location. Spatial differences were observed but these were not consistent with fishing effort by location. Data were patchy, and none related to the most heavily fished inshore sector of the Mahe Plateau. Aggregating all length frequency data for the whole of the Mahe Plateau to increase sample size did not improve the results. For one species, *A. virescens,* the length distribution changed dramatically in 1994 (Fig. 5.33) and this may have been related to a strong recruitment into the fishery that year. There was no indication of a change of hook size which may have been an alternative explanation.

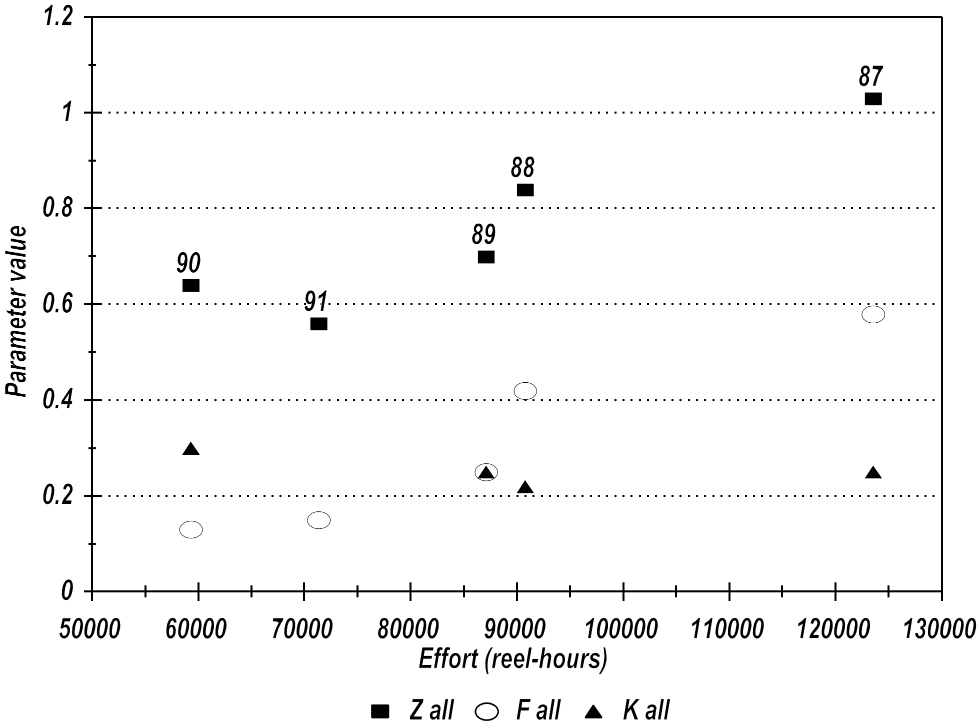

Figure 5.25 : Mortality estimates for *P. filamentosus* from all Tongan locations related to annual fishing effort for those locations, where the estimate of growth (K) varied annually.

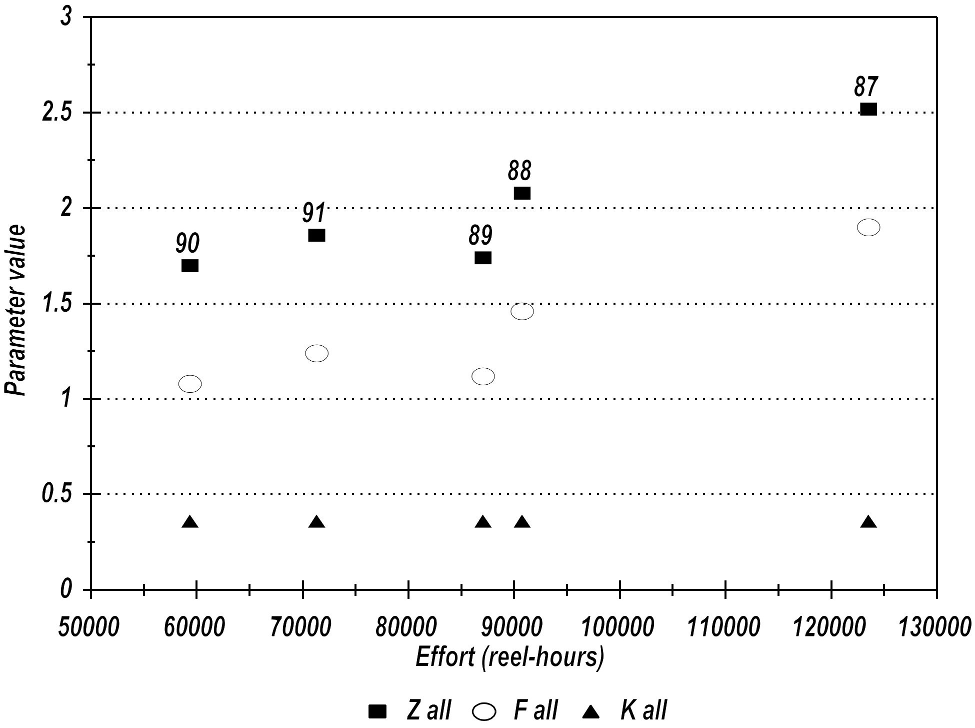

Figure 5.26 :Mortality estimates for *P. flavipinnis* from all Tongan locations related to annual fishing effort for those locations, where the estimate of growth (K) was fixed.

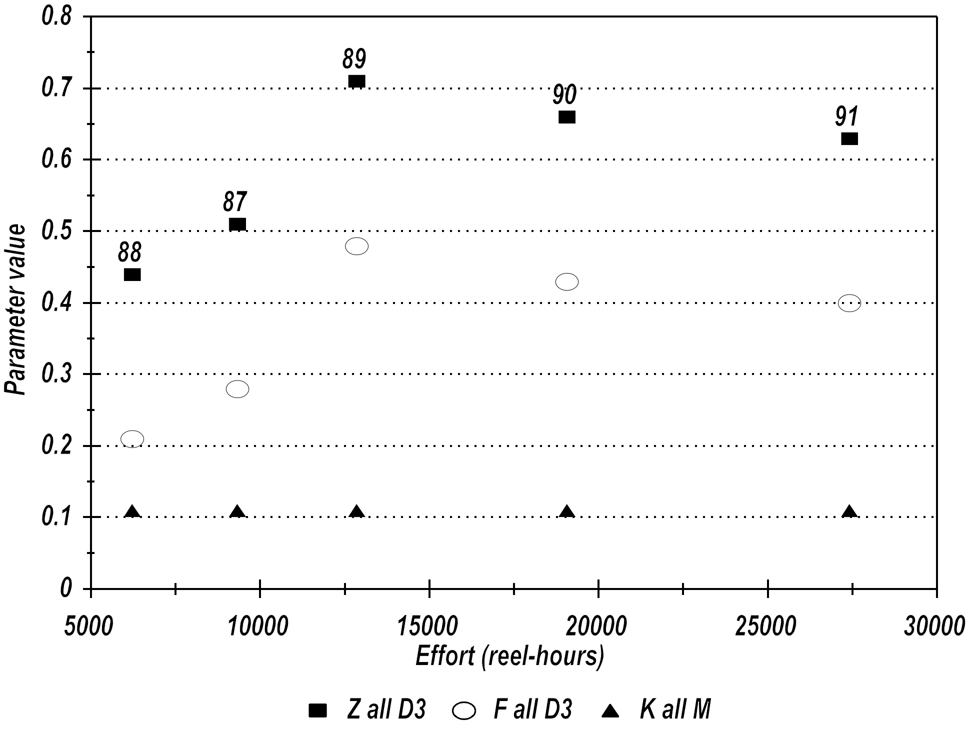

Figure 5.27 : Mortality estimates for *E. coruscans* from all Tongan locations at depths greater than 300m related to annual fishing effort for those locations and depths, where the estimate of growth (K) was fixed.

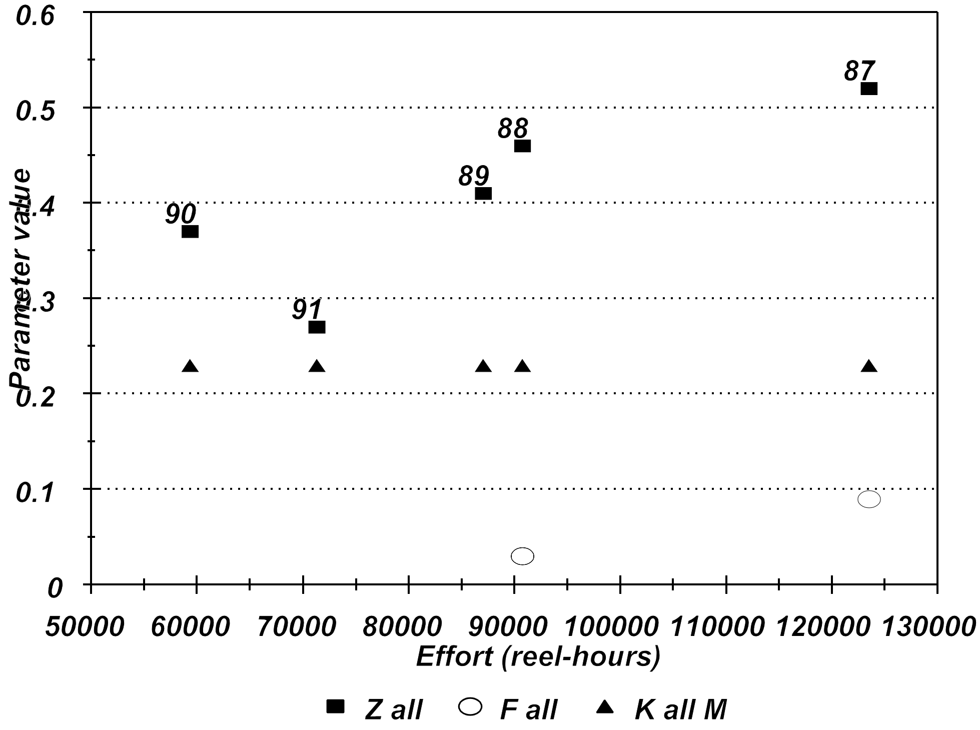

Figure 5.28 : Mortality estimates for *E. morhua* from all Tongan locations related to annual fishing effort for those locations, where the estimate of growth (K) was fixed.

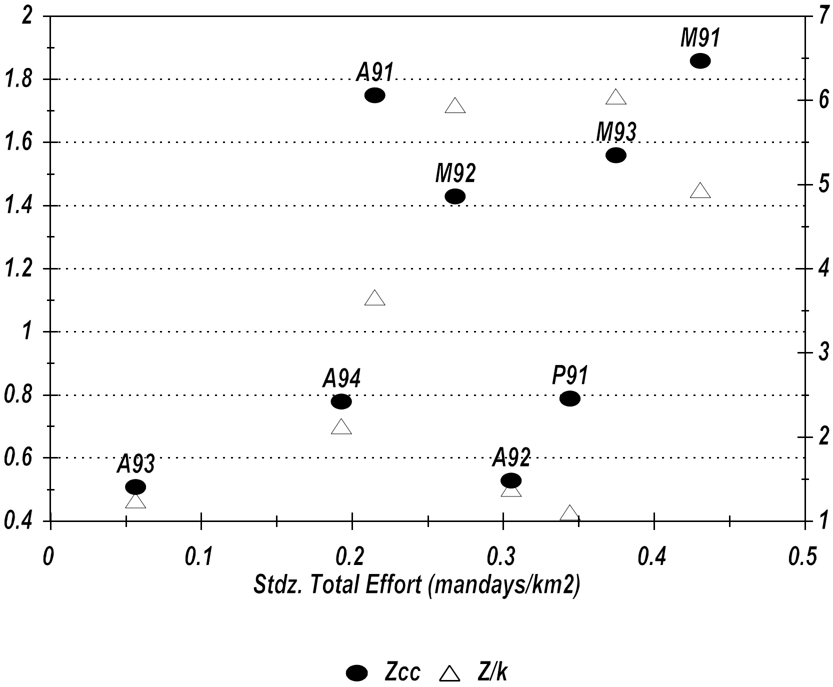

Figure 5.29 : Mortality estimates for *Aprion virescens* by year and Seychelles location (A= Amirantes, B=Banks south of Mahe Plateau, M=Mahe Plateau, P=Providence/ Farquahar) against standardised total effort per square kilometre of bank (Zcc=left axis, Z/K=right axis).

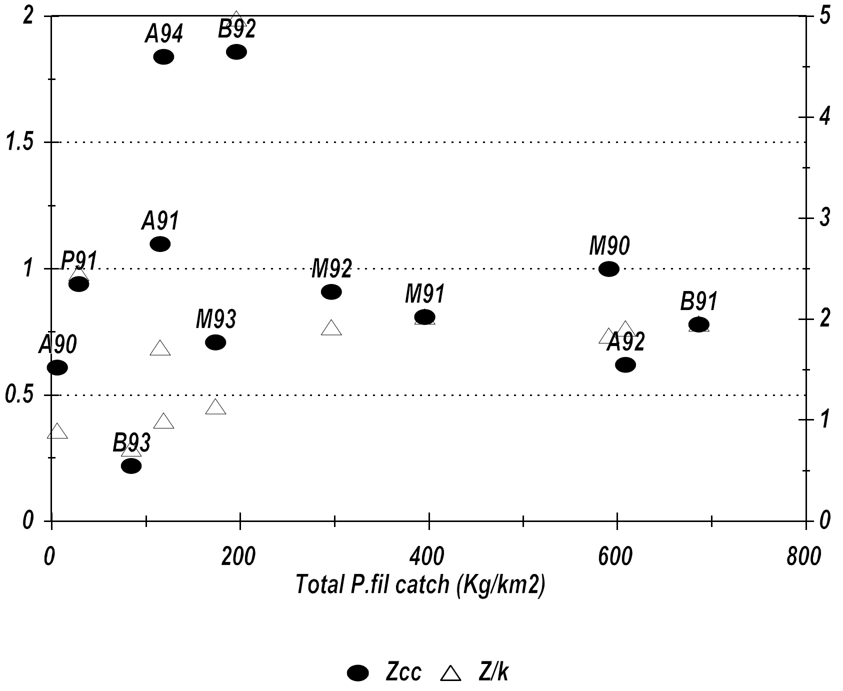

Figure 5.32 : Mortality estimates for Pristipomoides filamentosus by year and Seychelles location (A=Amirantes, B=Banks south of Mahe Plateau, M=Mahe Plateau, P=Providence/ Farquahar) against standardised total catch of that species per square kilometre of bank edge (Zcc=left axis, Z/K=right axis).

Figure 5.31 : Mortality estimates for *Lutjanus sebae* by year and Seychelles location (A= Amirantes, B=Banks south of Mahe Plateau, M=Mahe Plateau, P=Providence/ Farquahar) against standardised total effort per square kilometre of bank. (Zcc=left axis, Z/K=right axis)

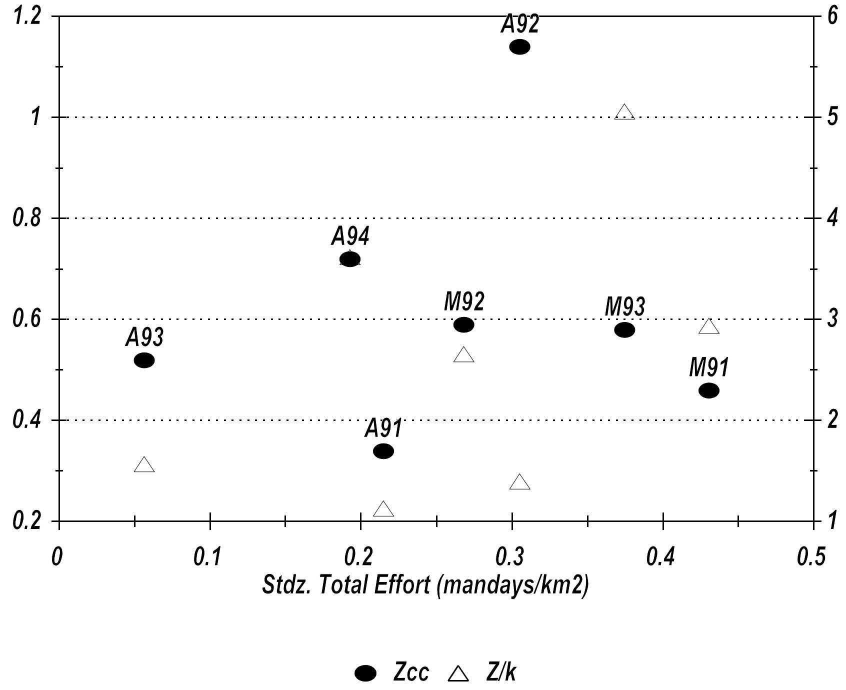

Figure 5.30 : Mortality estimates for *Epinephelus chlorostigma* by year and Seychelles location (A=Amirantes, B=Banks south of Mahe Plateau, M=Mahe Plateau, P=Providence/ Farquahar) against standardised total effort per square kilometre of bank (Zcc=left axis, Z/K=right axis).

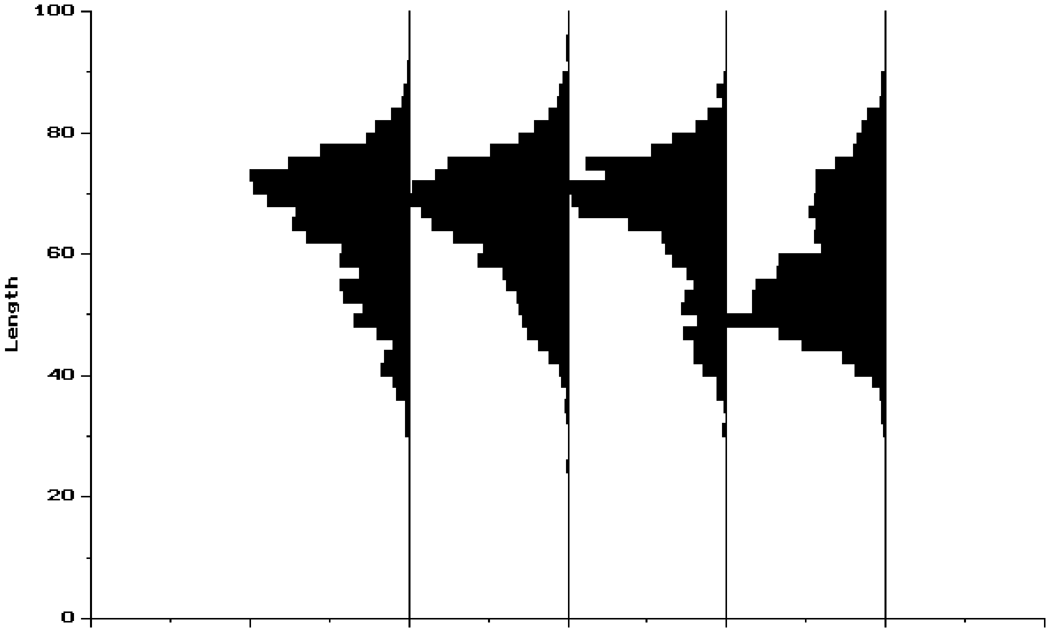

Figure 5.33 : Length frequency distributions observed for *A. virescens* caught on the Mahe Plateau by handlines in 1991, 1992, 1993 and 1994 respectively.

5.3.2 Multispecies effects

Multispecies responses to fishing at Indian Ocean and Tongan offshore reefs (Mees, C C in press. *In* Proceedings of the eighth International Coral Reef Symposium, Panama, 23-29 June 1996) provides a summary of the following section with respect to fishing effects.

■ Tonga

To investigate the effects of fishing, both time series and spatial data were examined for the Tongan deep reef slope fishery. However, bathymetric data, required in order to standardise catch and effort per unit length of 200 m depth contour for spatial comparisons, was limited to a few seamounts. Available information for those seamounts indicated no significant correlation between catch rate and fishing effort (eg. Fig. 5.34). Emphasis was therefore placed on examination of time series data.

Species catch rate data for the Tongan fishery indicated large changes in the relative importance of different species over time (Fig. 5.35). In particular a decrease in the catch rate of *P. filamentosus* was mirrored by an increase in that of *E. coruscans*. However, knowledge of the fishery suggests that the changes observed relate to technological rather than biological interactions (see MRAG, 1994). Most significant is depth related targeting of species in response to external (export) market demands.

A potentially large number of variables may affect catch rates and species composition of the Tongan fishery. These include :

- Seamount or deep reef slope of the banks. Abundance per unit area is less on banks than seamounts (see Section 3.3.1). Banks species may differ from those of the seamounts, and typically include lethrinids. However, this is a function of the depth predominantly fished on banks, and MRAG (1994) indicated that depth was a better indicator of species composition than seamount or bank. Between individual seamounts there exists the possibility of differences in abundance and species composition. Most fishing effort was directed at seamounts, except in 1988 when a greater effort occurred on the banks.

- Depth. Species composition and population abundance is a function of the depth fished (see Section 4.2). Total community abundance may vary with depth. Mean fishing depth on the banks was between 100-150m from 1987-1990, but increased to around 200m in 1991. That on seamounts increased from 200m in 1987 to around 350m in 1991.

- Latitude. Tonga extends from 15°00'-23°30'S. Over this distance it is possible that differences exist in population abundance and hence community structure. The fishery began in the north, but is now predominant in the south.

- Port of origin of fishing boats, and vessel type. New vessels have recently been introduced to the fishery which will have different fishing power. For the earlier data available (1987-1991), essentially a single boat type and fishing method was employed, although skipper and crew related differences may occur. However, vessels fishing from Nuku'alofa in the south tend to be more commercially orientated than those from the north (see Section 2.2.1) affecting both catch rates and species composition through different targeting strategies. A sampling bias exists towards boats from the south.

- Seasonal and annual variation.

Generalised Linear Interactive Modelling (Francis *et al*., 1993) may be employed to derive annual catch rate estimates standardised for, say, seasonal (quarter) and depth variations, as a simple example. The aim is to start from a complex model and reduce it to a simpler one in which the influence of individual factors may be determined. If complex interactions occur within the data it is not possible to simplify the model. A model of the form :

Stdz.cpue = 1+year+qtr+depth+year.qtr+year.depth+qtr.depth+year.qtr.depth

is first tested for interactions between factors of the model, ie. three factor (year.qtr.depth) and two factor (year.qtr, year.depth, qtr.depth) interactions. For example, if year.qtr was not significant, but year.depth and qtr.depth were, this would indicate that a consistent pattern of catch rates occurred seasonally over time (years), but that at depth the pattern of catch rates was inconsistent both seasonally and from year to year. If interactions within the data are not significant, the influence of the individual factors, year, quarter and depth, may be determined. Those factors which are not significant may be eliminated, simplifying the model.

In common with data from Seychelles (see below), available information from the Tongan fishery lacked homogeneity. Spatial and temporal patchiness of all sampled information, let alone by port of origin or individual boat, rendered the data inadequate for standardisation of catch rate for all of the identified variables (within data interactions were highly significant), and stratification of the data was necessary. Seamount data only was examined aggregated over location (latitude, again, sampled data by location were inadequate) for all boats (Fig. 5.36) but two (year.depth, qtr.depth) and three (yr.qtr.depth) factor interactions were significant ($p<0.05$). Eliminating location, seamount data only for southern latitudes (S of 21°S) was examined (Fig. 5.37; Table 5.3). For individual species, few data existed, or were lacking for each month each year by 100m depth band and even when aggregated quarterly and ±200m (Figs 5.38-5.39) data were patchy. Stratification of data by depth (100m bands) and location) eliminated large changes in the relative importance of different species over time and no consistent trends were observed (see MRAG, 1994), but sample size was small requiring the aggregation of the data into 200m depth bands for standardisation.

For the guilds of ‘all-species’ and the ’main 6 species’ two and three factor interactions were not significant. A significant variation in quarterly catch rates occurred, but (surprisingly) not with depth. Despite the apparent decrease in catch rate over time (see also MRAG, 1994), this was not significant. For the main 6 species individually, two or three factor interactions were significant with the exception of *E. septemfasciatus* indicating that further simplification of the model was not valid. Overlooking this, variation in catch rate for all species was significant quarterly (except *E. carbunculus*), by depth and annually. The standardised catch rates (Fig. 5.40) do not indicate large changes in the relative importance of different species over time as was the case for unstandardised data (Figs 5.35-5.39). It must be recognised, however, that within data interactions were overlooked in this standardisation process. For *E. coruscans* in particular, the standard error of the estimate of standardised annual catch rate was high, there were are large number of missing data points, and the estimate of standardised catch rate is lower than might be expected.

Species	GLIM	Model :	y*q*d	y*q*d-y.q.d		1+y+q+d	
or Guild	output	Factor :	y.q.d	y.q+y.d+q.d	q	d	y
	df1		11	19	3	1	4
	df2		653	664	683	683	683
All species	F		1.10	1.38	3.56	0.84	1.95
	P		0.3582	0.1294	0.0140	0.3595	0.1012
	Sig		N	N	Y	N	N
Main 6 Species	F		1.27	1.42	3.53	0.02	0.92
	P		0.2379	0.1083	0.0146	0.8860	0.4530
	Sig		N	N	Y	N	N
P. filamentosus	F		2.35	2.82	3.45	111.00	5.67
	P		0.0076	0.0001	0.0162	<0.0001	0.0002
	Sig		Y	Y	Y	Y	Y
P. flavipinnis	F		0.58	3.67	5.96	49.05	14.92
	P		0.8490	<0.0001	0.0005	<0.0001	<0.0001
	Sig		N	Y	Y	Y	Y
E. carbunculus	F		0.78	1.79	2.33	11.45	3.41
	P		0.6639	0.0211	0.0734	0.0008	0.0090
	Sig		N	Y	N	Y	Y
E. coruscans	F		2.49	3.59	8.28	45.93	9.54
	P		0.0046	<0.0001	<0.0001	<0.0001	<0.0001
	Sig		Y	Y	Y	Y	Y
E. morhua	F		0.44	2.15	10.90	24.68	30.50
	P		0.9375	0.0031	<0.0001	<0.0001	<0.0001
	Sig		N	Y	Y	Y	Y
E. septemfasciatus	F		0.03	0.91	13.70	17.31	9.06
	P		1.0000	0.5670	<0.0001	<0.0001	<0.0001
	Sig		N	N	Y	Y	Y

Table 5.3: Results of GLIM analyses of catch and effort data for Tongan seamounts south of latitude 21^{o}S for annual, quarterly and depth variation in catch rates.

Standardised data indicated no significant change in biomass (CPUE) of guilds of all species or the main 6 species over time for seamount aggregated data. However, catch rate and community composition changes may be masked by sequential fishing of seamounts (see Chapter 3). Insufficient data by individual seamount was available to include this as a factor in the GLIM analysis. Eighteen of the most frequently fished seamounts were thus examined individually. From unstandardised data :

- only two (seamounts '1320' and '1403', see MRAG, 1994 for codes) indicated the possibility of depletion, but the catch at these locations was predominantly monospecific (*E. coruscans* at 1320, *P. filamentosus* at 1403) so multispecies effects could not be investigated. GLIM analysis indicated that multiple factor interactions were not significant for the principal species at either location (p>0.1). For *E. coruscans* at seamount 1320 catch rate did not vary significantly with quarter, depth or year indicating that depletion had not occurred (p>0.05). For *P. filamentosus* at seamount 1403, quarterly variation was not significant (p=0.4055) but depth and year were (p<0.0001) indicating that significant depletion had occurred (see Mees and Rossouw, 1995).

- Two indicated similar changes in community composition to that observed for all seamount locations (1501, 1504 : decrease of *P. filamentosus* and increase of *E. coruscans*). However, total sample size was small and inadequate for detailed analysis using GLIM. The standard error of standardised outputs was high. Knowledge of fishing depth however, suggested that the changes were depth related.

- No other seamounts indicated consistent changes in species composition.

The analyses of seamount aggregated data indicated that changes in the relative abundance of target species have not occurred over time in relation to fishing. The apparent changes indicated in unstandardised and unstratified data may be explained by technical interactions and do not appear to be a biological response to fishing. Analysis of individual seamount data provides little improvement. Significant biomass change could only be demonstrated for one species at one location. Community responses to fishing could not be statistically explained, and stratification of the data offered the best indication. The limitations of data collected from the commercial fishery to adequately explain multispecies fishery effects are highlighted.

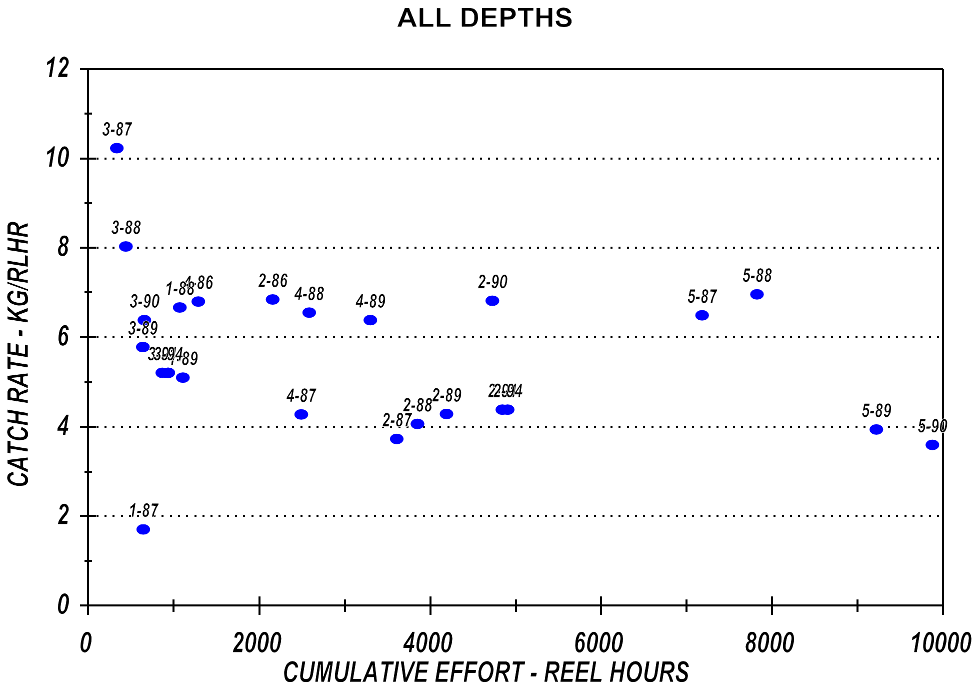

Figure 5.34 : Demersal species catch rate by year and seamount location (1-5, seamounts 1301, 1401,1403,1501 and 1504 respectively, see MRAG 1994, for years 86-94, indicated) against cumulative fishing effort per nautical mile of 200m isobath at each seamount.

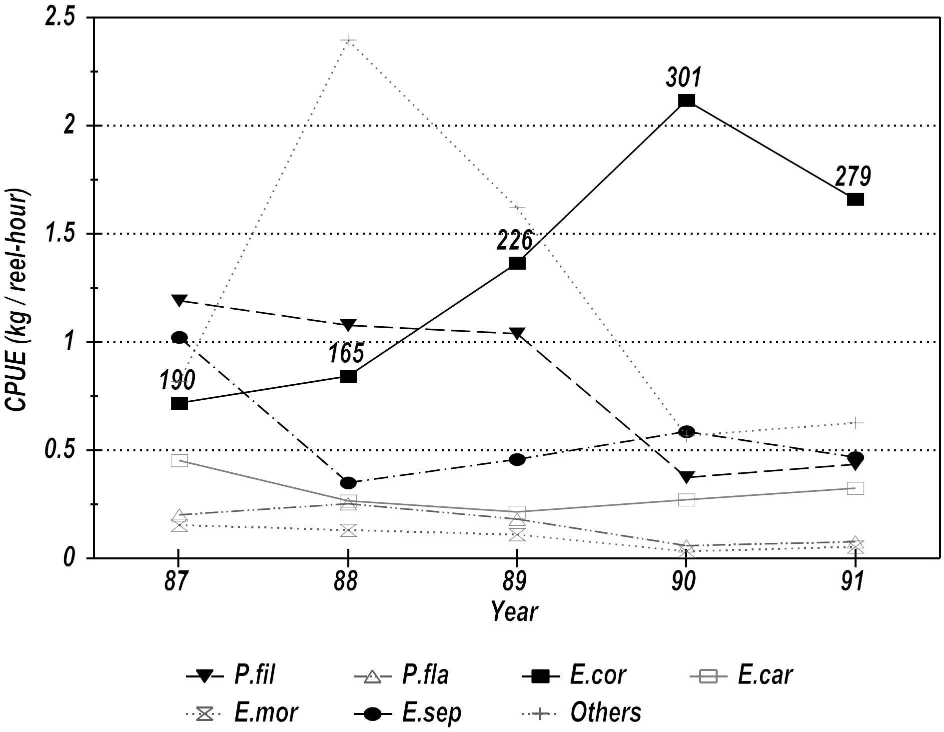

Figure 5.35 : Unstandardised catch rates observed for all sampled data (boat, depth, latitude, mount/bank information aggregated) for the main six species and a guild of all others caught in the Tongan fishery. Mean fishing depth indicated.

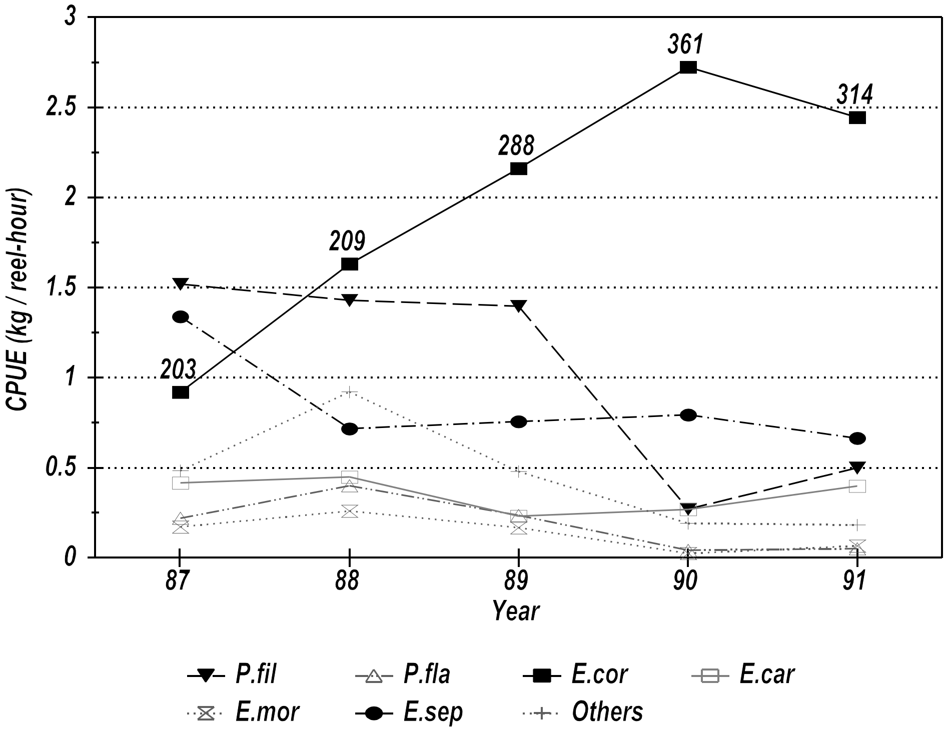

Figure 5.36 : Unstandardised catch rates observed for data sampled from seamounts only (boat, depth, latitude information aggregated) for the main six species and a guild of all others caught in the Tongan fishery. Mean fishing depth indicated.

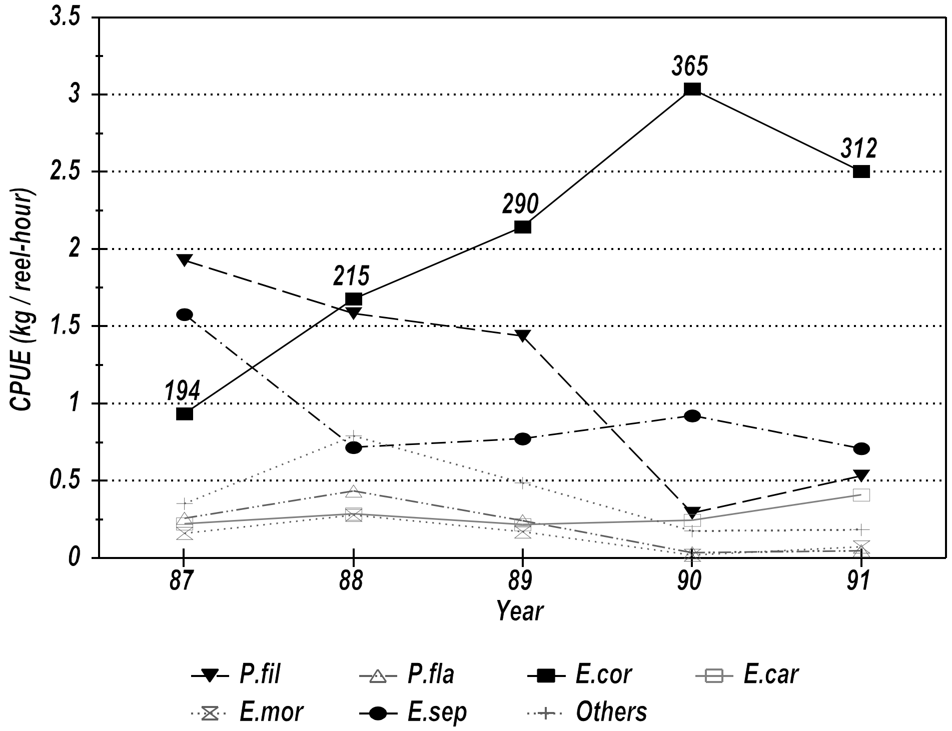

Figure 5.38 : Unstandardised catch rates observed for data sampled from seamounts south of latitude 21°S (boat, depth information aggregated) for the main six species and a guild of all others caught in the Tongan fishery. Mean fishing depth is indicated.

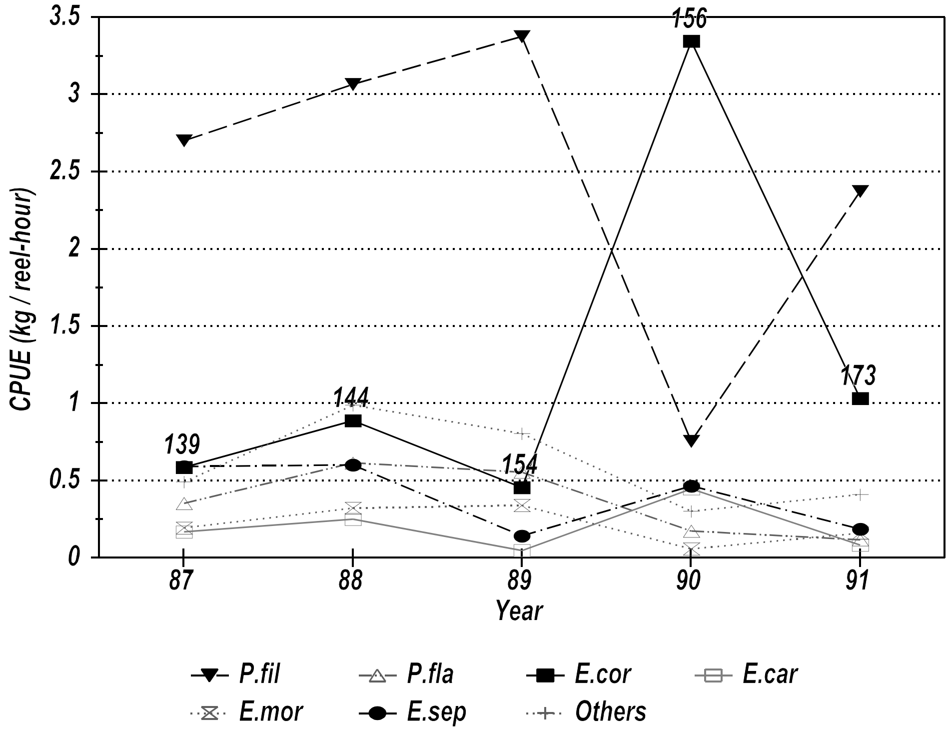

Figure 5.37 : Unstandardised catch rates observed for data sampled from seamounts south of 21°S in the 0-200m depth band (boat information aggregated) for the main six species and a guild of all others caught in the Tongan fishery. Mean depth is indicated.

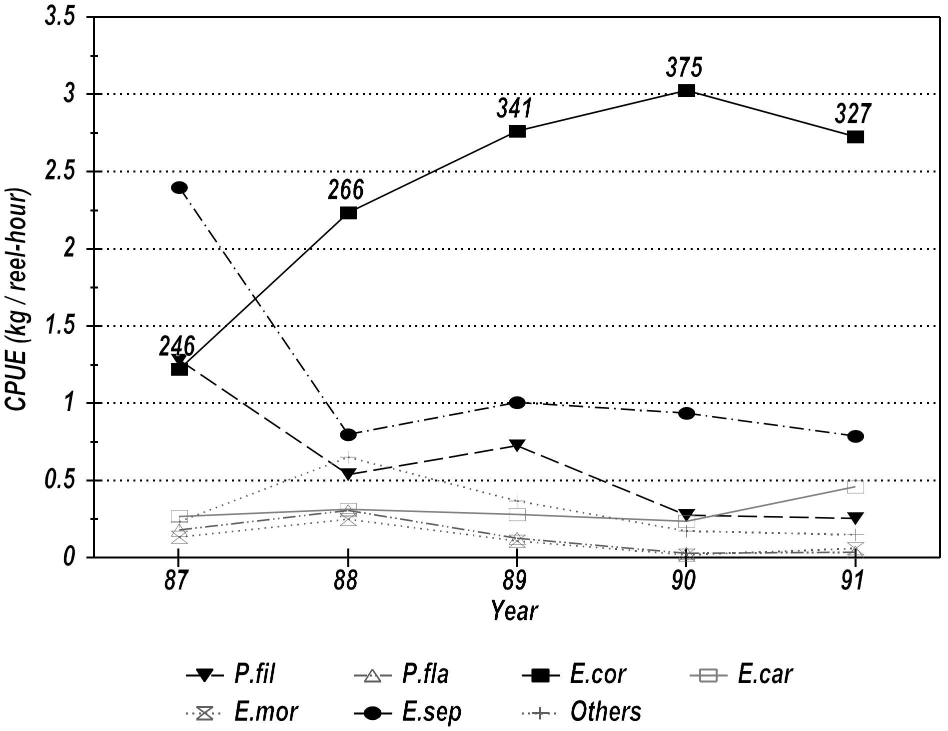

Figure 5.39 : Unstandardised catch rates observed for data sampled from seamounts south of 21°S at depths greater than 200m (boat information aggregated) for the main six species and a guild of all others caught in the Tongan fishery. Mean depth is indicated.

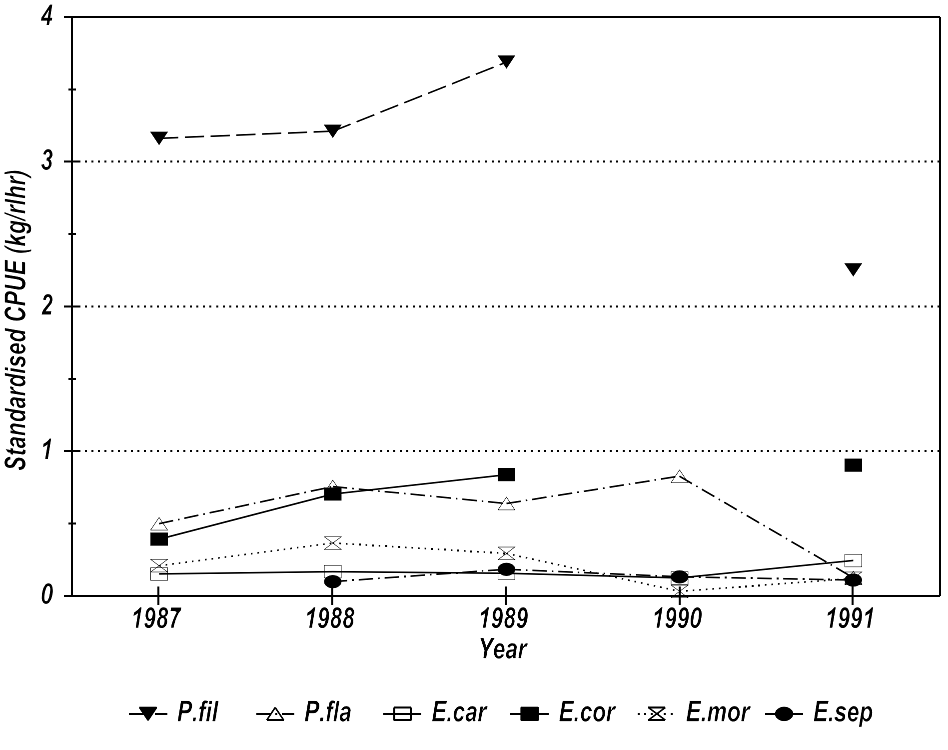

Figure 5.40 : GLIM standardised catch rates for the six main species caught from Tongan seamounts south of 21°S. Missing data points are those where the standard error was greater than the estimate of standardised catch rates.

- **Seychelles and Mauritius**

Both time series and spatial data were examined for effects of fishing with respect to biomass and species composition changes in Seychelles. Only spatial data was examined for Mauritius. Historical data from the literature indicated depletion of banks fishery resources over time in Mauritius (Fig 5.41), but a sufficient time series of data was not available to the project to test this. Data were not available to test for multispecies effects of fishing (see 2.2.3) but the fishery on the Mauritian Banks is largely monospecific (see 3.2.2).

Examination of Seychelles gross catch composition (proportion by family guild) for all boat types indicated considerable fluctuation at most locations over time. Lutjanids represented approximately 60% of the catch and lethrinids and serranids 20% each. Apparent changes in the family composition of the catch over time were indicated for a number of locations (statistical sectors, see Fig. 3.3) including the inshore (sector 1) area and north eastern (sector 3) and southern edges (sector 7) of the Mahe Plateau, and for the Amirantes Plateau (see MRAG, 1996a) : the proportion of serranids apparently declined in the catch whilst that of lutjanids increased. Spatial analyses suggested that the abundance of serranids decreased with increasing fishing pressure. However, this crude representation of the data does not fully explain the observations since it does not account for variation in the data. There remains the possibility that observed changes are technologically or environmentally related for time series and spatial data respectively, rather than a biological response to fishing. The composition of the catch by boat category varies (Fig 5.42), and the boat categories exploiting a particular area vary over time. For example, basic fleet schooners accounted for around 40% of the catch from the southern edge of the Mahe Plateau during the late 1980's but less than 1% of the catch by 1994, whilst schooners with electric reels and introduced whaler designs (the Lekonomi and Lavenir) took an increasing part of the catch.

To investigate multispecies effects it was therefore necessary to eliminate variability in the data. Available data are very ‘patchy’, and sampled information for all boat-gear categories at each location, season and depth do not exist:

- Seychelles demersal fishery is exploited by a number of boat types (see Section 2.2.1). In total 17 different boat-gear combinations were identified. Within boat-gear categories changes in vessel and gear technology have resulted in increasing fishing power. One category, Mothership-dory ventures with handlines, was equivalent to the Mauritian banks fishing method.

- Strong seasonal variation in catch rates occurs related to the prevailing climatic conditions. Catch rates are higher during the calm inter-monsoon periods centred on April and October in Seychelles, which are also periods of peak reproductive activity for some of the target species. Regional variations occur, with fishing on the southern Mauritian banks being restricted to the period between September and early June.

- Fishing depth affects catch rates and species composition. Fish are either targeted on the banks (shallow water <75m) or on the drop off at the periphery of the banks (>75m).

- Within gross fishing locations which are quite large, discrete areas of fishable habitat may exist (see Section 2.2.1).

Catch rates are an index of resource abundance (see Section 5.1.2). Meaningful standardisation of all available catch and effort data in order to explore multispecies effects of fishing was not a viable option. It was therefore necessary to examine only data relating to the two most common boat-gear categories in Seychelles :

- the basic fleet of schooners for offshore fishing locations,

- traditional whalers targeting demersal species (demersal catch >84% of total) for the inshore area of the Mahe Plateau (see Section 3.2.2),

each fishing with handlines only, and for spatial comparisons including the Mauritian Banks:

- handline fishing from mothership dory ventures

Stratifying the data in this way eliminated variation due to changes in boat category. Schooner data was available for the period 1983-1994. Accurate species composition data for whalers was only available from July 1989 (see Table 2.2). Data for the period 1991-1994 was available for the mothership-dory fishery. It will be recalled that with the exception of the southern edge of the Mahe Plateau (sector 7), all offshore locations in Seychelles were lightly exploited. The inshore (sector 1) of the Mahe Plateau was the most heavily exploited (see Fig 3.3). Except for the Chagos Archipelago, banks fished by Mauritian vessels are heavily fished.

■ *Time series data for basic fleet schooners on Seychelles offshore banks*

For most offshore fishing banks the stratified schooner data indicated no evidence of depletion (decreasing catch rate) of the guild of all demersal species, or any consistent changes in the relative abundance of species in the catch between 1983 and 1993 (MRAG, 1996a). The only exception was *E. chlorostigma* which apparently decreased in abundance at a number of locations on the Mahe Plateau (Fig 5.43) although the family guild, serranids, did not decline. Species and family guild composition did, however, fluctuate over the period. At a number of locations, demersal catch rates increased, related to a consistent increase in the catch rate of lutjanids (eg. the southern edge of the Mahe Plateau, Fig 5.44). Changes in fleet composition with older vessels leaving the fishery, and improvements in fishing technology (use of satellite navigation, for example) may partly explain observed catch rate increases. Not all vessels had fished consistently over the 10 year period, and insufficient data on technological changes were available to enable standardisation of effort for these possible effects. However, the main factor contributing to increased catch rates was believed to be a shift of fishing effort to deeper water in order to target *P. filamentosus* which was important in catches from around 1990 onwards (Fig. 5.45). *P. filamentosus* had not been, or only lightly exploited prior to this time and catches and catch rates were therefore high. This is well documented for other vessels in the schooner fleet (see Mees, 1990a). The shift in target species may also explain the apparent decline of *E. chlorostigma,* but depletion cannot be discounted.

Fishing effort at offshore locations in Seychelles is mostly light. Evidence for biological interactions and changes in the relative abundance of species in the catch due to fishing was not found, except possibly for a decrease in abundance of *E. chlorostigma*. Observed effects in the data were all explained by technological interactions. Gross species composition changes were the result of changes in exploitation patterns by the vessels in the fleet. Stratified data for basic fleet schooners suggested an increase in the abundance of lutjanids (Fig. 5.44), but this was explained by a change in target species to the deeper water *P. filamentosus* (Fig. 5.45).

- ***Time series data for traditional whalers at the inshore sector of the Mahe Plateau, Seychelles***

The greatest fishing effort related to the inshore sector of the Mahe Plateau exploited principally by whalers. A generalised linear interactive model (GLIM4, Francis *et al*., 1993) was employed to standardise catch rate data for traditional whalers targeting demersal species. Fishing occurs in shallow water in this sector, and the model investigated only annual and seasonal variation in the catch rates of a guild of all demersal species (Fig 5.46), family guilds (Fig. 5.48) and the principal species in the catch (Figs. 5.47 - 5.50). Catch rates fluctuated considerably, and no consistent trends over time were indicated.

Seasonal variation in catch rates was significant for all species and guilds examined, with the exception of the guilds of ‘all' and ‘other' lethrinids (Table 5.4). Despite significant annual variation in the catch rate of principal lethrinid species and the guild ‘other' serranids, no significant annual variation occurred for the family guilds lethrinidae and serranidae. The significant annual variation in ‘All demersal species' catch rates was thus explained by variation in the guild lutjanidae, and in particular by highly significant fluctuations in the catch rate of *A. virescens* (Table 5.4).

SPECIES	FACTOR	F	df1	df2	P	SIGNIFICANT
All demersal spp	season	20.42	3	270	<0.00001	Highly
	year	12.69	5	270	<0.00001	Highly
All lutjanids	season	22.31	3	270	<0.00001	Highly
	year	13.58	5	270	<0.00001	Highly
All serranids	season	3.947	3	270	0.0088	Yes
	year	1.633	5	270	0.1514	No
All lethrinids	season	0.9089	3	270	0.4372	No
	year	1.156	5	270	0.3312	No
L. sebae	season	10.03	3	270	<0.00001	Highly
	year	1.72	5	270	0.1313	No
A. virescens	season	21.93	3	270	<0.00001	Highly
	year	20.10	5	270	<0.00001	Highly
Lutjanidae	season	4.73	3	270	0.0031	Yes
	year	4.59	5	270	0.0005	Yes
E. chlorostigma	season	6.53	3	270	0.0003	Yes
	year	0.33	5	270	0.8970	No
Serranidae	season	11.28	3	270	<0.00001	Highly
	year	11.15	5	270	<0.00001	Highly
G. grandoculis	season	5.34	3	270	0.0014	Yes
	year	8.11	5	5	<0.00001	Highly
L. microdon	season	5.002	3	270	0.0022	Yes
	year	2.75	5	270	0.0192	Yes
L. crocineus	season	10.17	3	270	<0.00001	Highly
	year	5.419	5	270	0.0001	Yes
Lethrinidae	season	2.324	3	270	0.0753	No
	year	1.188	5	270	0.3154	No

Table 5.4 : Results of the GLIM analysis of species and guild catch rate data for traditional whalers targeting demersal species in the inshore sector of the Mahe Plateau. The analysis was used to standardise annual catch rates and test the significance of annual and seasonal variation (model=year+season).

Thus, despite the high level of fishing pressure at the 'inshore' area of Mahe Plateau (equivalent to that on the Mauritian banks per unit area) the data do not indicate depletion or consistent multispecies effects over the period 1989-1994, but suggest that significant fluctuations in biomass occurred in that period, both for individual species and guilds of species. Similar large fluctuations in the abundance of A. virescens in Hawaii have been observed (Parrish, Pers Comm). Standardised family guild catch rates suggested no change in abundance of serranids and lethrinids with increasing total fishing effort in sector 1 (all boats), but possibly a reduction in the abundance of lutjanids (Fig. 5.51). None of the data were adequate for stock assessment using production models.

Species composition and catch rates are a function of prior fishing activity which is known to have occurred for a considerable period in this location. The present catch rates and species composition may differ from that originally, but the time series was insufficient to detect these changes. Possible explanations for the observed variation in catch rates and biomass are that at the high level of fishing pressure in sector 1 fish stocks are in an unstable equilibrium. Due to the loss of the buffering of any natural variation in recruitment biomass fluctuations occur (see 5.1.2), Length frequency data for *A. virescens* indicate that in 1994 the size structure of the population was different from previous years, and a strong recruitment of smaller fish is suggested that year (Fig 5.33). Biomass increases in 1990 and 1994 may thus be environmentally related. Next, due to market demands, traditional whalers are increasingly being fitted with echo sounders to better target high value demersal species, trip length has increased over time and night fishing is more common. Thus technological changes may also be significant, but data are lacking to test this.

- ***Spatial analyses for mothership-dory fishing ventures at Indian Ocean Banks.***

Total annual demersal fish catch and effort were determined for 9 Indian Ocean banks (Table 3.1). To determine the total fishing effort at each of the banks, the relative fishing power of each vessel (boat-gear type) was determined (See MRAG, 1996a;b) and used to standardise estimates of effort by boat category. However, due to the variability in the data between boat categories, only mothership-dory catch rate information was used as an index of abundance to compare demersal resources between locations. Data were standardised for depth and season fished using a generalised linear interactive model (GLIM4, Francis et al 1993). Effort was standardised relative to one man-day fishing by handline from a dory in the depth range 0-75 m during the period of the SE Trade Winds (model: boat, season and depth). The relative fishing power of the Seychelles mother ship was assumed to be equivalent to that of the average Mauritian vessel. Additionally, standardised (boat and season) annual catch rates were determined by location for data stratified by depth band (shallow, < 75 m; intermediate > 74 m). Analysis by depth band and family-guild was restricted to locations 1-5 (Seychelles) and 9 (Chagos) due to the limited depth and species information available for Mauritius. For analyses relating to the shallow depth band and to lethrinids, all locations (1-9) were included.

Standardised annual catch rate data for guilds of all demersal species, lutjanids, serranids and lethrinids by location were correlated with prior fishing history (measured as the mean annual catch and standardised effort per square kilometre of substrate) and with various environmental parameters. By a stepwise iterative process, multiple regression of catch rate for the guild 'All demersal species' by location at depths less than 75 m against fishing and environmental parameters resulted in the sub-model :

cpue=constant+mean annual effort +substrate

Single linear regression of catch rate against each parameter indicated that the relationship to mean annual catch and effort, winter primary productivity and silicates was significant at the 5% level, although the fit was poor (low R^2). The strongest relationships were related to fishing pressure. However, the relationship of catch rate to prior fishing was not linear and semi-logarithmic transformation of the data improved the fit (see MRAG, 1996a;b). Subsequent comparisons therefore examined only the relationship between the natural logarithm of catch rate and fishing pressure, equivalent to a Fox production model (Table 5.5). However, in explaining resulting observations, the possibility of environmental influences cannot be discounted.

GUILD CPUE	DEPTH	CONSTANT	COEFFICIENT	STD . ERROR	T	P(2 TAIL)	R2	SIG.
All demersal	All	4.51	-2.527	0.664	-3.81	0.003	0.55	Y
Lutjanids	All	3.66	-1.937	1.514	-1.28	0.225	0.12	N
Serranids	All	3.19	-3.871	1.198	-3.23	0.007	0.47	Y
Lethrinids	All	3.45	-2.818	1.543	-1.83	0.093	0.22	N
All Demersal	<75 m	4.41	-2.371	0.890	-2.67	0.021	0.37	Y
Lutjanids	< 75m	3.48	-2.913	1.581	-1.84	0.090	0.22	N
Serranids	<75m	2.52	-2.149	2.255	-0.95	0.359	0.07	N
Lethrinids	<75m	3.26	-1.750	1.830	-0.96	0.358	0.07	N
All demersal	> 74m	4.25	-0.010	0.020	-0.47	0.648	0.02	N
Lutjanids	> 74m	3.44	0.023	0.034	0.67	0.513	0.04	N
Serranids	> 74m	2.82	-0.057	0.023	-2.49	0.032	0.33	Y
Lethrinids	> 74m	2.66	-0.042	0.036	-1.15	0.273	0.10	N

Table 5.5 :Results of the regression of the natural logarithm of guild catch rate against mean annual fishing effort per square kilometre at all Seychelles and Chagos banks.

These analyses show only weak correlations of abundance (cpue) relative to fishing pressure and environmental parameters. They indicate that :

- ‘All demersal species’ catch rates are inversely proportional to the fishing effort per unit area per annum at all-depths (Fig. 5.52) and in shallow water, but no significant trend was observed at intermediate depths (>74m). Catch rates observed for the Chagos Archipelago (9) are less than might be expected given the annual effort and yield removed suggesting a lower abundance of demersal species (see MRAG, 1996a;b). Closer, examination indicates that this relates principally to the greater depth band, and to the guild ‘lutjanids’.;

- as fishing pressure increases, the serranidae decline in abundance at all-depths (Fig. 5.53) and in the deep water stratum (>74m), but not in shallow water.

- no significant changes in the abundance of lutjanidae (Fig. 5.54) or lethrinidae (Fig. 5.55) occur at any depth related to fishing effort.

Spatial comparisons require that environmental variation must be accounted for. Whilst present catch rates showed the strongest correlation to prior fishing catch and effort, winter primary productivity and substrate type were also significant. The latter was based on scanty data and may be an artefact, but nevertheless this limits the applicability of the spatial approach to examine fishing effects. In particular, the change in composition of serranids, whilst apparently correlated to fishing pressure, may in fact be related to local environmental conditions. Species composition is known to vary with latitude, and it has been noted that the abundance of lutjanids and serranids declines at southerly latitudes (see Chapter 3). The Mauritian banks were not included in the analysis for this guild, which examined only the northerly and multispecies Seychelles and Chagos banks. Nevertheless, the apparent decline in abundance of serranids may in fact be environmentally related and more detailed habitat and oceanographic data specific to the banks is required to test this further.

■ Summary

Despite some problems, which limit the usefulness of commercial data for assessing the effects of fishing :

- lack of fine definition in spatial data;
- lack of homogeneity (spatial and temporal patchiness in commercial data);
- the number of variables that need to be accounted for (climatic and seasonal events, technological changes, fishing depth, night vs day fishing) and small sample sizes due to the scattered nature of the information meaning that data were inadequate to obtain statistically valid results for the lightly fished locations;
- short time series of data (especially for the heavily fished Seychelles sector 1 where it is probable that any changes to the species composition in this location occurred prior to the data available);

locations with sufficient data were identified to enable the effects of fishing to be statistically tested. In summary, long term time series of data for Tonga and Seychelles did not indicate depletion or multi-species effects which could not be attributed to environmental or technological factors. No prey release was observed. Species composition changes that did occur related to technological interactions. Spatial data indicated that depletion of serranids had occurred due to fishing, although improved information relating to environmental parameters is required to verify this. Depletion of serranids was also implied from time series data.

The lack of a response to fishing is an agreement with the predicted effects of fishing (Section 5.2.4) which indicated that any prey release effects would be small, and undetectable due to variation in typically available data. Depletion of serranids was consistent with the removal of top predators.

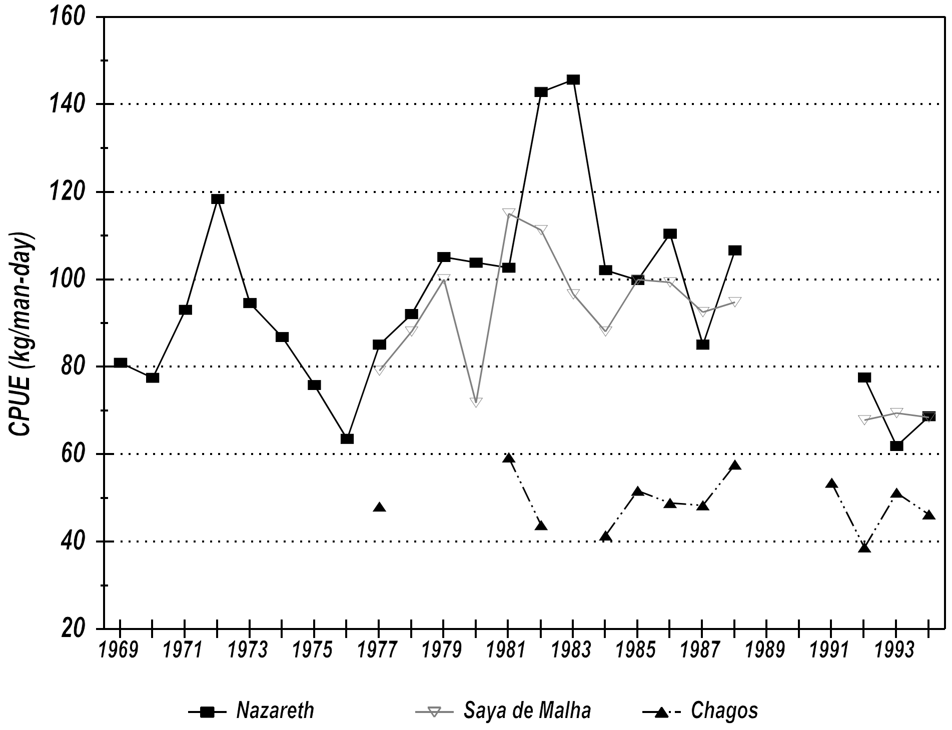

Figure 5.41 : Annual catch rates (all species) for Mauritian mothership-dories fishing on the Saya de Malha, Nazareth and Indian Ocean banks.

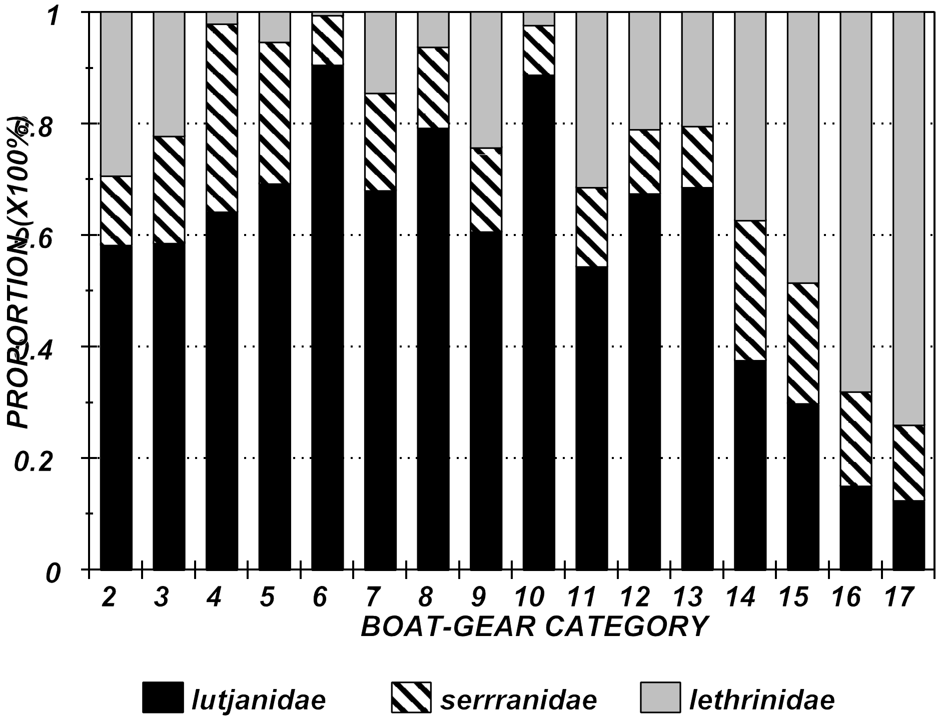

Figure 5.42: The average family guild composition of catches by boat-gear category from the inshore sector of the Mahe Plateau, 1985-1994

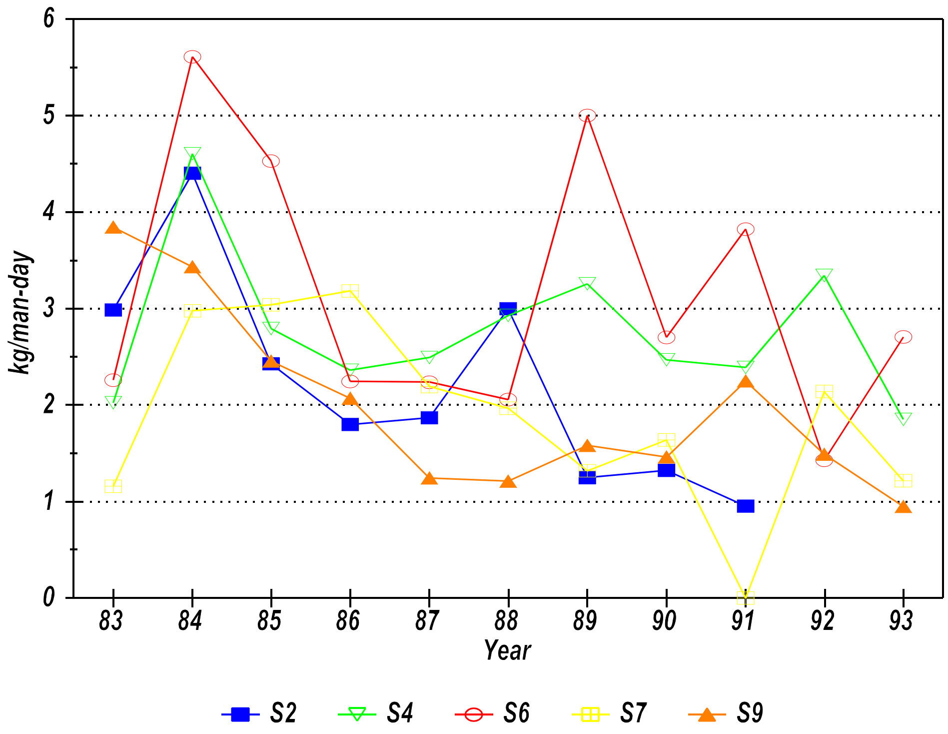

Figure 5.43 : Catch rates for *E. chlorostigma* caught by basic fleet schooners with handlines from a number of different locations (sectors) in Seychelles

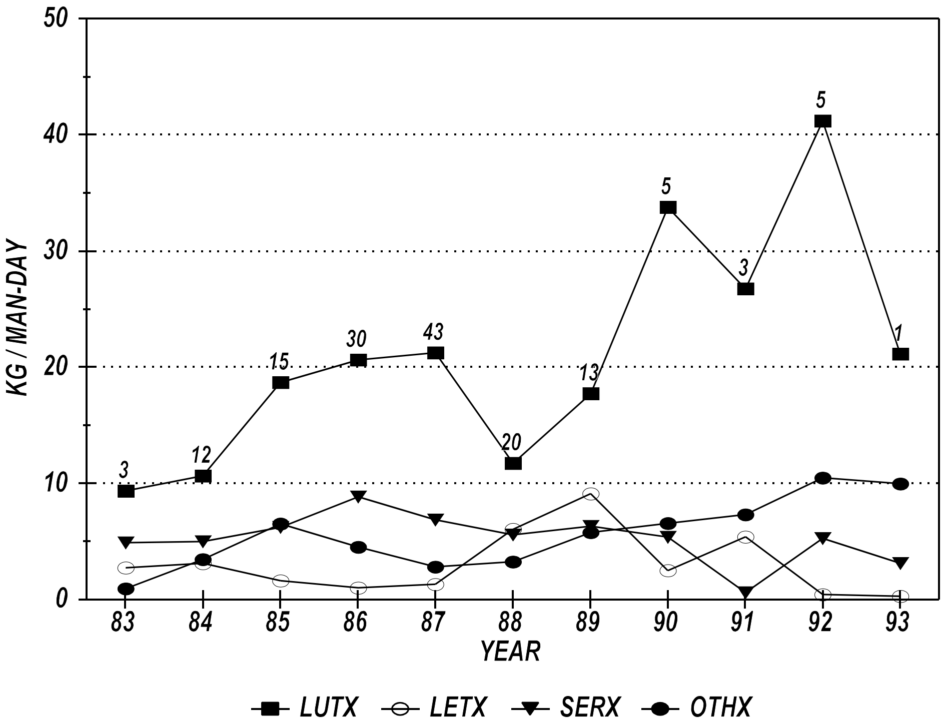

Figure 5.44 : Family guild catch rates for basic schooners using handlines only at the south edge of the Mahe Plateau (number of sampled trips indicated)

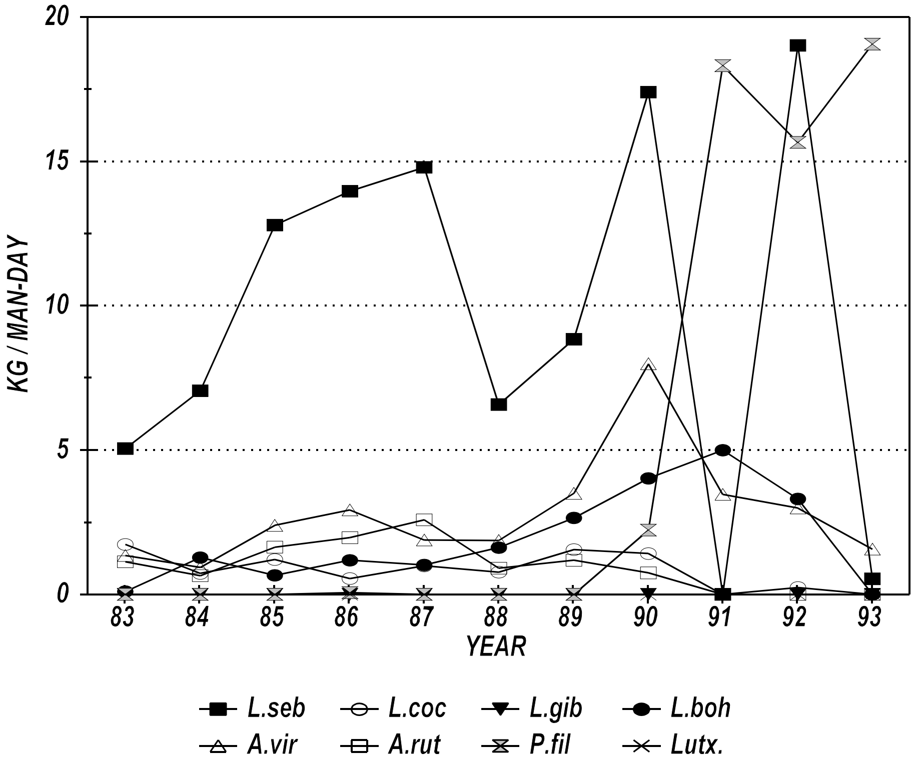

Figure 5.45 :Catch rates for the main species caught by basic schooners with handlines only at the south edge of the Mahe Plateau, Seychelles.

Figure 5.46 : Observed and standardised catch rates for the guild of all demersal species caught by traditional whalers with handlines targeting demersal species from the inshore sector of the Mahe Plateau, Seychelles.

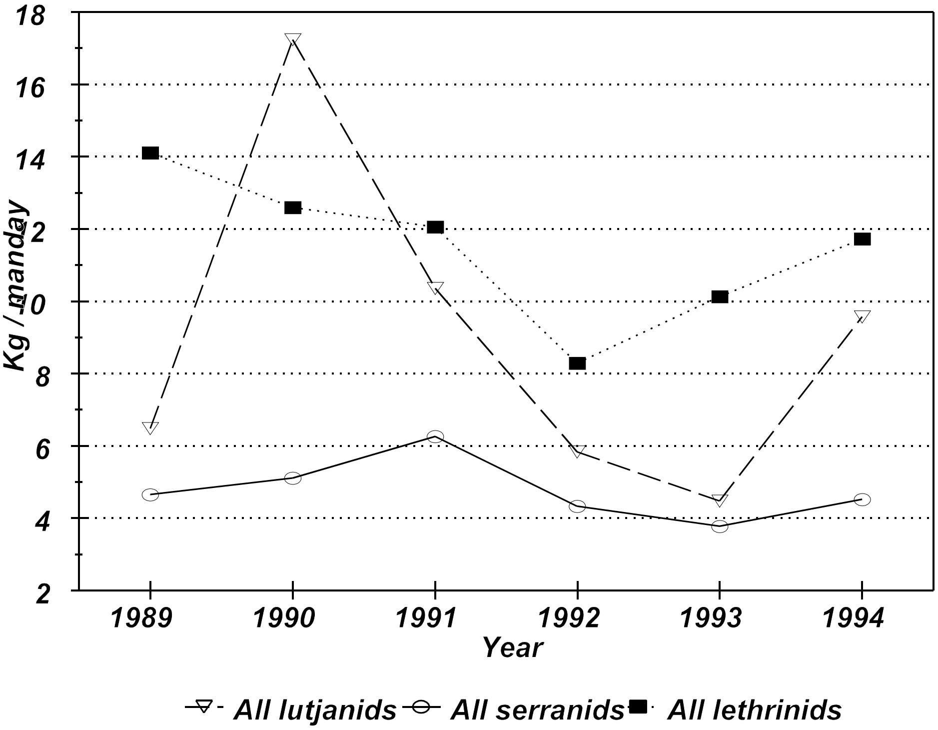

Figure 5.48 : Standardised family guild catch rates for fish caught by traditional whalers with handlines targeting demersal species from the inshore sector of the Mahe plateau, Seychelles.

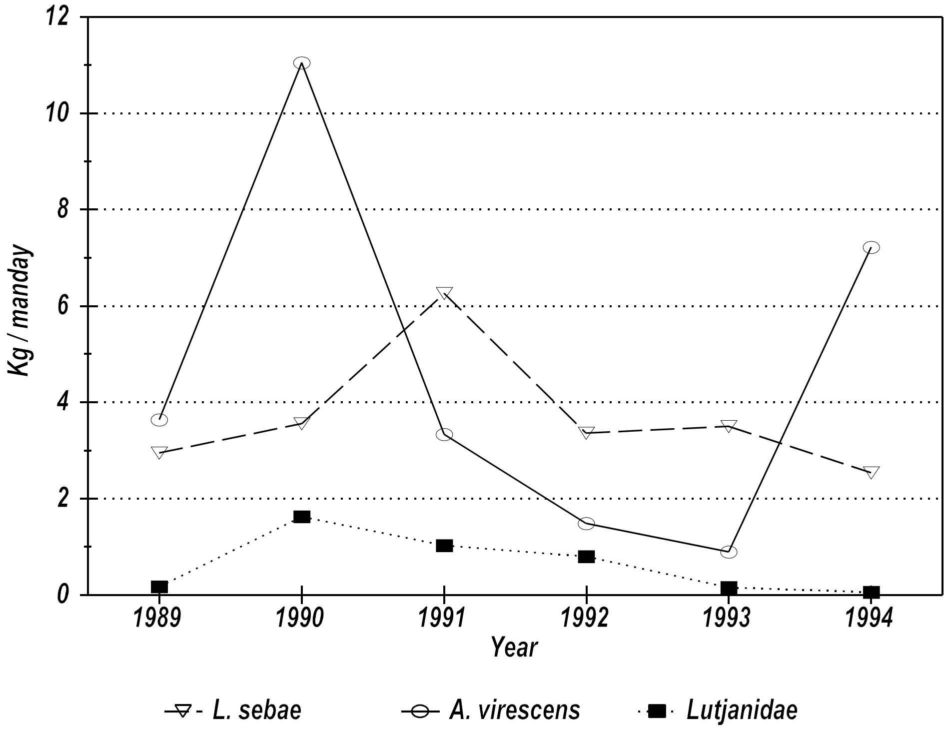

Figure 5.47 : Standardised catch rates for the principal lutjanid species in the catch and a guild of all remaining lutjanids for traditional whalers with handlines targeting demersal species from the inshore sector of the Mahe Plateau.

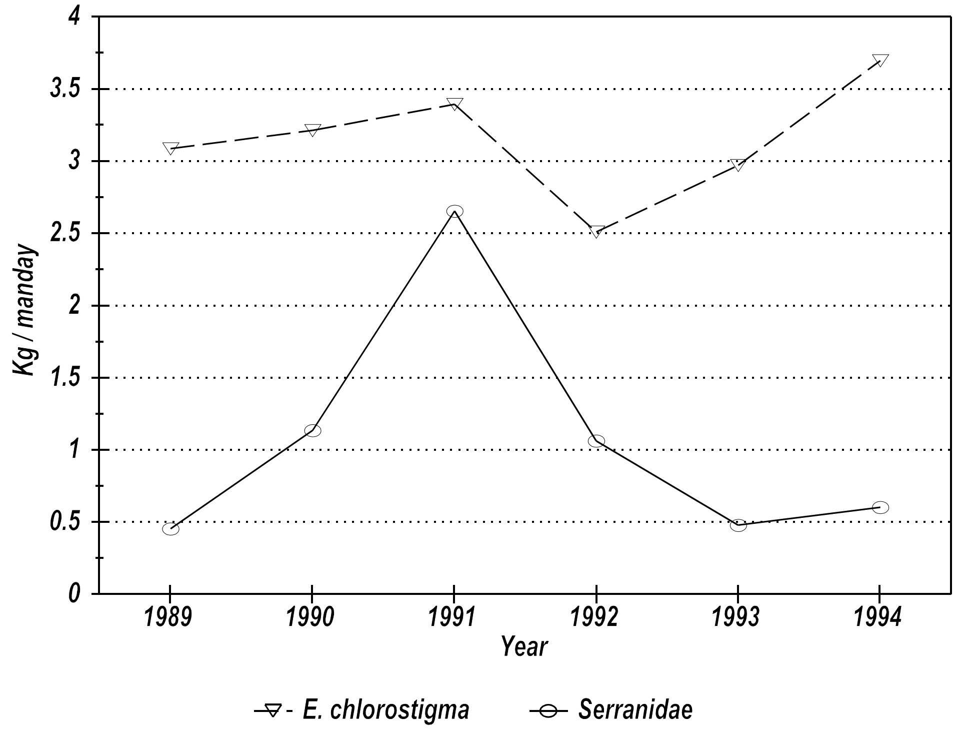

Figure 5.49 : Standardised catch rates for the principal serranid species in the catch and a guild of all remaining serranids for traditional whalers with handlines targeting demersal species from the inshore sector of the Mahe Plateau.

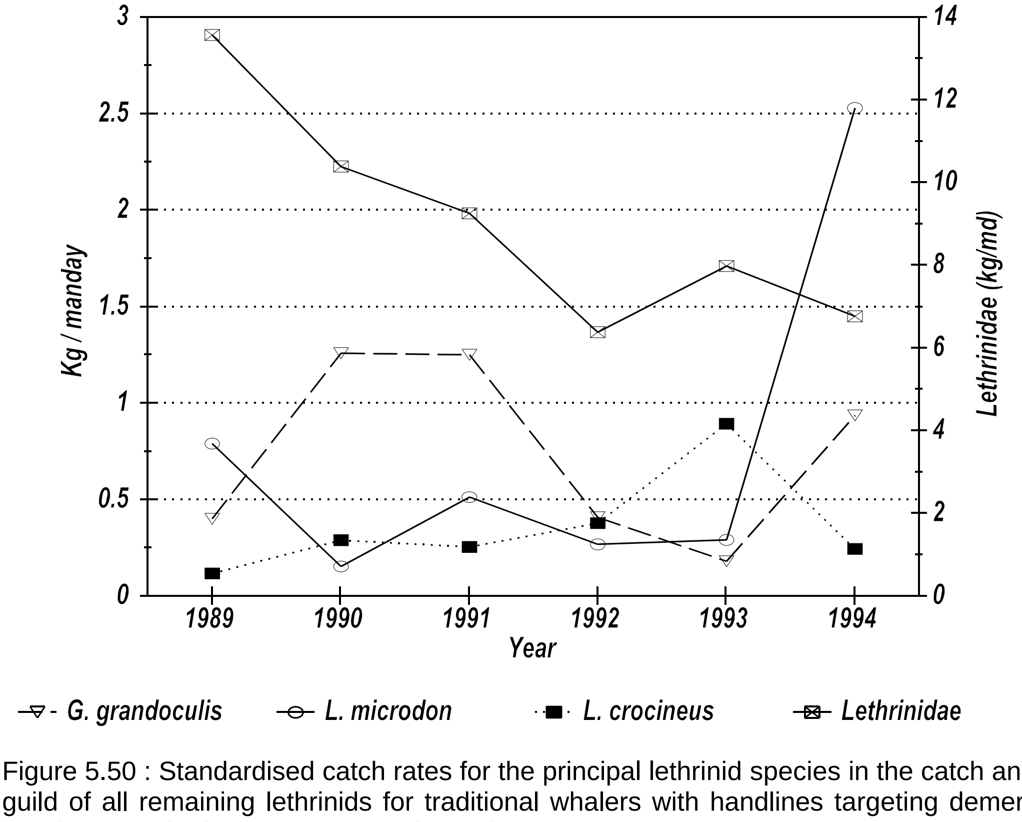

Figure 5.50 : Standardised catch rates for the principal lethrinid species in the catch and a guild of all remaining lethrinids for traditional whalers with handlines targeting demersal species from the inshore sector of the Mahe Plateau.

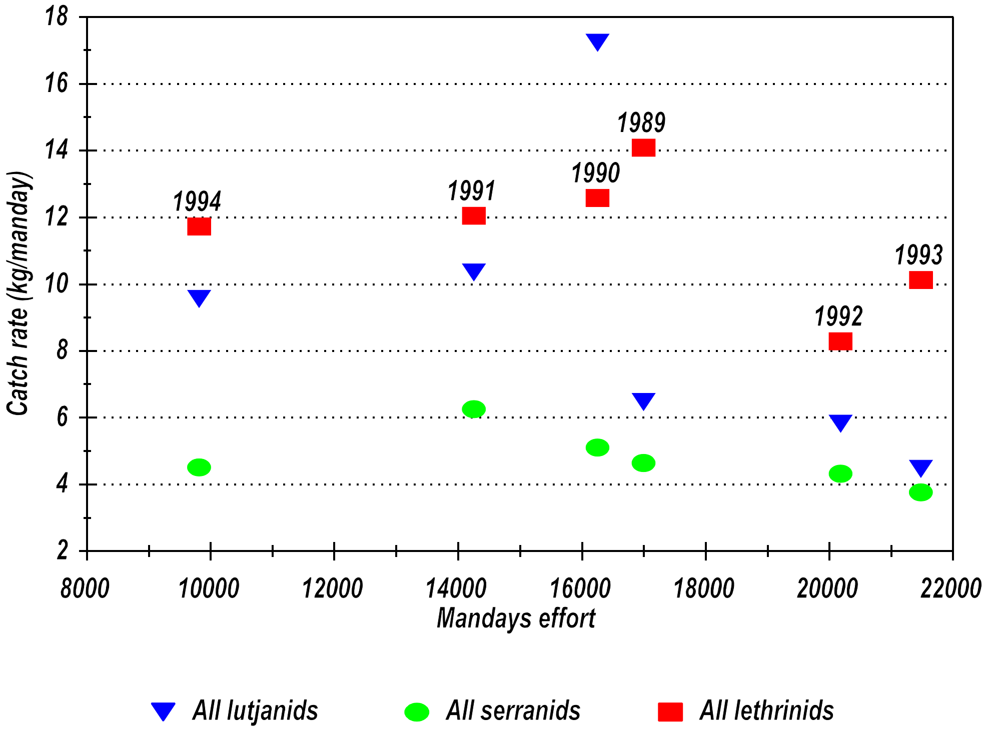

Figure 5.51 : Standardised family guild catch rates for traditional whalers with handlines targeting demersal species against total effort for all whalers catching demersal species from the inshore sector of the Mahe Plateau.

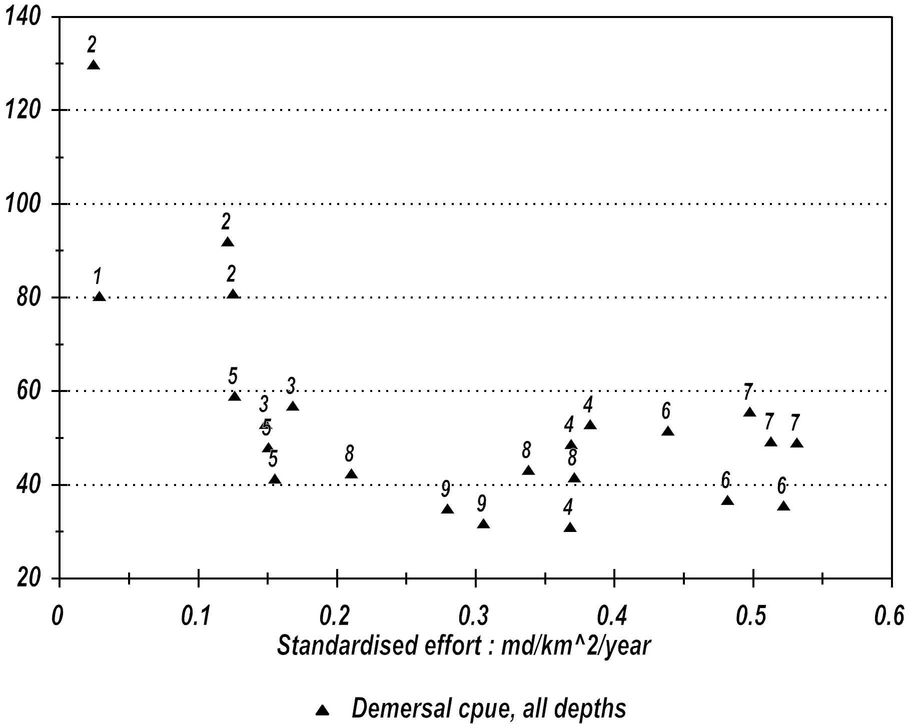

Figure 5.52 : Standardised demersal catch rates by Indian Ocean bank (1-9, indicated) against mean annual standardised fishing effort per square kilometre of bank

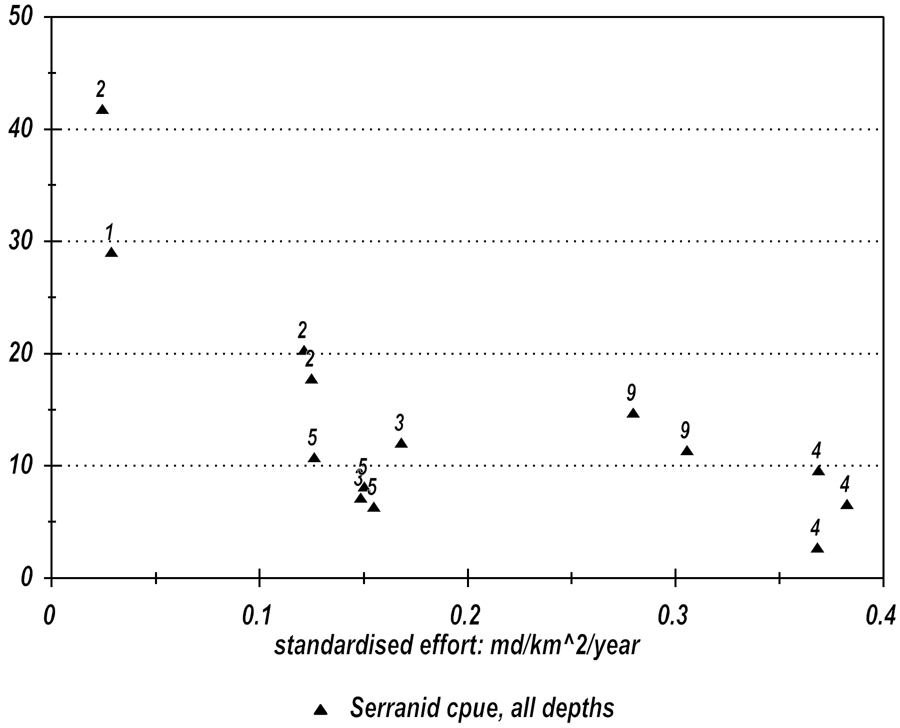

Figure 5.53 : Standardised serranid catch rates at Indian Ocean banks in Seychelles and Chagos only (1-5,9 indicated) against mean annual standardised effort.

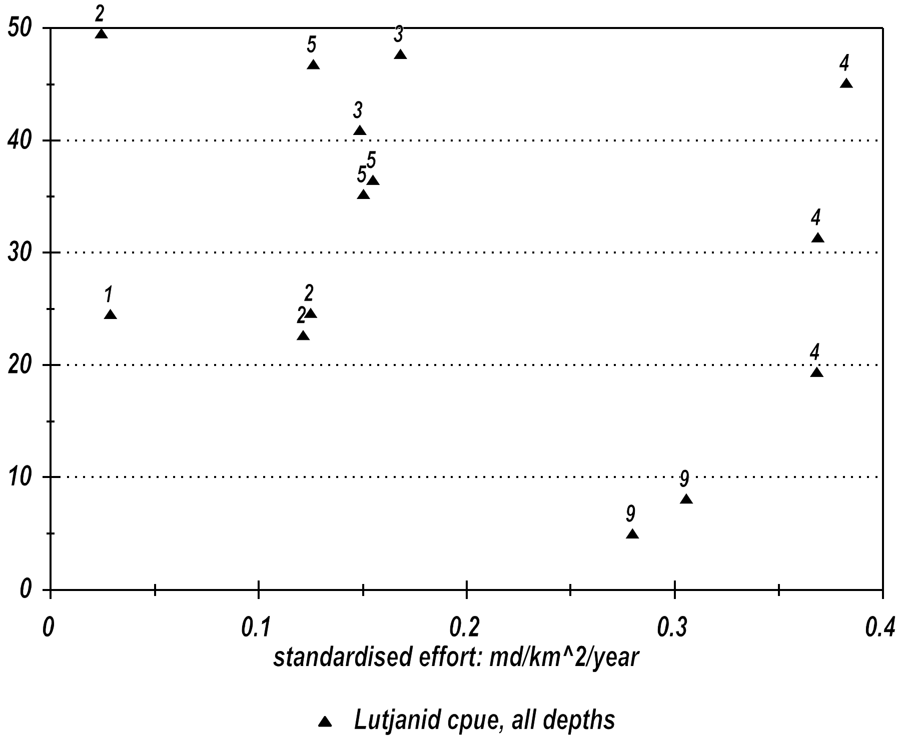

Figure 5.54 : Standardised lutjanid catch rates by Indian Ocean bank, Seychelles and Chagos only (1-5 and 9, indicated) against mean annual standardised fishing effort.

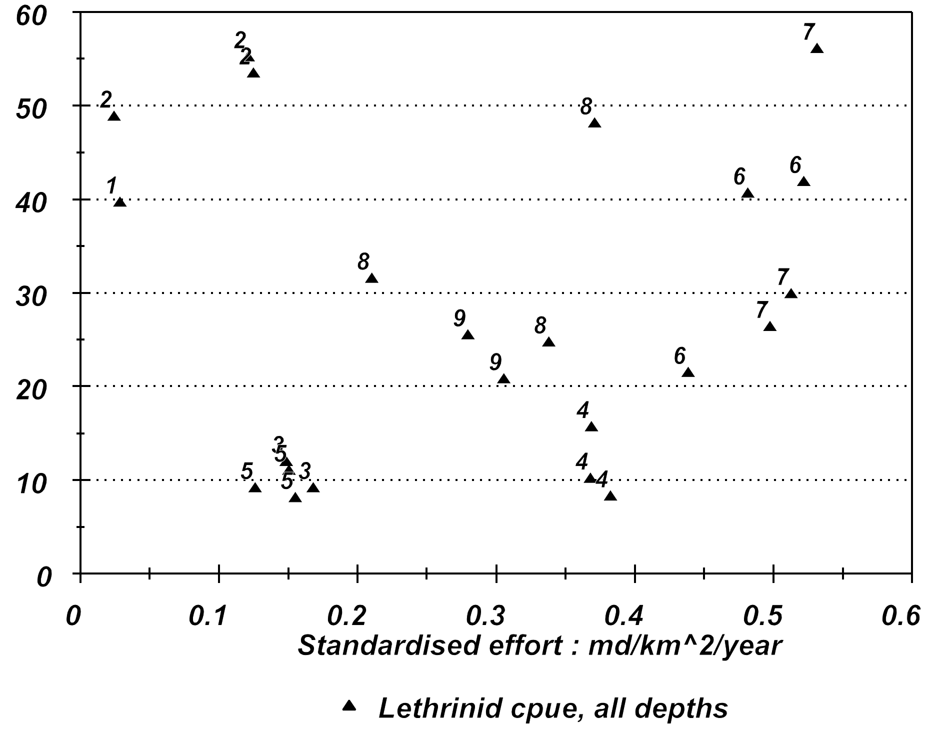

Figure 5.55 : Standardised lethrinid catch rates by Indian Ocean bank (1-9, indicated) against mean annual standardised fishing effort.

6 MANAGEMENT OF MULTISPECIES FISHERIES

6.1 Fishery Management

This chapter describes the background to fisheries management, the specific objectives and current management strategies adopted in case study countries, and data requirements for management. Simulation studies indicate the effect of different management strategies on single and multispecies resources and derive guidelines for management. Further conclusions and recommendations for the development of a management strategy (plan), however, are not given here, but were presented in the Final Report (Section 1.5.2).

6.1.1 Background

The objectives of fisheries management may be diverse, may be contradictory in their aims and will include social, economic and biological considerations. In addition to maintenance of the resource, management should focus on the avoidance of severe ecological, economic and social imbalances, the reduction of conflicts between resource users, and equitable income distribution from exploitation of the resource (Wilman, 1983). It should also be recognised that for any management strategy to work, it should take into account all of the factors affecting the resource from initial harvesting through the marketing chain to the consumer. Management also implies the need for monitoring, control and surveillance and a strong legal framework, and the costs and benefits of any strategy are an important consideration. Managers must therefore clearly prioritise objectives in order to develop an appropriate management strategy. Biological solutions to management alone are not always the most appropriate, and do not generally agree with the political realities - demands for increased employment for example. However, fishing stocks at levels beyond sustainable yields cannot be tolerated so alternative means of maintaining them are required. Single solutions are generally not practical, and a range of management measures are required within an overall management strategy.

The type of management advice required for any fishery will depend upon the degree to which it is exploited. Polovina (1992) indicates fishery models which may be useful for management under different fishing pressures, beginning with an unfished stock where simple yield per unit area methods can be applied to determine the level of capitalisation required, to ecosystem overfishing where complex management procedures are required to account for interactions. Management generally aims to prevent overfishing, and to optimise the benefits gained from exploitation, whether they be maximum food production or generation of the maximum economic rent from the fishery. For underutilised resources this implies the introduction of additional fishing units (boats) and developments in gear technology, ie. an increase in fishing effort. To prevent overfishing the optimum level of effort must be ascertained and further development limited. Additionally the optimum age at capture of targeted fish is important to maximise yield. Where overfishing has already occurred, often severe management measures are required in order to allow resources to recover.

The present study relates to management strategies for multispecies fisheries and is confined to examination of biological management controls as a component of any wider strategy addressing the issues raised. Discussion is presented of social and economic factors, but the application of socioeconomic management tools or the development of bio-socioeconomic management models is beyond the scope of the study. It focuses specifically on the management of multispecies demersal banks and deep reef slope fisheries for lujanids, serranids and lethrinids. These are atypical of many near-shore coastal fisheries exploited frequently by subsistence and artisanal fishermen in developing countries in that they target high value species often destined for the export market. These species are also frequently sought after in local markets, including those servicing the tourist industry. The vessels required to exploit the resource require a higher level of capitalisation than for nearshore resources. The nature of the fishery is thus more commercially oriented. An appropriate management strategy would thus be to optimise maximum economic yield on a sustainable basis from the fishery. However, in the case study countries, social issues are also relevant affecting the strategy to be adopted.

Management strategies should ideally be simple, requiring the minimum of data, and should be easy to enforce. This is especially true for complex artisanal multispecies fisheries where classical management methods based on the temperate model for commercial fisheries are considered inappropriate (eg. quotas, TACs) due to the mixed nature of the catch, difficulties of enforcing regulations, and requirement for good data to predict stock sizes. However, for multispecies demersal banks and deep reef slope fisheries, their economic importance suggests that expenditure on at least some data collection and enforcement is worthwhile, and classical management methods may have value. A well designed management programme will utilise a number of methods. Co-management of the resources, that is, the sharing of responsibility and authority between the government and local fishers/community to manage the fishery (Pomeroy and Williams, 1994) is also seen as essential to achieve viable and enforceable resource management. This is an extension of the present basis of community management which exists in some parts of the world such as Oceania (eg Ruddle *et al,* 1992) and Japan, and of the concept of territorial use rights in fisheries (TURF's, Christy, 1982). In co-management Government provides scientific advice on optimal management strategies, whilst the implementation of the strategy is devolved to the community.

6.1.2 Management methods

Biological management of capture fisheries resources is generally aimed at preventing overfishing and optimising yield. Methods available have principally been developed for temperate water single species stocks. This raises the question of management strategies for multispecies fisheries - for example, should overfishing be allowed on one species in order to increase catches of another as has been proposed; can single species be managed within a multispecies system? The utility of different management methods may be considered in terms of their applicability to the management tools available (see below), and generally to the management of the multispecies resource. This section examines the practical utility of the methods available.

Three types of overfishing are defined for which management advice is required :

- Growth overfishing, where the fish are caught below the size at which optimum growth has occurred to maximise yield;

- Recruitment overfishing, where the spawning stock biomass has been reduced limiting recruitment, and potentially leading to stock collapse

- Ecosystem overfishing, where fishing results in changes in abundance and species composition and the resulting ecosystem is undesirable or inefficient for fisheries.

The principal methods for optimising yield and preventing overfishing include regulation of age at first capture and regulation of effort or catch. Yields may also be increased by enhancement of the natural production.

- **Regulation of age at first capture**

This involves establishing the optimum size at capture in order to allow a cohort of fish to grow to its optimum biomass and to reproduce. Minimum size limits or gear restrictions may be imposed. Minimum size limits have been applied to demersal fisheries (eg. Huntsman and Waters, 1987; Ibrahim *et al*., 1988) but are generally not considered to be a viable option, particularly for deep slope species due to expansion of gas bladders and evisceration leading to high mortality of returned fish. Survival rates of released fish are inversely proportional to the depth at which they are caught, and are high for relatively shallow water. For example, in a study where survival was monitored in cages, only 56% of *L.campachenus* caught at depths of 37-40m survived on release (Gitschlag and Renaud, 1994).

Direct management of size at capture may therefore only viably be achieved through gear restrictions. Some demersal fisheries, for example that on the northwest shelf of Australia utilise trawls, and regulation of mesh size is a viable option (Sainsbury, 1984). Except for a trial period in Seychelles when fish traps and gill nets were used, for study fisheries hooks and lines are the principal fishing method. Regulation of hook size is thus the potential means of regulating size at capture. Hooks are relatively unselective to large fish and Ralston (1982) found that small hooks were nearly as efficient at catching large snappers from the deep bottom snapper fishery in the Marianas as larger hooks. He examined the selection curve for hooks to determine which model (a normal or logistic curve) best described sampling characteristics of fish hooks (Ralston, 1990). Neither model fitted well, but for the ascending left hand selection ogive he found a sigmoid curve. This is where selection is important in relation to fish size and hook size. However, small differences in size at capture occur with relatively large changes in hook size. An example relates to longline fisheries where modification of fish size is achieved within a range of 67-217% increase in hook size (cited in Ralston, 1982). For *L. mahsena* on the Saya de Malha bank, Bertrand (1988) found no relationship between hook size and fish size. By contrast, by utilising larger hooks Correa-Ivo and Sobreira-Rocha (1988) found that juveniles of *Lutjanus purpureus* were protected. In other fisheries, changes in size at capture have been demonstrated, e.g. for tuna the optimum size at capture increased with hook size (Cortez-Zaragoza et al, 1989). Thus, whilst it would in theory be possible to regulate size at capture in demersal fisheries through hook size, this may not be practical.

Indirect methods of limiting size at first capture may include closed seasons or closed areas. Closed seasons are most appropriate for short lived fast growing species where fishing is restricted until spawning has occurred. This is not a viable option for study demersal species, although closed seasons around spawning time may be a viable management tool for other objectives. Closed areas to limit size at capture may be a viable management tool where the location of nursery grounds is known. Selective targeting, for example, by depth band, may also be useful, although the present study indicated no significant relationship between size and depth. Enforcement would also be difficult.

Regulation of age at first capture is a single species management method. For multispecies fisheries it would be possible to asses the optimum size and age at capture for each component species. However, limited information on hook selectivities is available and would need to be ascertained for each species. Assuming that size selection was then feasible through gear regulation, it is likely to be the case that a different hook size would be required for each species. A targeted (single species) fishing strategy would be required, and enforcement would be difficult. For the multispecies complex a ‘best’ hook size could be defined to optimise total yield, but would result in species composition changes due to differential catchabilities of the component species to the gear. Methods for estimation of a ‘best’ gear size for trawl fisheries are given by Sinoda et al. (1979) and Sainsbury (1984).

- **Regulation of fishing effort or catch**

Fishing mortality may be limited by restricting catches through quotas, or by limiting effort by means of restrictive licensing of vessels (fishermen) and gear. Closed seasons and closed areas will also serve to limit catch and effort. The aim is to ensure that sufficient numbers of fish survive to reproduce (ie. the spawning stock biomass is maintained, usually above a threshold of 20-35% of initial biomass, Mace and Sissenwine, 1993), and that the resource may be sustainably exploited (ie. catch rates are maintained).

Management measures may be applied directly or indirectly to the fishery. For example, fishing effort may be limited directly by restricting the number of vessels licenced to fish, or indirectly through the provision of alternative employment, or the introduction of economic disincentives to fishing through taxes and price controls. Export controls may also be used to limit catch and effort and are potentially useful for study fisheries. Where possible, diversification of the fishery through the development of alternative resources may also lead to a reduction of catch and effort.

Catch quotas are implemented through a number of means, such as setting a Total Allowable Catch (TAC) for the fishery, and closing it once that catch has been removed. Individual Transferable Quotas (ITQ's) are a variation on this whereby fishermen buy units of a TAC. Management through regulation of catch is expensive and requires detailed stock assessment, monitoring and enforcement. For multispecies resources quotas would need to be established for each component species. High grading, the discard of lower value species, and where fish are marketed by size, discard of lower value sizes occurs as fishermen aim to maximise the value of their quota. Where targeted fishing is not an option, such as where the gear is unselective by species, and there is no habitat or depth separation, a multispecies fishery may be closed when one species reaches its quota whilst other resources are lightly exploited. Catch quotas are inappropriate for multispecies resources (see also MRAG, 1996b).

Restrictive licencing to limit fishing effort is commonly applied, but is not suitable for subsistence and artisanal fisheries where alternative food sources are not available. In commercial fisheries it can lead to overcapitalisation where the fishing power of a licenced unit is able to increase. Limited entry is however, relatively easy to implement and enforce. The method alone does not prevent excessive effort on the most valuable species of a multispecies resource, and regulation of effort by species is not practical.

Gear restrictions may control size at capture and the species composition of the catch. Ecosystem manipulation such as predator control has been proposed as a means of managing multispecies fisheries (see Section 5.1.1) to enhance the sustainable yield. This would be achieved by applying different fishing mortality to each species through gear and targeting effects to adjust their catchabilities.

Closed seasons may be applied to protect spawning fish, or to allow them to grow to optimal size. Thus they serve an additional function to that of limiting effort. Generally they are considered to be of little value in multispecies fisheries where different spawning periods may occur for each species. For study species such as the lutjanids where spawning occurs throughout the year benefits may be small. However, where the location and timing of spawning aggregations is known (typically for serranid species, see Section 4.6) there is merit in introducing an area specific closed season.

Closed areas (or reserves or marine protected areas) reduce the amount of the resource vulnerable to capture and eliminate or reduce fishing effort on part of the stock. As a fisheries management tool, reserves are also expected to enhance fisheries yields outside the reserve through movement of adults and increased recruitment to fishing grounds, and to lead to localised 'recovery' of depleted resources. In addition to protection of fish stocks, they may have other important functions such as conservation of important habitat types (eg. nursery areas). Closures may be permanent or temporary, and in some cases fishing may be permitted under certain conditions or with certain gear. The variety of closed areas are reviewed by Roberts and Polunin (1991).

Closed areas are generally considered applicable to small scale artisanal and multispecies fisheries. Complete understanding of population and community dynamics is not required, simplifying data collection, and, together with some form of community management, they may more easily be enforced than classical management methods. A body of literature exists on reserves and confirms their role in permitting the recovery of depleted resources. The extent of spillover effects to adjacent areas is however less well understood. Circumstantial evidence from a study in the Philippines suggests increases in fish catches outside reserves which were related to periods of closure (Russ, 1989). If this occurs, it is not known whether it is sufficient to compensate for the net loss of yield arising from closure of part of the fishing grounds and it may be the case that overall yields can locally decline. Not all fishermen regard reserves as beneficial. They may lead to a false sense of security and a belief that the resources are being ‘managed’ when they are not. As a fisheries management tool reserves may also be an expensive option, and require strict enforcement.

For study fisheries, closed seasons and areas may be of some value to protect spawning stock, but the extent of knowledge on larval dispersal and recruitment patterns is limited (see Chapter 3). Considerably more information is required to describe these, and to develop spatial models to establish the number, size and relative locations of reserves for effective management.

Pulsed fishing, that is extended temporary closures which may be applied on a rotational basis, may also be useful for study fisheries. Where recruitment overfishing has not occurred the rate of recovery may be predicted from age based models such as MIDAS assuming constant recruitment. Mees (in press) examined the recovery of areas in Seychelles subject to periodic fishing, but not severely depleted, and for *P. filamentosus* found that biomass increased at a rate of 228 kg km^{-1} year^{-1} (of 100m isobath), similar to the estimated sustainable yield. For more heavily fished locations recovery would be slower. Observations by fishermen in Tonga indicate that seamounts south of Tongatapu at which catch rates had declined, recovered sufficiently to permit fishing again after about 5 years (Latu ant Tulua, 1992). Russ and Alcala (in press) monitored recovery of snappers and groupers in reserves in the Philippines and found that the mean density of predatory fish increased linearly but that biomass increased in a curvilinear fashion. Increases in biomass were slow for the first 3-5 years and subsequently increased leading to recovery after 10 years. Unregulated fishing over a period of 1-2 years eliminated gains from closures. For effective management on a hit and run basis closures would need to be for periods of at least 5 and ideally 10 years. These situations relate to areas of localised overfishing. Due to spatial connectivity of the resources recovery may occur. Where recruitment overfishing has occurred for a totally isolated location, recovery may be unpredictable. Following depletion of snappers and groupers in the Dominican Republic, no recovery was observed after 20 years without commercial fishing (Sullivan, pers comm.). *L. synagris* had not recovered on the Cuban shelf after protection from fishing for 10 years (Claro, 1991). This suggests that catch or effort limits need to be applied in addition to rotational closures.

- **Fisheries enhancement**

Managers may also aim to increase yields above the optimum carrying capacity of the natural environment through stock enhancement. This may occur in certain situations, and for certain species through seeding of reefs with juveniles or habitat enhancement by constructing artificial reefs. A considerable literature exists related to fisheries enhancement, and for demersal species, in particular to artificial habitats for enhancement of serranid resources (eg. Beets and Hixon, 1994). However, in the context of demersal banks and deep slope fisheries these methods are considered to be of little value. Reseeding and artificial habitats are expensive, and would not be cost effective for the large areas concerned; the benefits provided are questionable; artificial reefs may not enhance resources but simply aggregate them increasing catchability and the potential for overfishing; placement of artificial reefs may disturb nursery habitats for desired species.

Introductions of exotic species have occurred in the Hawaiian Archipelago, and of 11 species of lutjanids, lethrinids and serranids, only 3 have established significant populations. One species, *Lutjanus kasmira* has become widely established and contributes an annual commercial catch of about 30 t. This appears to have increased the total yield of demersal species (*in* Moffit, 1983). However, as a general rule, introductions of exotic species are not to be encouraged.

6.1.3 Management tools

A number of tools exist to assist managers in deriving appropriate criteria for the management of fish resources and are fully described elsewhere (eg. Gulland, 1983; Hilborn and Walters, 1992). These have been principally designed for single species and include :

- surplus production models such as Schaefer (Schaefer, 1954) and Fox (Fox, 1975) which express yield as a function of fishing effort;

- analytical models such as yield per recruit (Beverton and Holt, 1957) which explain the surplus yield by examining size specific production of a cohort of fish taking into account growth and mortality processes;

- cohort/virtual population models (Pope, 1972) which estimate the number of individuals that must have been present to account for a known catch after losses due to natural mortality;

- Age structured models whereby a non-equilibrium catch and effort type model is derived from a yield per recruit model (Schnute, 1985). These have been extended to develop size structured models (Schnute, 1987, Schnute *et al*, 1989). They are used principally as simulation models.

Kerr and Ryder (1989) review current approaches to the analysis of multispecies fisheries, whilst the ICES approach for the North Sea and Atlantic, principally through multispecies virtual population analysis (MSVPA), is described in Daan and Sissenwine (1991). Polovina (1992) examines the applicability of various models to multispecies fisheries in the tropics. Models range from those applied to single species in parallel, which assume no interaction; those which group all species together and treat the whole as a single species (eg. total biomass Schaeffer model); to complex ecosystem box models where all interactions are modelled (eg. the ECOPATH model; see Table 6.1). MIDAS, developed for the present study is a fully age structured simulation model.

METHOD	MODEL	EXTENSION OF MODEL TO MS SITUATION	COMMENTS	DATA REQUIREMENTS
Comparison of like areas	Yield per unit area	Directly compare like areas	Useful first approach. Requires the least data of all to derive first estimate of potential yields	Environmental characteristics, area of habitat, Catch and effort by location for Munro approach (need not be time series data)
		Plot Y/area vs F/area similar to production models to determine MSY		
Single species	Production models (Schaeffer / Fox)	Total biomass Schaeffer model	Sometimes useful, e.g. apical predators - not for prey species	Time series catch and effort data by species / location
		Multispecies Schaeffer model / Lotka Volterra models	No useful results to date, Impractical for large no's species	Must aggregate data to be useful
	Analytical Y/R models (Beverton and Holt)	Apply individually to single species and sum yields	Can be useful, but interactions?	Demographic variables (K, M, Loo, R etc) by species. Length frequency data can be usefully applied
		Modified Y/R models incorporating interactions	These models are complex Too many parameter estimates are required and the models need to be simplified to be useful	
		Summation of single species Y/R equations		
	Cohort (Virtual Population) analysis	Single species cohort analysis plus regression to evaluate interactions	Very complex. Data intensive Need species specific variables plus estimates of interaction (predation)	Demographic variables Catch at age, Time series catch and effort, gut content analyses
		Multispecies VPA		
Ecosystem models	Ecosystem box models, e.g. ECOPATH	Designed to incorporate all trophic levels and interactions	Highly complex in that all interactions must be understood. Data intensive Unreliable due to errors inerrant in est. all variables	Demographic variables, est. of primary production, catch, effort - DO NOT need time series data

Table 6.1 : A summary of model types currently applied to multispecies fisheries

Adaptive management has been proposed as a means of ascertaining, through experiment, the most appropriate model for single and particularly multispecies management (Walters, 1986; Sainsbury, 1988; Hilborn and Walters, 1992). This approach has been applied to the demersal trawl fishery in north western Australia (Sainsbury, 1991)

■ Single or multispecies management

For the management of multispecies fish stocks, May (1984) suggested that single species models are inadequate, whilst ecosystem models are unreliable due to the errors inherent in estimating all the various parameters required. MRAG (1993) suggested that for multispecies fisheries, if biological interactions (competition and predation) are not important, a single species approach using dynamic pool models may be adequate. However, this implies the estimation of a huge number of parameters to describe a multispecies system and simpler models involving aggregation of species may be more appropriate. No reliable theoretical rules exist to assist in this process, and the means of identifying guilds may be largely subjective. Where predation may be significant, and in the absence of detailed information on stock recruitment relationships, yield per recruit assessment methods offer the best approach.

Study species share similar biological characteristics, and whilst interactions between them occur, they are considered to be relatively minor. Single species assessments may thus be adequate for these fisheries. Hunstman *et al* (1983) applied single species yield per recruit models to 9 species of the US south Atlantic Bight snapper and grouper fishery. Results for each species were similar and suggested a common management strategy for all species. Where length at capture could be kept high, a high yield could be maintained regardless of effort, and thus a management strategy based on minimum size at capture was proposed. Single species assessment was also used to establish the yield of deep slope snappers and groupers in the Marianas (Polovina and Ralston, 1986) and the multispecies yield was determined as the product of the yields for seven species for which biological parameters were available, plus an eighth group (all other species) for which an averaged value of yield to biomass for the seven known species was used. Single species yield per recruit models have been applied to the Hawaiian fishery to evaluate the impact of fishing mortality on spawning stock biomass (Polovina, 1987).

Single species assessments may also be appropriate for the economically most important species of a multispecies fishery. However, if a single species approach is to be adopted for study species then different management models may be appropriate relative to the biological characteristics of each species. For example, owing to the reproductive strategy of spawning aggregations for serranids, in particular, Shapiro (1987) suggests that management plans for protogynous species need to be based on yield models especially designed for such reproductive systems (see Section 4.6). Bannerot *et al* (1987) examine management strategies for snappers and groupers in the Gulf of Mexico and Caribbean in relation to their reproductive strategies.

Aggregation of species into guilds has also been applied to study species. Ralston and Polovina (1982) used cluster analysis to define guilds of snappers and groupers from the Hawaiian deep slope handline fishery. A Schaefer stock production model to determine total yield fitted these data better than that for individual species, but only marginally better than by aggregating all species together. Total biomass production models have not been so successfully applied for guilds incorporating different trophic levels. However, where competition and species substitution in the catch may occur, aggregation reduces variability explaining why the total biomass model fits better than that for individual species (Polovina, 1992).

Single species models and aggregate single species models however, do not fully explain the multispecies system. Competitive interactions for bait and changing catchability between species will not be indicated. Catchability of single species may change as fishing progresses and the most aggressive fish are removed first. Other species then become susceptible to the gear. Polovina (1986) demonstrated variable catchability in snappers due to competition for bait. Catchability relates catch rate to population size and single species and aggregate assessments of this parameter can lead to inaccuracies in population estimates. The question remains as to whether, for demersal fisheries based on typical study species, single species assessments and management advice based on them are adequate, or whether more complex multispecies models which explain biological interactions are required in order to derive appropriate management guidelines. This is explored in Section 6.3.

6.2 Existing management strategies for case study fisheries

Single species and aggregate single species models have been applied to each of the case study fisheries with the principal aim of determining biomass and sustainable yield estimates (see Table 6.2; Section 3.2.2, MRAG 1994; 1996a; 1996b). Yield per unit area methods have been applied comparing case study locations with ecologically similar areas, and by raising estimates derived from smaller areas. Depletion studies based on the short term Leslie model and longer term Allen and Schaefer and Fox surplus production models have been used with varying success. Total biomass models were generally found to fit the data better than those for single species. Single species resource assessments were derived from length converted cohort analyses and, assuming equal catchability, raised to estimate total biomass. The sum of single species assessments, and raised single species assessments tended to suggest a higher yield than estimates based on aggregated data. These assessments allow managers to make decisions related to catch and effort controls on the fishery, and translate to the potential for further development and investment, or the need to limit or reduce investment in the fishery.

Yield per recruit models have been applied to single species for *P. filamentosus* from Tonga (Mees and Rossouw, 1995) and Seychelles (unpublished) and *L. mahsena* from the Saya de Malha Bank, Mauritius (Lebeau and Cuef, 1976). In fact the analyses of the data for *P. filamentosus* utilised MIDAS, and differed from a yield per recruit analysis in that a stock recruitment relationship was also incorporated. These analyses allow managers to make decisions related to the optimum level of effort (fishing mortality) and the size and age of fish at first capture.

Complex multispecies models incorporating interactions between species have not previously been applied to study fisheries. However, management based on single species and the use of aggregate single species models would appear adequate. This is confirmed in the present simulation studies (see Section 5.2.4). Single species management is particularly appropriate in the case of Mauritius, where the catch is predominantly mono-specific, and that of Tonga. Whilst the Tongan fishery is multispecies in nature, 6 target species constitute around 80% of the catch and there is relatively distinct depth separation between species. Fish may thus be targeted effectively allowing, for example, regulation of individual species catches. Seychelles fishery, and that in the Chagos Archipelago is more complex and multispecies in nature. A distinction may be made of fish caught on the relatively shallow banks and those in intermediate depths and deeper water at the edge of the banks, and these ‘habitats’ could be managed separately. Single species management may be suitable for *P. filamentosus* in particular, which forms up to 50% of the handline catch from the intermediate depth band. In shallow water, targeting is clearly possible, and fishermen will aim to catch favoured species such as *L. sebae* which fetch a higher price, or may avoid potentially toxic fish as occurs for *L. bohar* in Chagos. However, single species management based on quotas for these species would not be appropriate. Targeting / avoidance cannot guarantee that a species will / will not be caught, and indeed, discards of *L. bohar* are known to occur in Chagos. Hence this raises questions of a multispecies nature :

- what are the consequences of targeting or avoidance of certain species;
- can a suitable management plan be found to ensure the sustained exploitation of desirable species such as *L. sebae.*

Location	MSY	85	86	87	88	89	90	91	92	93	94	95
Tonga										(11-12)	(1-12)	(1-6)
Banks	112	NA	NA	91%	109%	58%	29%	65%	NA	7%	41%	15%
Seamounts	217	NA	NA	166%	66%	105%	79%	65%	NA	8%	100%	48%
Total	329	NA	NA	140%	80%	89%	62%	65%	NA	7%	80%	37%
M6 as % of total catch		NA	NA	82%	55%	68%	86%	83%	NA	82%	74%	85%

Location	MSY	85	86	87	88	89	90	91	92	93	94
Seychelles											
Astove/Cosmoledo	85								25%		
Providence/Farquahar	345							26%	109%	9%	
Amirantes	590	11%	2%	8%	8%	6%	3%	14%	23%	3%	10%
Mahe Plateau -Inshore	458	97%	185%	132%	125%	160%	201%	182%	188%	186%	117%
Mahe Plateau - Remainder	4223	8%	17%	18%	15%	16%	23%	26%	13%	13%	21%
(less trawlable grounds)	1923	17%	37%	39%	32%	36%	51%	56%	28%	29%	45%
Banks S. of Mahe Plateau	408			1%				50%	14%	6%	1%
Total	6108	14%	26%	23%	20%	24%	31%	37%	33%	28%	24%

Location	MSY	85	86	87	88	89	90	91	92	93	94
Mauritius											
Saya de Malha	2500	88%	111%	135%	106%	87%	35%	71%	113%	123%	126%
Nazareth	2000	54%	60%	74%	71%	4%	28%	40%	49%	62%	79%
St Brandon	600	65%	89%	63%	124%	90%	102%	88%	107%	138%	68%
Total	5100	72%	88%	102%	95%	55%	40%	61%	87%	101%	101%

Chagos Archipelago	750	22%	17%	32%	42%	18%	34%	40%	41%	27%	41%

Table 6.2 : Estimated annual sustainable yield for study locations , and the proportion of the yield caught each year over the period 1985-1995.

This Table indicates the relative level of exploitation of demersal fish resources in each case study location, and suggests the potential for further development of the fishery or need for restrictive management. It does not indicate localised areas of depletion which may have occurred within the areas identified. Landings in excess of MSY may be expected from newly fished areas, and catch rates will be high initially, but will decrease as equilibrium yields are attained. Consistent fishing at levels in excess of MSY is a cause for concern.

Tongan yields and catches relate to the 6 main species, which are expressed as a proportion of the total catch. MSY is based on depletion estimates per nautical mile of 200m isobath, raised for the whole fishery. Estimates derived from production models suggesting a higher yield are not considered to be valid. Available data including that extrapolated from the first six months of 1995 (96%) suggests that the Tongan seamount fishery is fully exploited whilst additional potential remains on the deep reef slopes of the banks. The estimates do not include the shallower water potential available on the Tongan Banks, where there may be scope for further development.

Mauritian and Seychelles catches relate to the total demersal catch. The yield for Seychelles banks is that estimated for shallow and intermediate depth strata. Previous estimates have separated out the potential from trawlable grounds and this is indicated. Mauritian banks are fully / over exploited. The MSY estimate derived by Albion Fisheries Research Centre for Nazareth bank is less than that indicated, and exploitation on this bank is high. Chagos is not fully exploited. Except for the inshore sector of the Mahe Plateau, Seychelles banks are not fully exploited. Potential for development exists on the more distant fishing grounds. Inshore locations are overfished and there is a need to limit catch and effort.

Management of single species also implies the need for species specific resource assessments. Total biomass production models tend to provide more reliable results than those for single species in a multispecies complex. Assessments based on analytical models require a large number of parameter estimates for each species. Single species management could therefore only be justified for a few o the economically most important species. Can management strategies based on a few species plus a guild of all others be developed?

6.2.1 Management objectives

- **Tonga**

The deep water fishery was originally developed with the objective of relieving fishing pressure on inshore reefs, and was aimed at providing small scale artisanal fishermen with an alternative resource. This objective has not been achieved. Vessels designed for the fishery operated unprofitably with a few exceptions, and numbers declined. A commercially oriented export fishery developed and new more efficient vessels were introduced. It has been suggested that new objectives related specifically to the commercial export fishery are required (Anon, undated). Of the alternatives of maximising employment, maximising yield and maximising profitability it was proposed that new objectives should be :

- maximising profits through the creation of a small highly efficient fleet. Maximum economic yield from the fishery was estimated to be around two thirds of maximum sustainable yield (Latu and Tulua, 1992, although estimates were based on a Schaefer model for data aggregated over all seamounts, which is not considered appropriate, see Chapter 3; apparent catch rate declines were not statistically significant, see Section 5.3.2). The modernisation of the fleet may change the relationship of MEY to MSY.

- protect the resource from overfishing.

This approach may be the most appropriate from a fisheries perspective, but does not account for social objectives originally implied. It is understood that the objectives should include:

- diversification of the inshore fishery in order to relieve pressure on inshore reefs

- provide additional fish protein to the local market

- (provide additional employment and revenue to local fishermen)

Management objectives are not clearly defined and it is considered that this should be a priority for Government. It has been stated that the deep reef slope / seamount fishery and that on the banks are essentially separate fisheries with some species overlap. Management strategies with different social and economic objectives for each fishery could be developed.

■ Seychelles

The objectives of the fisheries sector relevant to the demersal fishery are defined in the National Development Plan, 1990-1994 (Anon, 1989) :

- To enhance its contribution to nutrition
- The creation of the maximum amount of work opportunities
- The maximisation of foreign exchange earnings
- The creation of optimum linkages with other sectors
- The ensurance of stable development in the industry
- The conservation of marine resources to ensure long term viability of the industry

The Development Plan indicates that while opportunities for part time and subsistence fishing must remain, the operations of the demersal fishery must be fully commercialised and there should be a move towards a more capital intensive fleet capable of exploiting the more distant fishing grounds. Development of the demersal fisheries in more remote locations should include promotion of fishing within the outer islands. The plan aimed to increase production from the demersal fishery to 5000 tonnes by 1994.

■ Mauritius

The broad objective of Mauritian Government policy was to increase fish production to 20kg *per caput* (in excess of 10,000 t in total), to stimulate the consumption of frozen fish and develop the fishery to replace imports of frozen fish. The aim was to provide a source of protein at a reasonable price. These objectives could not be met by the demersal banks fishery alone and implied the development of alternative fisheries (see SWIOP/MAFNR, 1989). Specific objectives for the management of the demersal fishery need to be clearly defined, and given the high level of exploitation of the banks should include prevention of overfishing (rehabilitation of the resource). Whether the objective is to increase economic efficiency in the fishery, or to maximise yield as a source of protein also needs to be clearly defined as this affects the most appropriate choice of management instrument.

Mauritius also need to develop collaborative management objectives with other Indian Ocean nations. The Saya de Malha bank is partly in international waters, and thus for management to be effective, agreement is required with other potential users of the banks resources (Reunion, Seychelles, recently Sri Lankan vessels).

Chagos Archipelago

The inshore fishery of the Chagos Archipelago is managed by the British Indian Ocean Territory Administration under the Fisheries (Conservation and Management) Ordinance of 1991. The principal objective is conservation rather than economic or biological maximisation of returns from the fishery. It is policy to ensure that all fishing is undertaken with due regard and concern for the stability of fish stocks, conservation of biodiversity and appropriate long term management of the resources. Fishing occurs in recognition of the historical fishing rights of Mauritian vessels.

6.2.2 Present management strategy

■ Tonga

In 1875 Tonga's constitution declared that all land and sea belonged to the King of Tonga, further defined in the Royal Proclamation of 1887 and the Land Act of 1927. This resulted in an open access fishery and any traditional community management authority was abolished (Petelo *et al.* 1995). Existing fishing regulations are described in the Fisheries Act, 1989, and the Fisheries Regulations (1992). Of relevance to the deep water fishery are :

- The Director of Fisheries must prepare fishery management plans
- All local fishing vessels over 6m must be registered and licenced
    - Licenses may be refused if it is necessary to give effect to any licensing programme specified in a fishery plan
    - Licences may specify restrictions relating to the type of fishery, location and gear
    - Different classes of local vessel may be prescribed and restrictions placed on the area or distance from shore which each class may fish
    - The Ministry of Fisheries may establish local committees to advise the Minister on the number and allocation of licences.
- Permission must be obtained for all foreign involvement in fisheries
- All foreign fishing vessels must have a valid license for Tongan waters (presently foreign fishing vessels are excluded from the deep water fishery)
- The Minister may declare any area a reserve for subsistence fishing and define types of vessel and gear that may be used to fish in the reserve
- A license is required to operate a fish processing facility
- The Minister may make regulations relating to closed areas and limited entry into all or any specified fisheries.

At present no management plan has been endorsed, and regulation of the deep water fishery is minimal.

■ Seychelles

The management strategy adopted to meet social and economic aims is described in the National Development Plan, 1990-1994 (Anon, 1989). In order to meet the objectives of distributing fishing effort more evenly and further offshore, and increasing catches from the demersal fishery, management strategies have included promotion of new vessels, provision of credit, incentive schemes, management and technical services and technology and fisheries extension development, and the development of new fisheries. These were promoted through a number of projects outlined in the 1990-1994 National Development Plan (Anon, 1989). Particularly relevant are the fleet replacement programme, fishing boat construction, schooner fleet management, promotion of demersal fisheries in the outer islands, development of fisheries technology, fisheries extension programme, and various infrastructure development projects. These programmes have led to the introduction of new vessel designs and fishing methods. However, progress has been slow. A particular problem is the cost of exploiting fish distant from Mahe. To improve the situation, recent policy has also been to move away from parastatal management of the fishing fleet towards entrepreneurial ownership. A mothership line fishing plan was also developed (Harris, *et al,* 1994), although it is understood that this is not considered the ideal solution, and presently there are no mothervessels operating in Seychelles.

Infrastructure to support fisheries development has been provided or is planned and includes provision of marketing facilities, ice, and outlets for fishing equipment. In line with recent policy, marketing practices have been relaxed and the state monopoly removed.

The principal legal instruments for the control of fishing are the Fisheries Act (1986), the Fisheries Regulations (1987) and the Licencing (Fisheries) Act (1987). Seychelles Fishing Authority (SFA) is required to collect and analyse statistical and other information on fisheries and to prepare and keep under review plans for the development and management of fisheries. In preparing plans, SFA shall consult with fishermen and others affected by the plan, and where practical with regional bodies.

The legislation is in place for the Minister (of Agriculture and Fisheries) to prescribe management measures including : closed seasons, closed areas, gear specifications, fishing methods or gear types, specification of species sizes or other characteristics of aquatic organisms that it is permitted or forbidden to catch, and schemes for limited entry into the fishery (Fisheries Act, 1986).

Certain regulations are directly relevant to the demersal fishery. By law, a local fishing vessel licence is required before fishing is permitted in Seychelles waters except for non motorised vessels less than 7 m in length. Except for research purposes under licence, foreign vessels are excluded from fishing within territorial limits or on the surface of, and up to 5 nautical miles beyond the edge of the plateaux (continental shelf, defined by the 200 m depth contour). Certain coastal areas on Mahe and Seychelles are closed to fishing. Fishing for demersal species by means of trawl nets is prohibited. Regulations specific to demersal fish species do not exist except in as much as they are included incidentally in other regulations, e.g. the restriction of mesh size in the trap fishery which occasionally catches a number of demersal species.

Thus, whilst the instruments exist with which to manage the demersal fisheries, at present there is little direct management, and no specific demersal fishery management plan exists. The most relevant plan is that for mothership line fishing ventures (Harris, *et al,* 1994) which proposes restrictive licencing (numbers of licences and locations permitted to fish), allocation of (aggregate multispecies) catch quotas, and designation of closed areas (as fishery controls / to permit recovery of depleted areas). The number of licences for outboard powered vessels are also restricted to limit effort on nearshore coastal resources including demersal resources.

■ Mauritius

In order to meet the objectives of increased local frozen fish production Mauritius has developed the appropriate infrastructure including port facilities and an efficient cold chain. At times in the development of the fishery, however, infrastructure developments have not maintained pace with the fishery, limiting catches and viability of operations (SWIOP/ MAFNR, 1989). To stimulate the industry various mechanisms have been employed including concessional harbour dues, reduced duties on items used in freezing fish and on fuel, and concessional loans. To meet the objective of providing an affordable source of protein price controls have been employed. They were relaxed for the period 1981-1987 but are presently in place under regulations in the Supplies Control Act. Price controls have limited the profitability of fishing operations, but have also acted as a management tool in reducing fishing effort. Opinions are divided as to the utility of price controls as a fisheries management instrument (SWIOP/MAFNR, 1989).

Direct management instruments applied to the banks fishery are indicated in the Mauritian Fisheries Legislation and include :

- a prohibited fish list (Fourth Schedule of legislation). This relates to the potential for ciguaterra (fish poisoning) in certain species. On the southern Mauritian banks it effectively limits fishing to shallow water, and contributes to the mono-specific nature of the catch. In the multispecies Chagos fishery it leads to species avoidance and discards with associated problems for resource assessments.

- minimum size limits (Fifth Schedule), 30 cm for Dame berrie (*L. mahsena*). This regulation principally relates to the coastal artisanal fishery, and it is not clear that it relates to the banks fishery or that it is the same species (local names are used in the legislation).

- a catch quota system, introduced in 1994 and modified in 1995 (reviewed in MRAG, 1996b). The system is implemented through Individual Transferable Quotas (ITQ's). The Total Allowable Catch in 1995/6 was set at 4,750 tonnes of all species. The TAC will be reduced annually until evidence of recovery of the resource is indicated. The quota is subdivided between fishing companies and by fishing banks.

- limited entry. Cabinet recommended in 1994 that the number of vessels be limited to the present 17, no new vessels would be licenced, those dropping out of the fishery would not be replaced.

Chagos Archipelago

The principal legal instruments for the control of the BIOT fisheries are the BIOT Fisheries (Conservation and Management) Ordinance, 1991 and the Fishing Regulations, 1993. Management of the fishery is achieved through restrictive licensing (vessels and gear) and closed areas. The Director of Fisheries may also specify Terms and Conditions of licencing which include all radio and logbook reporting requirements, and the requirement to carry scientific observers (BIOT, 1995). With respect to the demersal banks fishery the following currently apply :

- A maximum of 6 mother-vessel licences each for a maximum duration of 80 days may be issued (based on a typical mother-vessel carrying 20 dories). Licences are issued only during the traditional fishing season in Chagos, end of April-September.
- Fish may not be caught by any fishing method other than : hook and line; hand-held cast nets strictly for the purposes of catching bait
- Fishing within lagoons is prohibited
- Fishing within closed areas, as may be specified from time to time, is prohibited.

Additionally it has been proposed that the waters around the islands of Peros Banhos, Salomons and Egmonts be designated as Marine Protected Areas. The primary reason for this is to protect the fragile reef and lagoon systems around these islands, and potential benefits to fisheries on other banks in the Archipelago, for example from spillover effects, are considered to be small. However, catches and catch rates at these locations are also small so the impact on the fishery will be minimal. As an effective fisheries management tool closed areas need to be located on the principal fishing banks.

6.3 Predicted effects of different management strategies

6.3.1 Effort allocation

One of the most common management methods is the allocation of effort so that the maximum sustainable yield (MSY) can be extracted from the fishery. A number of methods exist for estimating MSY, from simple rules of thumb to very complex and data-hungry estimation techniques.

In Section 6.4, yield effort curves are shown for each of the major species. These were derived by calculating the equilibrium yield for various fishing mortalities. The magnitude of the maximum potential yield and the fishing mortality at which it can be obtained may easily be estimated from the curves.

The way in which these curves change shape depending on various biological parameters was discussed qualitatively in Section 5.2. The position and size of the maximum is also affected, and a number of authors have quantified the effect of certain biological parameters on MSY and F_{msy} , e.g. Beddington & Cooke (1983). In Kirkwood *et al* (1994), the dependence of MSY on M, M/K, Lc, Lm and density dependence in the stock recruitment relationship is discussed. Those latter results would suggest that, given the typical parameters of tropical reef and bank species, the MSY should be about 0.3M times the virgin exploitable biomass (ExBo). This is borne out by the graphs in Section 6.4, which show the MSY of the deep slope species (typically M=0.25) to be around 0.08 ExBo, and that of the banks species (M=0.5) to vary around 0.15 ExBo.

The above authors have also discussed the effort at which the MSY is obtained, and Polovina (1987) proposed the following simple rule for setting effort levels:

- if Lc50<Lm50 then set F=M
- if Lc50 $\geq$ Lm50 then set F=2M

where Lc50 and Lm50 are the 50% lengths at capture and maturity respectively.

The influence of Lc and Lm on selecting an appropriate F in a multi-species fishery will be discussed in the next section.

Given that it is theoretically possible to estimate the MSY and choose a fishing mortality for each individual species, an important consideration is how best to apply these rules to multi-species fisheries. Each species' catchability could, in theory, be used to calculate the optimal level of effort for each species. This would, however, only be useful if effort could be regulated on a species basis. Where effort is uniformly applied, the level chosen will be appropriate for some species and less so for others.

Figure 6.1 shows the effect of catchability difference on the composition of the catch of two otherwise identical species subjected to uniform effort. The two species have the same biology and thus the same F_{msy}, so where the catchabilities q and q' are the same, the catch biomassess are equal. As the relative catchability ratio q/q' changes, the catch composition also changes.

The solid line in the figure corresponds to the level of effort that will result in F'=M' for the base species (catchability q') and for the other when q=q'. This is the fishing mortality for both species (M=M'). At this effort level, the yield shows reasonable tolerance to small variations in catchability in the second species. There is greater variation in yield where the effort is over or under the ideal level. This graph is appropriate for all stocks with M/K=2, and so applies to both Indian Ocean banks and Pacific deep slope species.

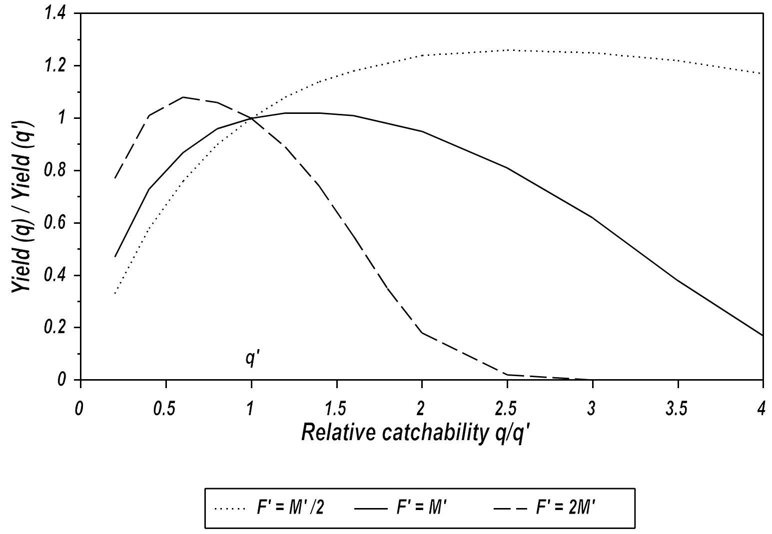

Figure 6.1 : Effect of relative catchability on catch composition

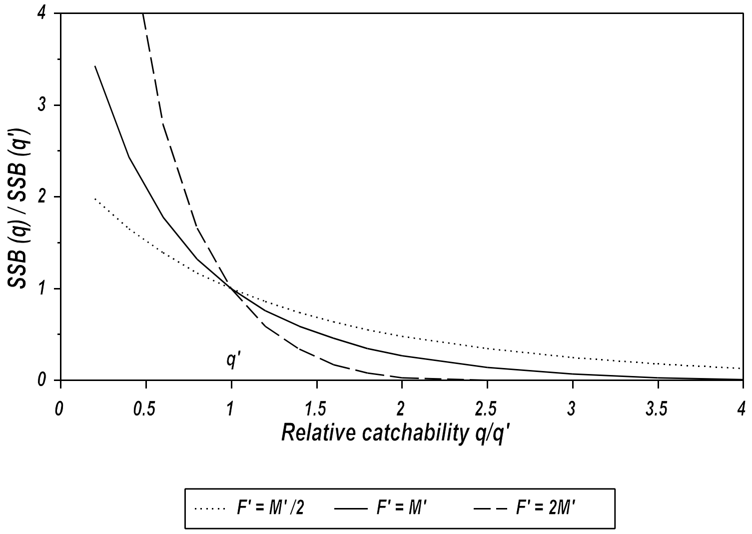

Figure 6.2 : Effect of relative catchability on species composition

Figure 6.1 can also provide some insight into this sensitivity where the two species in question are different. Suppose our second species has biological parameters requiring a different optimal fishing mortality. For example, if M=2M', then the best effort would lead to a fishing mortality F=2F'. Fishing at the previous best level F'=M' would be equivalent to fishing the second species at F=½M. The graph showing sensitivity to catchability would therefore have the shape of the dotted line in the figure, corresponding to F'=½M'. The yield biomass ratio of the two species for q=q' would not be equal to 1, as it was before, but the figure still accurately represents potential deviations from that initial biomass ratio. Clearly, as far as yield is concerned, the most vulnerable species are those which require a lower F than the species for which effort is optimised and which have a higher catchability. Yield will also be limited, though less so, for those species which could support a higher F but have lower catchability.

The above relates to composition of the catch, but is, of course, not a reflection of the composition of the biomass. Figure 6.2 illustrates the extent to which catchability differences impact on the species composition of the resource. Increased catchability has a significant impact on biomass, mitigated only where the acceptable mortality rate is higher than the applied one.

A general guideline that follows from the above is that managers should estimate:

I) the effort level $E_{sp1} = F_{opt}/q$ for the species with the lowest F_{opt}. Guidelines for finding F_{opt} (the most appropriate F for a species) and for likely candidates for the species with lowest F_{opt} will be discussed in the next section.

ii) the effort level $E_{sp2} = F_{opt}/q$ for the species with the highest catchability.

A conservative guideline designed to maximise total biomass while protecting the most vulnerable species in the resource would be to apply the lowest of these two efforts to the fishery, i.e.:

$E = \min(E_{sp1}, E_{sp2})$

The above is a guideline for a situation where only overall effort can be managed. Targeting will be reflected in the relative catchabilities, but here it is assumed that it is not possible to change the targeting or set species catch or effort limits. If the difference between E_{sp1} and E_{sp2} above is large, applying the above guideline will provide the best security against overfishing, but it will leave some portions of the resource under exploited. Measures to decrease the catchability of the first species or increase the catchability of the second would improve the utilisation of the resource.

If for economic reasons it is acceptable to over exploit the most vulnerable species in order to improve catches of other ones, then, with the help of its yield-effort curves, F_{opt} for this species may be increased above its optimum level, with the awareness that this might endanger that stock. In such a case, estimates of biological parameters such as Lm and SRR for this stock become more important. A better approach would be to encourage different targeting patterns to change the relative catchabilities of the various species.

6.3.2 Size specification

Another important means of management is by setting the minimum length at which fish may be caught. The practicalities of such management for tropical reef banks and deep slope fisheries was discussed in Section 6.1. Selection of an appropriate minimum length at capture for a species is largely dependent on the length at which the fish become mature, and so any discussion of length issues must necessarily encompass both of these points. In this section, as before, the abbreviations Lc and Lm are used to denote $Lc50/L_\infty$ and $Lm50/L_\infty$, the 50% lengths at capture and maturity respectively, expressed as ratios with the asymptotic length L_∞.

Suppose a stock has Lm=0.6 with a 10% ogive, i.e. Lm75=Lm50+10%, and is fished from Lc=0.4, also with a 10% logistic selection ogive. The proportion of immature fish in the catch, by length, is illustrated in figure 6.3. There is a large proportion of the catch which has never matured and so cannot contribute to the next generation. The closer Lm and Lc become, the lower this proportion becomes, and the shorter the span of lengths which are affected.

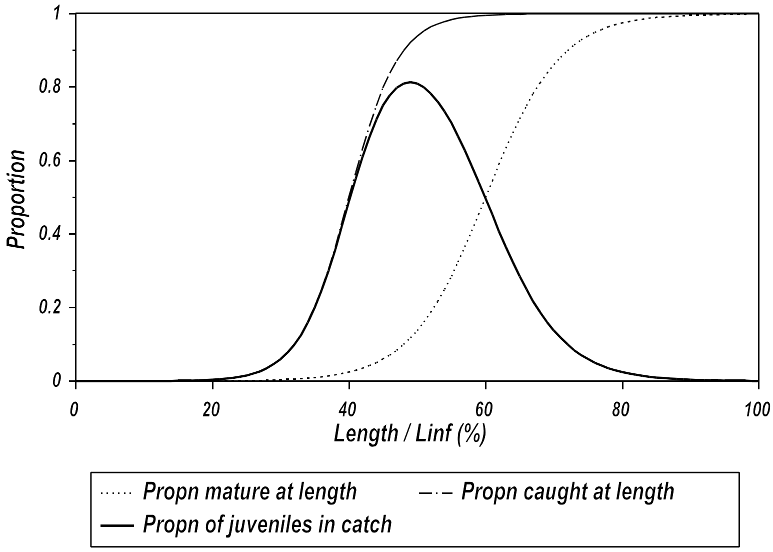

Figure 6.3 : Distribution of immature fish in the catch

Clearly, the most interesting parameter in this discussion is the ratio Lc/Lm. However, the magnitude of this ratio has varying significance depending on the specific value of Lm. As might be expected, fish that mature early, are more tolerant of Lc 10% below Lm than those that mature late would be. This relationship is encapsulated in figure 6.4.

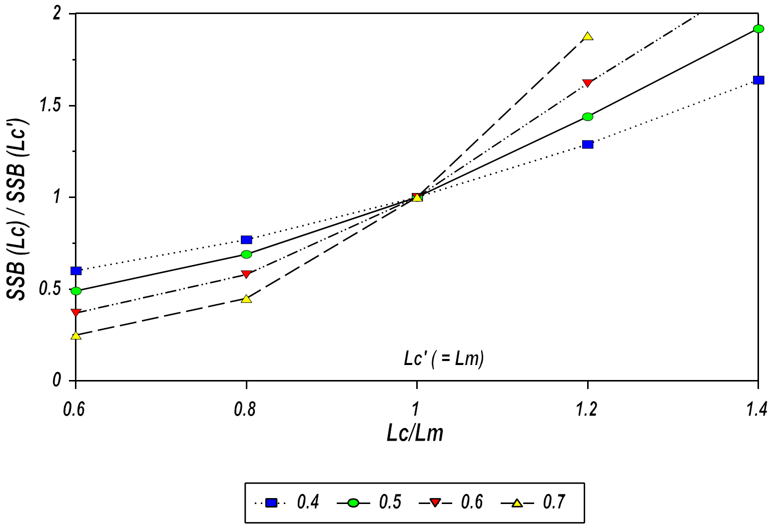

Figure 6.4 : Variation in adult biomass due to change in Lc, for different values of Lm; F=M.

Each line represents a specific Lm, and shows how the adult biomass is changed by variations in the relative value of Lc, under fishing pressure F=M. In fact, the relationship between Lc and Lm is symmetrical, so changes in Lc/Lm produced by changing Lm in the simulations produced the same biomass ratios. This relationship is constant for constant M/K, and therefore relevant to both study sites.

In this and later calculations, Lm and Lc have both been given knife-edge selections. As discussed in Section 5.2.3, the effect of the ogive is small, and is insignificant to these comparisons.

The above graph may also be read as showing the biomass ratio that would result from simultaneously fishing two identical species with different Lc, for example two species with similar parameters but different mouth shapes which therefore exhibit different hook selection. This issue will be discussed further later in this section.

Figure 6.5 shows in an analogous manner how the relative yield of two such species is influenced by the ratio Lc/Lm. This also depicts the error that might be made in predicting yield, if Lc has been incorrectly estimated.

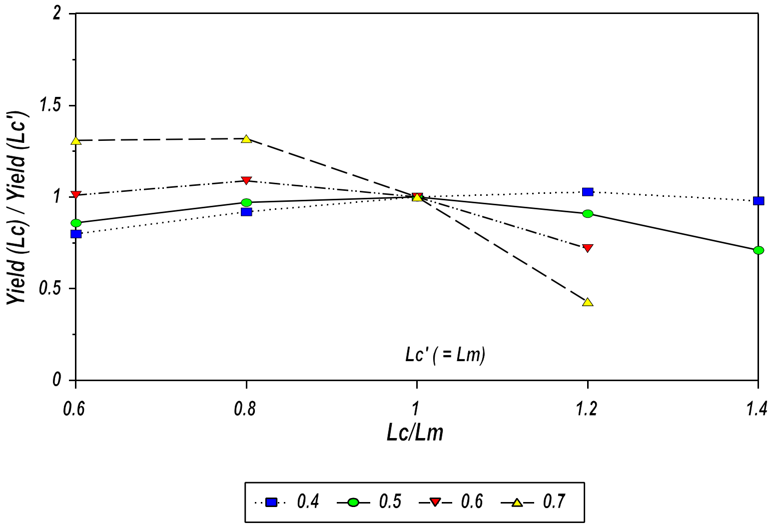

Figure 6.5 : Variation in yield biomass due to change in Lc, for different values of Lm; F=M.

The figure shows that there is generally little or no benefit to be had from setting Lc greater than Lm. In fact, where Lm is high, this can result in loss of yield as exploitable biomass is reduced.

The Indian Ocean banks species studied all have high length at maturity, with Lm=0.6 to 0.7. (Data was not available from which to estimate Lm for deep slope species.) With such high Lm, there is no discernable benefit to or loss from having a lower Lc, at these effort levels. However, as effort increases, reduction in yield due to low Lc/Lm becomes significant, as shown in figure 6.6.

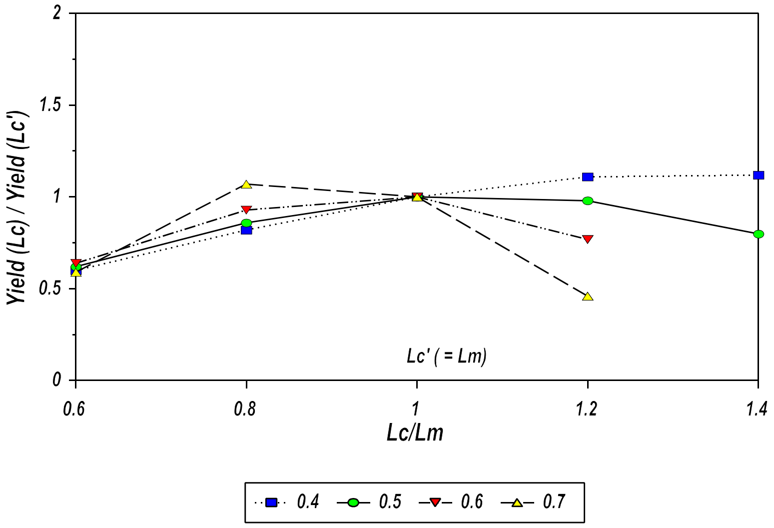

Figure 6.6 : Variation in yield biomass due to change in Lc, for different values of Lm; F=2M.

■ Coping with Lc

A closer look at figures 6.5 and 6.6 suggests another, potentially useful relationship. Consider first the case where F=M. If Lm=0.5, clearly the most appropriate Lc is Lc=Lm=0.5. Where Lm is 20% lower than this, Lc=Lm+20%=0.5 provides a higher yield. For Lm=0.6, the 'best' yield from an admittedly coarse graph is at Lc=Lm-20%=0.5, and similarly for Lm=0.7, Lc=0.5 provides higher yield. It seems that for F=M, Lc=0.5 is appropriate, irrespective of Lm. Considering further the case where F=2M, the same pattern emerges, with the higher Lc=0.6 seeming more appropriate for this effort level.

Such an effect would have important implications for the case study regions, where Lc is often difficult to regulate. It suggests a criterion for setting F, based on estimates of current Lc. By reversing the argument of the last paragraph, if Lc=0.5, the best F is approximately F=M, whereas if Lc=0.6, then F=2M is allowable. If this criterion is followed, it would not be necessary to estimate Lm in order to maximise yield to within a reasonable tolerance.

An important point here is that it is not practical to manage these fisheries by means of catch quotas, as discussed in Section 6.1. Rather, effort needs to be limited by one of a number of possible means. This implies that it is more important to estimate appropriate levels of effort to maximise yield than to predict the precise amount of yield that will be obtained. It seems that knowing the value of Lm is more important for estimating yield (and adult biomass levels) than for setting suitable effort.

In order to investigate the above rough argument more thoroughly, consider figure 6.7. This shows the equilibrium yield as a function of relative fishing mortality for various values of Lc, where Lm=0.6. Yield is expressed as a function of exploitable biomass, where, for consistency of comparison, exploitable biomass is defined as the biomass of fish exceeding the minimum length of capture, set at Lcmin=0.4. Fishing pressure is defined in terms of the ratio F/M. This figure relates to all species with relatively low M and M/K=2, which means that is will be appropriate for all case study species with Lm=0.6.

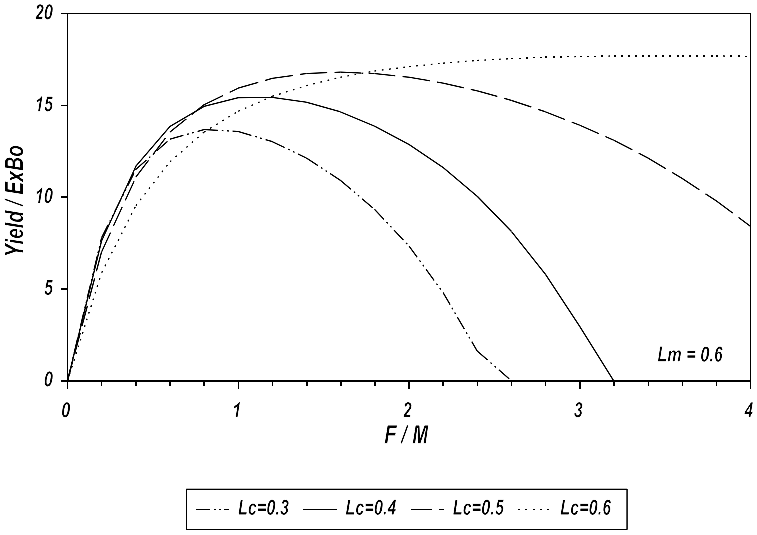

Figure 6.7 : Equilibrium yield - effort curves for various Lc; Lm=0.6.

For each value of Lc, F_{msy} can be read from the figure. For example, where Lc=0.4, the MSY is at just over F=M. Now compare these curves with those in figure 6.8, where Lm=0.7.

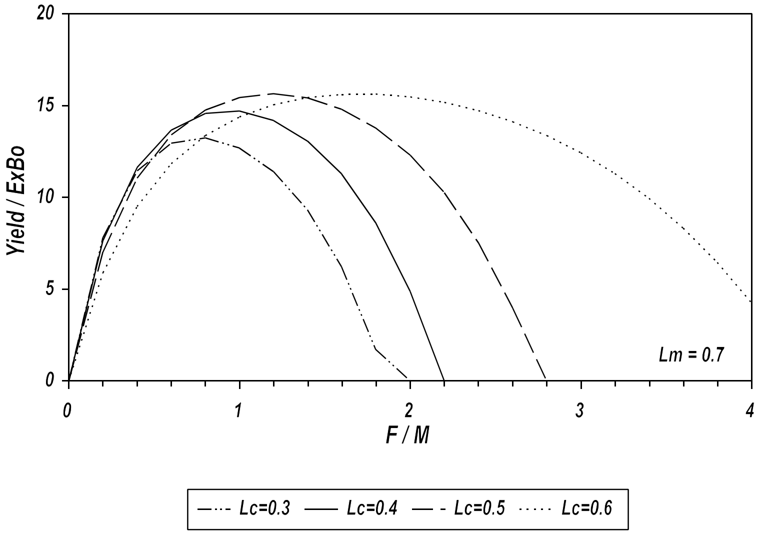

Figure 6.8 : Equilibrium yield - effort curves for various Lc; Lm=0.7.

The MSYs have decreased by a few percent of ExBo, and the F_{msy} has decreased to just under

F=M. In general, however, the effort level at which these maxima occur has not changed greatly between the two graphs. The major difference is at the highest Lc, where F_{msy} is much higher for Lm=0.6 than for Lm=0.7. A similar set of curves can be plotted for Lm=0.5 and, though they are not shown here, they continue the trend set by the other two. Figure 6.9 summarises the range of F_{msy} values calculated, for Lm ranging between 0.5 and 0.7.

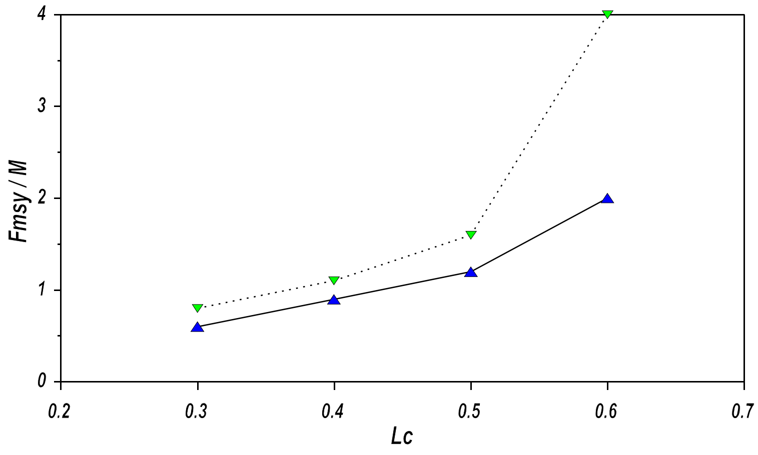

Figure 6.9 : Appropriate fishing pressure for a species with a given Lc

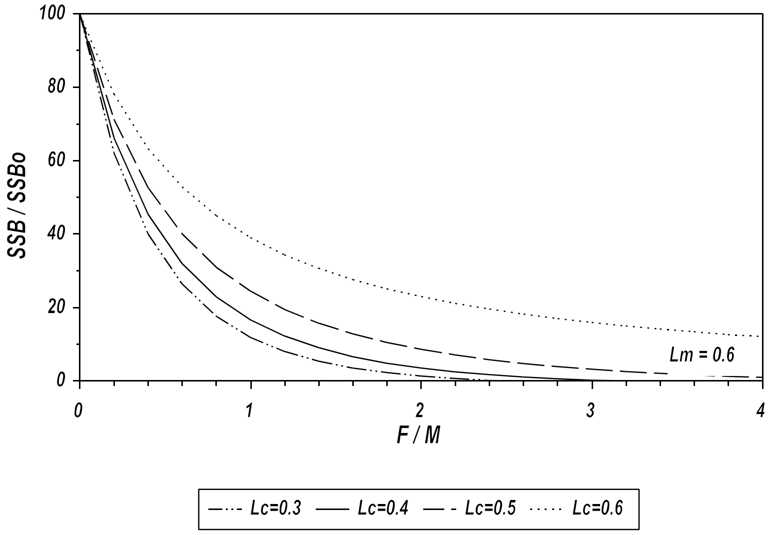

Figure 6.10 : Equilibrium spawning biomass for various Lc; Lm=0.6.

The importance of figure 6.9 is that, for a given Lc, it shows the range of values which F_{msy} might have, depending on the usually unknown quantity Lm. It defines the sensitivity of the F_{msy} to Lm at that Lc. However, it can also be asserted that there is no great loss in yield over the range of these Lm's, so there is no loss in applying the precautionary strategy of setting the desired fishing mortality F_{opt} to the lowest of these lines. In fact, there is a significant biomass benefit to doing just that.

Figure 6.10 shows the spawning biomass as a function of fishing pressure for various Lc, where Lm=0.6. For each Lc, the lower F from figure 6.9 results in an SSB of approximately 20%SSBo, whereas the higher F leaves the SSB much lower. It is a general recommendation in fisheries management that SSB should not be allowed to drop below 20% of virgin levels. By using the solid line in figure 6.9, the maximum yields are ensured while preserving safe levels of spawning biomass.

There is scope for further simulations to fully quantify this relationship, and its dependence on other biological parameters, especially density dependence in the stock recruitment relationship. However it is felt that it will be both applicable and useful for the tropical demersal reef bank and deep slope species represented in this study.

In summary, then, it is not necessary to measure the length at maturity Lm for the species to be managed. It is important, however, to measure Lc. Given that Lc cannot easily be changed, the measured value of Lc can be used to set the maximum fishing mortality that should be applied to the species. A word of caution is appropriate for new or lightly exploited fisheries. Lc is often initially high, because of greater catchability and abundance of older, larger fish. It has been shown that Lc decreases over time (see Section 5.1), and as it decreases, the desirable effort levels will decrease. Reducing effort is hard to do, however, so it would be wise, for a relatively new fishery, to try to estimate where Lc is likely to end up. A relevant future study would be to model the change in Lc as the age structure of a population changes under fishing pressure. In the absence of such a model, it is recommended that where current Lc exceeds 0.5, F is chosen according to Lc=0.5 until it is certain that Lc is relatively stable.

Where a number of species are to be managed with one rule, this guideline can be combined with that of the previous section to determine overall effort levels. This means that it is not necessary to select an F_{opt}, or estimate Lc, for all the species involved. Referring to the previous guideline, the F_{opt} for the most catchable species is needed, as well as for the species for which F_{opt} will be lowest. A few may need to be estimated for this latter case, but the calculations can be targeted at those species with low M, low K, high L∞, long lifespans or small Lc.

This work also suggests that, if species are to be combined into guilds, it is most appropriate to pool data from species which have similar Lc, catchability and M/K. Tropical reef banks and deep slope species all appear to have M/K suitably similar to one another. Where different species are distributed homogenously, school together and have similar size, they are likely to have similar Lc and catchability, as they will react similarly to the gear. Such fish may comfortably be treated as a guild, and catchability measured for only one representative species, or for the guild as a whole.

■ Setting Lc

In the previous section the point was made that it is largely impractical to control Lc by gear restrictions in tropical demersal hook-and-line reef fisheries. Nevertheless, it is interesting to consider the effect of using a single gear specification on a multi-species fishery. Recommendations can be derived which may be of relevance for gears where the length at first capture is under more direct control of management.

If a number of species of different sizes co-exist, then a gear which takes fish of e.g. 40cm may be catching one species at Lc=0.3 and another at Lc=0.6 (as before, Lc is the length at first capture Lc50 expressed as a fraction of L∞). Unless Lc can be optimised individually for each species, some species are sure to be fished suboptimally for any given gear. As discussed on page 133, controlling Lc is not feasible for these fisheries.

Figures 6.11 and 6.12 show the effect of a fixed length at first capture on two baseline species which differ only in their L∞ values. Initial biomassess are the same, and effort is distributed equally between the two species. Figure 6.11 represents the deep slope case, and the two species have L∞ = 100cm and L∞ = 160cm respectively. If the best Lc for each species is Lc=Lm=0.5, this would mean species 1 should ideally be caught from 50cm onwards, and species 2 from 80cm. A truly multi-species management would enable this to be implemented. The total yield biomass of both species as a function of effort for this ideal situation is depicted by the solid line in the figure. The yield is expressed as a percentage of the total biomass of both species in the unexploited state.

If, instead, the minimum size is set at 50cm for all fish, this would correspond to Lc=0.5 for species 1 and Lc=0.31 for species 2. This is the dotted line in figure 6.11. For low effort, the yield is the same as the previous case, diverging significantly after MSY is exceeded. The discontinuity in the graph indicates the point at which the larger species is exterminated and only the smaller remains to be fished optimally.

In similar vein, a minimum size of 80cm would correspond to Lc=0.8 for species 1 and Lc=0.5 for species 2. The dashed line in the figure shows the total yield for this case. At low effort, the yield is rather lower than in the other two cases. It is only at F>2M that this option becomes preferable to the smaller size threshold.

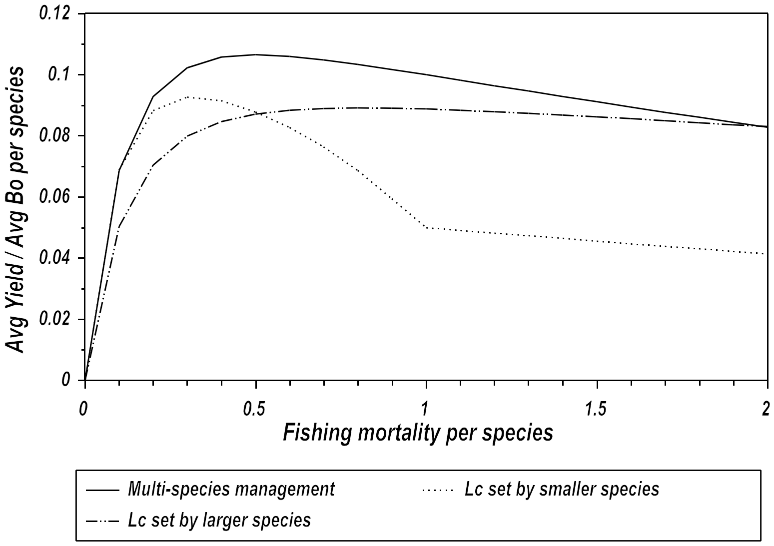

Figure 6.11 : Effect of Lc management on the total-yield:effort curve for two baseline deep slope species with different L∞.

Figure 6.12 shows the analogous situation for the baseline banks species. The two L∞ values were 60cm and 100cm. Again, it is preferable to set the size limit according to the Lc50 of the smaller of the two species where fishing mortality is less than 2M.

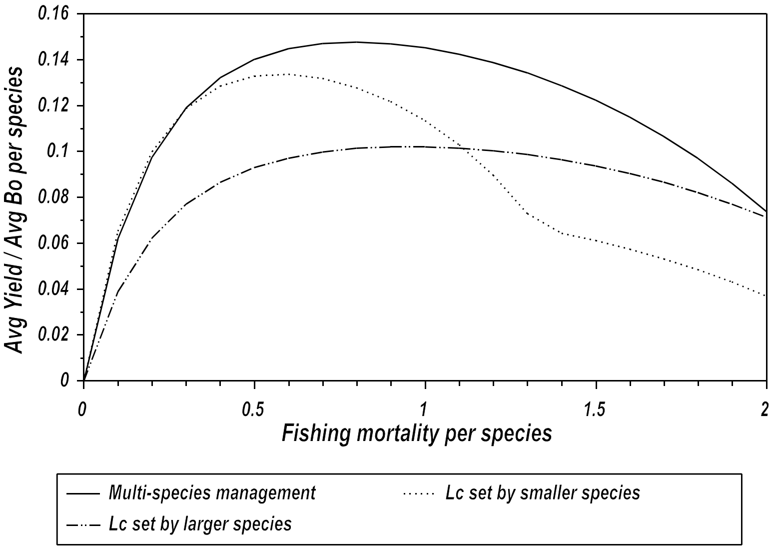

Figure 6.12 : Effect of Lc management on the total-yield:effort curve for two baseline banks species with different L∞.

It is interesting to note that choosing the lower Lc50 for F<2M and the higher one for F>2M also ensures that the species composition of the catch does not deviate more than about 20% from initial levels. Figure 6.13 shows the proportion of the smaller species in the deep slope catch at equilibrium for various levels of fishing mortality, for the three cases discussed above. It is clear that the size limit can be used to shift the catch composition towards a more desirable species, but this is done at the expense of some yield biomass, and could threaten the other stock.

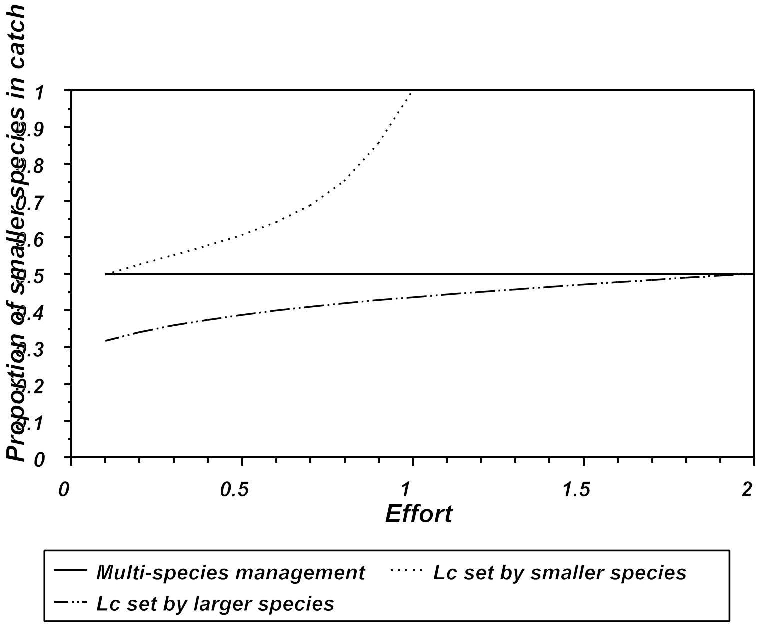

Figure 6.13 : Proportion of smaller species in the deep slope catch

These results can be generalised to more species with different parameters. By setting the size limit to that appropriate for the *smallest* species, and the F to that appropriate for the longest lived one (often the *biggest*), yield would be maximised. If overall F exceeds 2M for the longest lived species, then the overall size threshold should be increased to the appropriate level for that species (i.e. its length at maturity), or it may become endangered.

6.3.3 Resource manipulation

In the above discussion, only issues relating to total yield biomass and stock biomass have been taken into account. Price differences will affect the desirability of certain species, although generally this will be reflected in the relative catchability of the species, which is partly determined by fishermen's targeting effort.

Certainly, the choice of species to be considered in setting overall effort levels will be influenced by price. It may well be decided to sacrifice some yield or population biomass of a species in order to optimise effort for a more valuable one. The guidelines developed above will allow managers to assess the risk of such a tradeoff.

There has been some suggestion that it may be worthwhile to fish predator species heavily in order to improve the catch of their prey. Section 5.2.4 indicates that this is unlikely to provide measurable benefit to the fishery.

6.3.4 Time and area controls

- **Closed areas**

If a species is fairly sedentary, then closing an area of the fishery would protect a portion of the spawning biomass, ensuring recruitment for the following year. The portion of the stock that is exposed to fishing would still have the biomass reduction that was calculated in the previous sections, but it is assumed that low resultant recruitment would be supplemented by larval migration from the closed area. In Section 5.2.2 it was shown that deep slope and banks stocks in a closed area would recover at rates of approximately 8% and 12% respectively per year.

In modelling terms, the effect of this is equivalent to increasing the density dependence in recruitment of the stock. Unfortunately, with the data that is available it is impossible to relate the benefit of a closed area to the shape of a stock recruitment relationship. The effect of density dependence on the yield-effort curves was demonstrated in figures 5.6 and 5.5. For higher values of d, yield at effort increases, although this is most significant where fishing mortality is high. The primary advantage of a high d is that robustness to overfishing is improved. Increased yields would have to be sufficient to offset the loss of yield from the closed area, but the current model is not suitable to quantify these effects.

The current SRR assumes that as long as the SSB is above 20%SSBo, deterministic recruitment will be within 10% of virgin levels. Even the most severe SRR considered, d=0.6, would only cause a 20% reduction in recruitment at this SSB level. The guidelines given in the previous two sections are designed to ensure that this criterion is met, or nearly met, for all the commercially important species in the fishery. Thus, if these guidelines are implemented, closed areas are unlikely to provide a significant advantage.

■ Closed seasons

Simulations showed that applying a proportionally greater amount of effort during a proportionally smaller part of the year made very little difference to MSY, F_{msy} or $SSB_{msy,}$ even when that part of the year was timed to avoid spawning periods. Once again, this is because the stocks recruitment is density dependent so recruitment is not generally affected by spawning population size, except at very low adult biomass levels. Unless effort is very high, such levels will not be reached.

In doing these calculations, a fixed amount of effort was merely reallocated to different times of the year. This is based on the assumption that fishermen would make up for the loss of fishing time by applying more effort during the periods when they may fish. If, however, it is possible to use a closed season as a means of reducing effort, then there may indeed be benefits to the fishery in doing so.

■ Pulse fishing

The previous two subsections argued that closed seasons have no particular benefit to a single species except as a means for limiting effort. This section addresses the effect of pulse fishing, which is essentially a combination of a long closed season and closed area, on long term trends in species composition.

Consider a deep slope stock (or two stocks) distributed over two seamounts, suddenly subjected to a low level of effort (F=0.15) distributed equally between locations. The total yield (as a proportion of total virgin exploitable biomass) declines from initial levels with time, as shown by the solid line in figure ?. If instead, all the effort (F=0.3) was directed first at one location, and then the other, total yield would oscillate about this line, as indicated in the figure. The two lines depict different frequencies of rotation between the two sites. On average, in the long term, the yields from pulse and continuous fishing (with the same effort) are the same, but pulse fishing leads to greater variation in yield which is clearly less desirable for the industry.

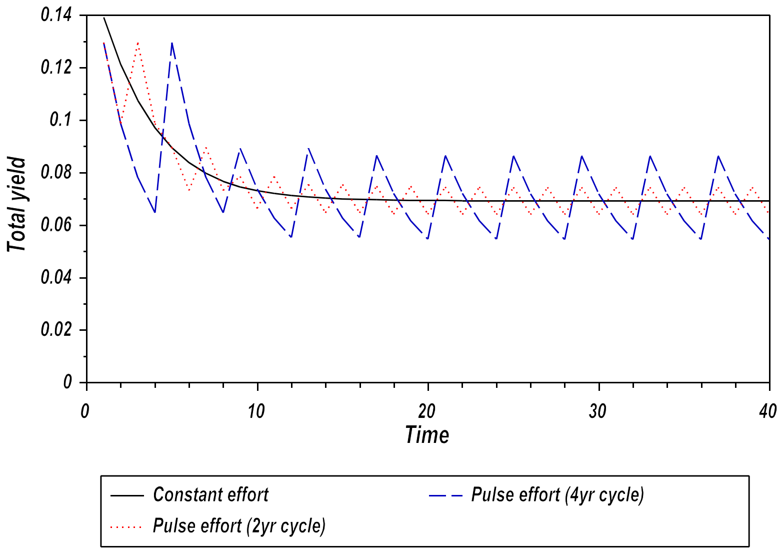

Figure 6.14 : Total yield response to pulse fishing, alternating between two stocks

Pulse fishing does lead to an increase in spawning biomass, but the increase is marginal and also oscillates, as shown in figure 6.15. The above calculations would serve equally well for two biologically interacting species being targeted alternately. It might have been anticipated that, if prey release occurred, some long term drift in species composition could result from pulse fishing. However, simulations showed that this is not the case. The prey release increased the size and changed the pattern of the oscillations in yield and biomass so that they took twice as long to repeat, but the symmetry remained, ensuring that average yield was maintained. The gains in average biomass were still not significant enough to warrant the discontinuity in fishing returns.

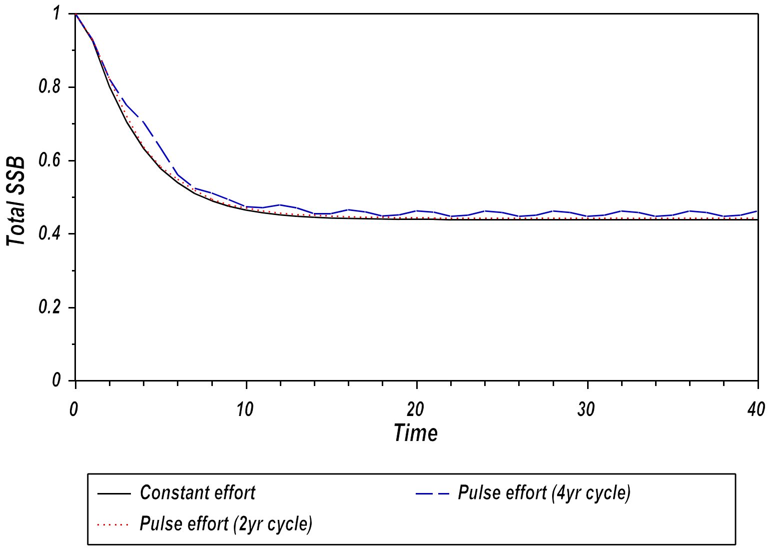

Figure 6.15 : Total biomass response to pulse fishing, alternating between two stocks

Simulations were also performed for shorter periods of intense fishing followed by longer (3 times as long) periods of recovery. The average yield was still not significantly better than would have been obtained with an even effort distribution. The only particular benefit from pulse fishing is that clear depletion will occur, making parameter estimation from stock assessments more reliable.

In the case where the two stocks have different biological parameters or catchabilities, pulse fishing still results in yield and biomass oscillations around the respective levels that would have resulted from equivalent, uniform effort.

6.3.5 Data requirements for biological guidelines

The findings of this study into the predicted effects of management suggest a prioritisation for data collection and research. The fact that such a prioritisation could be made is a result of certain assumptions that could be made about constraints under which fishery would operate. These constraints are appropriate for the tropical demersal reef bank and deep slope species under study, and are discussed in the relevant sections of this report.

- It is assumed that Lc cannot be controlled. The best that can be done is to measure it, and set effort levels accordingly. Therefore, deriving estimates for Lc is a high priority, although it need not be done for all species. When a fishery is new or lightly exploited, the Lc is initially high and drops as the large sizes of fish are removed. In such fisheries where Lc has not stabilised yet, it is wiser to use an estimate of future likely Lc than to use the real measured value.

- It is assumed that catch limitation is impractical. Therefore, it is more important to know the effort range within which yield is maximised, than to be able to predict the maximum sustainable yield that would be so obtained. This means that it is not necessary to estimate Lm.

- The study species all have estimated M/K close to 2. This simplifies many of the relationships studied, and is largely responsible for the simplicity of the guidelines. It is a large contributing factor to the relative unimportance of the parameter Lm. It is therefore important to monitor that this ratio does not deviate significantly from 2. Sensitivity of yield and biomass to the parameter M/K as discussed in Section 5.2.3 suggests that the results are unlikely to deviate greatly for $1<M/K<3$. Further studies are necessary to confirm this, however.

- The above means that M and K will need to be estimated for certain species (see guideline summary). M is also necessary for setting the desirable fishing pressure, since all F's are scaled to M. The only requirement for K is the ratio M/K, but L_∞ will be needed in order to derive Lc, so a growth curve will have to be fitted. The growth parameter t0 is not necessary, so simplified growth curve fitting procedures may be used.

- It is particularly important to have an estimate of the relative catchabilities of species which are to be measured and/or managed as a guild. Similarly, relative catchability of guilds should also be known. Fish which are treated as a guild should have similar catchability and Lc. This does not necessarily require data analysis for all species - where fish are of similar size and habits and are homogenously distributed, it can be assumed their catchabilities are approximately the same.

- Absolute catchability will be needed for certain species.

- Length weight parameters are not very important, and are not required for the implementation of the management guidelines.

- Details of a selection ogive for the gear are not nearly as important as a good estimate of Lc50.

- If an estimate of virgin biomass is available, it will enable the expected yield to be predicted, to within the tolerance represented by unknown parameters such as Lm and absolute catchability.

- It is assumed that effort limits cannot be set by species, so the guidelines presented in this chapter are designed to result in an overall effort limit which will ensure the maximum return from the fishery while protecting all the species within it.

6.3.6 Summary of biological guidelines for management

12. Estimate relative catchability for the major species or guilds (guilds may be comprised of similarly sized species which school together of otherwise present a homogeneous profile to the gear).

13. Estimate Lc and M for the following species:
    - the most catchable one
    - the biggest one (highest $L\infty$)
    - the longest lived (lowest M)
    - the slowest growing (lowest K)
    - any that is caught with a wide range of lengths, particularly juveniles.

14. If the fishery is new or lightly exploited and $Lc > 0.5$, then work with a projected long term value of $Lc=0.5$ until it appears that Lc has stabilised at the higher value.

15. From Lc, estimate F_{opt} for these species, using figure 6.9.

16. Estimate absolute catchability q for:
    - the most catchable species and
    - for the one with the lowest F_{opt} as calculated above.

17. Calculate $E_{opt} = F_{opt} / q$ for the above species.

18. Choose the smallest of the two E_{opt}s and set overall effort

    $E = \min(E_{opt})$

19. This method identifies the various categories of most vulnerable species, and sets effort to protect the most vulnerable one. If economic or sociological considerations place priorities on a certain species, E_{opt} can be calculated for it, and the effect of such an effort on the more vulnerable species can be estimated. Informed choices can then be made about the risks and benefits of overfishing some species in order to optimise yields of others.

6.4 Management of the case study fisheries

The biological parameters derived for each of the major case study species are listed in tables 4.2 and 4.3 on page 49. The level of fishing mortality for certain species over time for Tongan species, and over time at different banks for Seychelles species is illustrated in Figures 5.25-5.31 (see also Technical Reports, MRAG, 1994;1996a;b). For *Lethrinus mahsena* fishing mortality on the Saya de Malha bank was estimated to be in the range 0.2-0.4 (Lebeau and Cueff, 1976; Bertrand *et al*., 1986). Yield effort curves for the most important species are shown in figures 6.16 - 6.28 . Length at capture is expressed as a proportion of L_∞. The figures were derived from the best overall parameter estimates, and using ogive selection rather than knife edge for length at capture and maturity (tables 4.2 and 4.3). Where length at maturity was not known it was assumed to be 0.5 Lmax (Grimes, 1987). It should be noted that variation in these parameter estimates may occur by location. For example, length at first capture varied from north to south in Tonga (Mees and Rossouw, 1995). Effort, and therefore fishing mortality, will vary by location and over time. These figures hence illustrate the general relationship of yield to effort for each species. More detailed analyses are required specific to times and locations for further management advice, as was performed for the example of *P. filamentosus* from Tonga (Mees and Rossouw, 1995) and Seychelles (Mees and Rossouw, in prep.).

The relationship between effort and length at first capture has been described in Sections 5.2.3 and 6.3.2 :

- It is advantageous to keep Lc as high as possible. Increasing length at first capture increased the maximum yield available from the fishery, but greater effort was required to achieve it. At the greatest lengths at first capture, little additional benefit could be gained by relatively large increases in effort.
- The larger the length at capture the greater the resistance to overfishing. Less emphasis need be placed on effort controls.
- Where length at capture was approximately length at maturity, high yields could be achieved, but fishing mortality should not exceed 2M (the natural mortality Polovina, 1987). Although higher yield could be achieved at higher lengths at first capture, considerably more effort was required at little additional benefit, and is therefore not recommended.
- This study indicated that it was not necessary to determine length at maturity, and that the maximum fishing mortality may be determined from knowledge of current length at capture. It is thus possible to establish if present effort levels are appropriate. Where length at maturity is not known, it was suggested that effort should be determined for $Lc=0.5L_\infty$ rather than current Lc values where these exceed $0.5L_\infty$, and that at this point F=M is appropriate.

Current overall estimates of Lc for the case study fisheries are relatively high and effort was generally low except at certain locations during certain years (Table 6.3). This information, and that given in figures 6.16 - 6.28 suggests that overfishing is not generally a problem, but once again it is stressed that in specific locations overfishing could have occurred and that similar assessments are required over time and by location. Furthermore, it should be noted that a particular deficit of the Tongan and Mauritian data is lack of any information on maturity. The real values could suggest a different picture. However, according to the precautionary rule of Lc=0.5, F=M, effort was high in certain years for the two eteline species. In relation to Seychelles, no length frequency information was available for the most heavily fished inshore sector of the Mahe Plateau for which fishing mortality by species is therefore unknown. This was also the case for the Mauritian banks, mortality estimates being obtained for *L. mahsena* from the literature on Saya de Malha bank only.

Pristipomoides filamentosus in Seychelles would appear to be most at threat to overfishing. The length at capture was low relative to the length at maturity and on the banks south of the Mahe

Plateau in 1991 and in the Amirantes in 1994 fishing mortality was high. These correspond to periods of intense fishing by the mothership dory fishing venture. The figures illustrated indicate the equilibrium yield at given fishing mortality. Single years of high effort at lightly fished locations would be expected to produce yields higher than those projected (Fig. 6.24) due to the fishing down of larger old individuals. The resource would only be at threat if consistently fished at these high levels, or if during a single fishing period the spawning stock biomass was severely depleted (to below 20% of unexploited spawning stock biomass). This applies equally to other species in Table 6.3 where it has been indicated that fishing mortality exceeds the optimum. In Seychelles on the Mahe Plateau, it is suggested that effort could in fact be increased despite the low length at first capture (Mees and Rossouw, in prep.) since fishing mortality is currently around 0.3.

Species	Lc as % L∞	Lc as % Lm	Range of F/M	Notes	Suggests fishing beyond optimum
Tonga					
P. filamentosus	0.78	113%	0.3-1.5	Lm=0.5Lmax	
P. flavipinnis	*0.76*	125%	2.2-3.8	Lm=0.5Lmax	
E. coruscans	*0.53*	115%	0.9-2.0	Lm=0.5Lmax	
E. carbunculus	*0.53*	99%	0.7-2.3	Lm=0.5Lmax	Yes
E. morhua	*0.60*	90%	0.1-0.6	Lm=0.5Lmax	
E. septemfasciatus	*0.46*	100%	0.3-0.6	Lm=0.5Lmax	
Seychelles					
A. virescens	*0.71*	97%	0.4-2.3	Amirantes '91 F>2M	Yes
L. sebae	*0.60*	92%	0.8-1.7		?
E. chlorostigma	*0.46*	90%	0.1-1.7	Lm=0.5Lmax	?
P.filamentosus	*0.49*	73%	0.6-2.8	Amirantes '94 and Banks '92, F=2.8M, otherwise F<2M	Yes
L. nebulosus	*0.66*	103%			
L. mahsena	*0.50*	100%		Lm=0.5Lmax	
Mauritius					
L. mahsena	*0.48*	95%	0.9-1.8	Lm=0.5Lmax	?

Table 6.3 : Length at capture as a proportion of length at maturity for principal case study species, and the range of fishing mortality as a product of M the natural mortality (F/M) applied to each species.

It must be emphasised that these calculations are wholly deterministic and so do not take annual environmental variability of the stock into account. It is essential to keep spawning biomass high enough to buffer against years of low recruitment.

For certain other Seychelles species length at capture appears high and could be reduced relative to current fishing mortality (*A. virescens* (Fig. 6.22); *L. sebae* (Fig. 6.23); *L. nebulosus* (Fig. 6.26)). However, it should be recalled that the length frequency data from which length at capture and fishing mortality was derived relates to lightly fished locations and thus some caution should be exercised in suggesting any reductions in length at capture. Certainly the data suggest that at these locations fishing effort could be increased. This relates to the offshore Mahe Plateau and more distant banks. For *E. chlorostigma* from the lightly fished locations however, no increased fishing mortality is recommended at the current length at capture (Fig. 6.25). This is consistent with the observation that depletion of serranids may be occurring even at lightly fished locations (see Section 5.3).

For *Lethrinus mahsena* on Mauritian banks, if fishing mortality is 0.4 (Bertrand *et al,* 1988) then at the current length at capture obtained from the literature, the resource is being exploited at about MSY (Fig. 6.28). Fishing mortality for this species by bank needs to be assessed as a matter of priority. Also, if as has been suggested, the length at capture is decreasing, a fishing mortality of

0.4 would be detrimental and effort should be reduced. The length at capture is $0.48L_\infty$ suggesting that fishing mortality should be approximately that of natural mortality. It is in fact closer to twice that value indicating reduction of effort is necessary.

At the current length at capture of the two eteline snappers from Tonga overfishing is unlikely, and increasing effort should not be detrimental for these species (Figs 6.18 and 6.19). However, lacking an estimate of length at maturity, it should be noted that Lc is $\sim 0.5L_\infty$ and therefore that significant increase in fishing mortality is not advisable. For the serranid species, *E. septemfasciatus* (Fig. 6.21) and *E. morhua* (Fig. 6.20), however, depletion, and indeed elimination of stocks is possible with increasing effort, unless length at capture increases. However, up to 1991 fishing mortality for these two species was apparently low. Continual monitoring of length at capture and fishing mortality on these species is advisable. For the *Pristipomoides* species, length at capture for *P. filamentosus* (Fig. 6.16) could be reduced in order to improve yields, provided effort remained at current levels (see also Mees and Rossouw, 1995). Fishing mortality on *P. flavipinnis* (Fig. 6.17), however, has been high, and while at current length at capture this poses no threat, reduction in length at capture is not advisable.

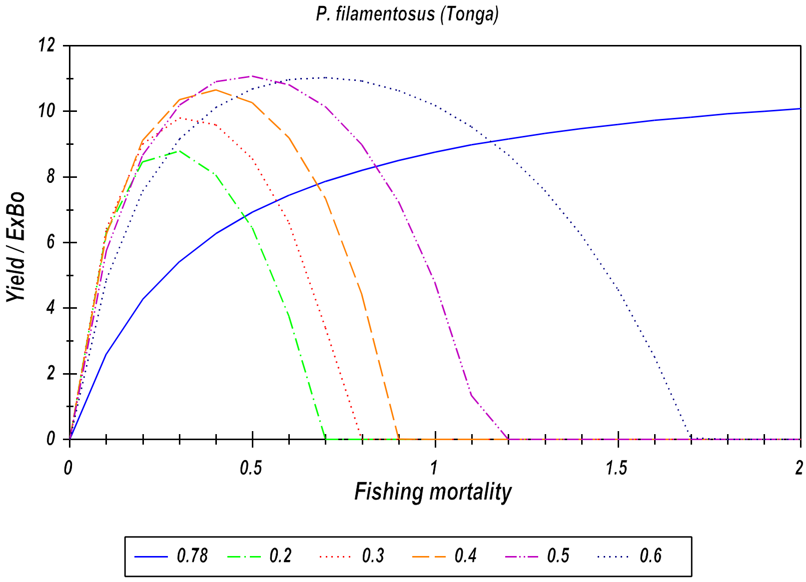

Figure 6.16 : Equilibrium curves for Tongan deep slope stocks of *P. filamentosus* with different lengths at first capture (Lc), including the current estimate (solid line). Estimated F ranged between F=0.1 and F=0.6 during 1987-1991.

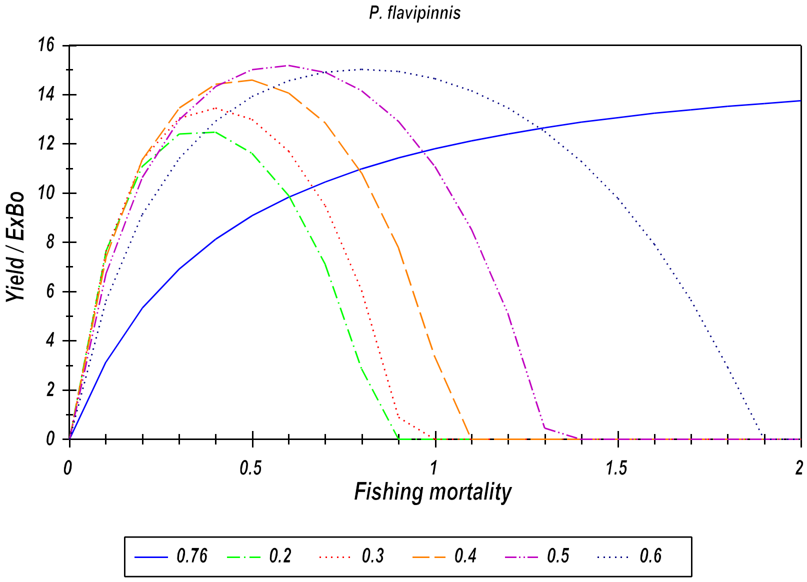

Figure 6.17 : Equilibrium curves for *P. flavipinnis* with different lengths at first capture (Lc), including the current estimate (solid line). Estimated F ranged between F=1.1 and F=1.9 during 1987-1991.

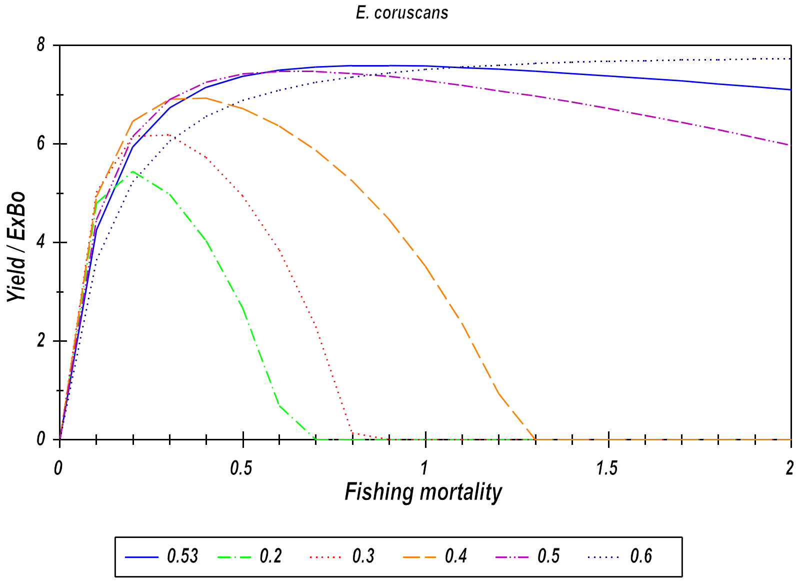

Figure 6.18 : Equilibrium curves for *E. coruscans* with different lengths at first capture (Lc), including the current estimate (solid line). Estimated F ranged between F=0.2 and F=0.5 during 1987-1991.

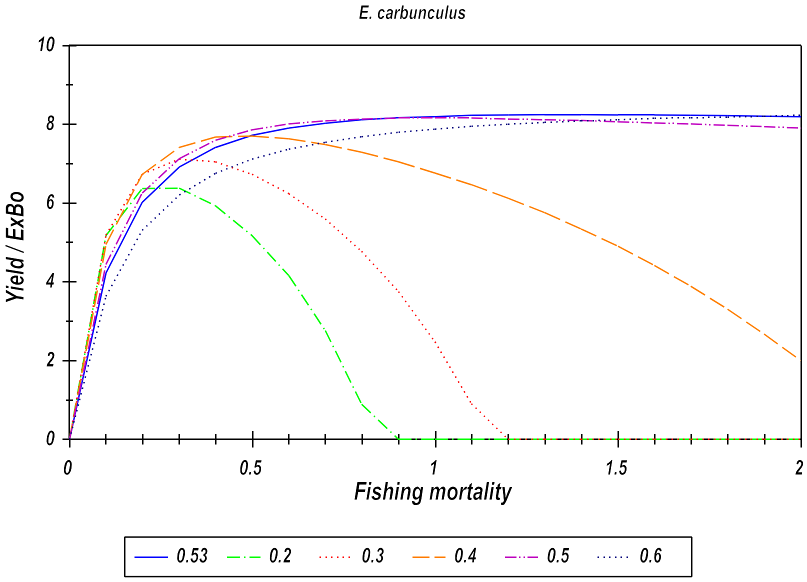

Figure 6.19 : Equilibrium curves for *E. carbunculus* with different lengths at first capture (Lc), including the current estimate (solid line). Estimated F ranged between F=0.2 and F=0.6 during 1987-1991.

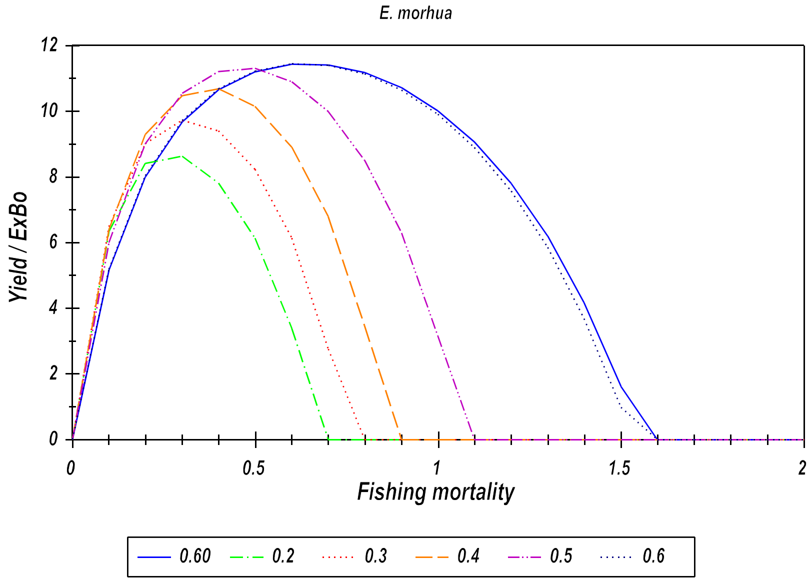

Figure 6.20 : Equilibrium curves for *E. morhua* with different lengths at first capture (Lc), including the current estimate (solid line). Estimated F < 0.2 during 1987-1991.

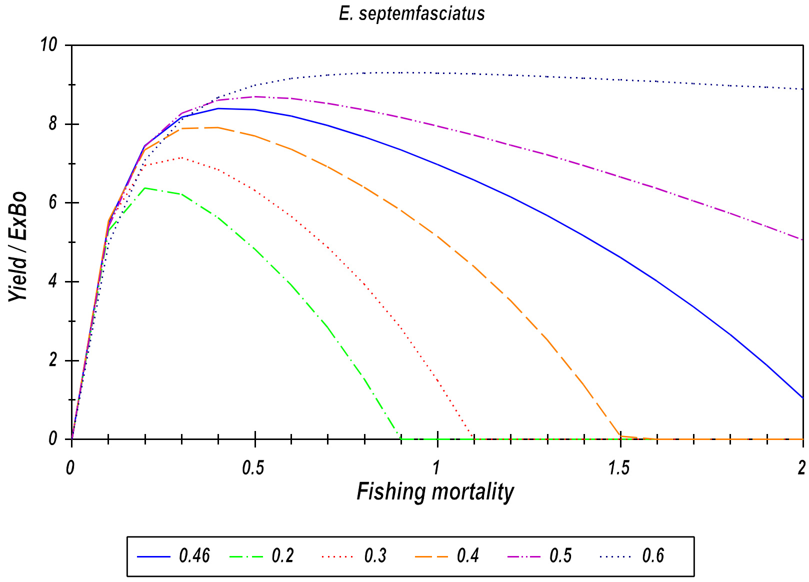

Figure 6.21 : Equilibrium curves for *E. septemfasciatus* with different lengths at first capture (Lc), including the current estimate (solid line). Estimated F ranged between F=0.1and F=0.2 during 1987-1991.

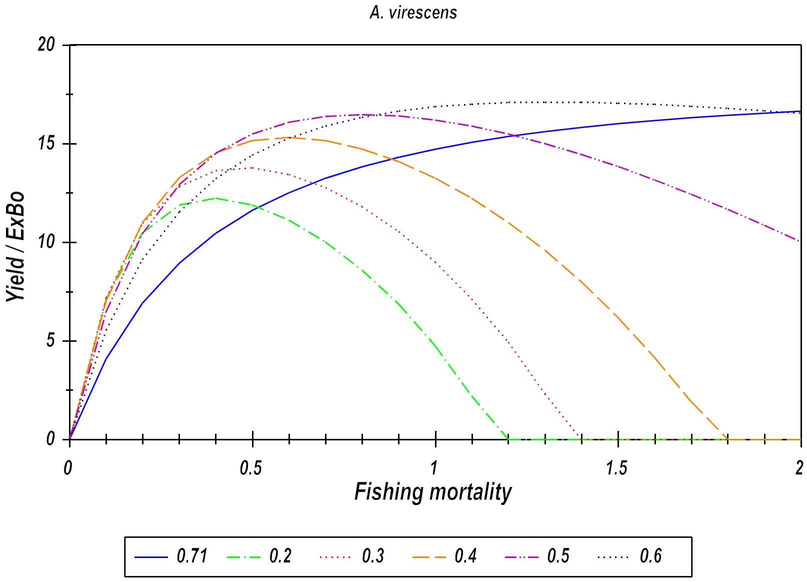

Figure 6.22 : Equilibrium curves for *A. virescens* with different lengths at first capture (Lc), including the current estimate (solid line). Estimated F on the Mahe plateau ranged between F=0.9 and F=1.3 during 1991-1993. For other areas, F~0.2 more recently.

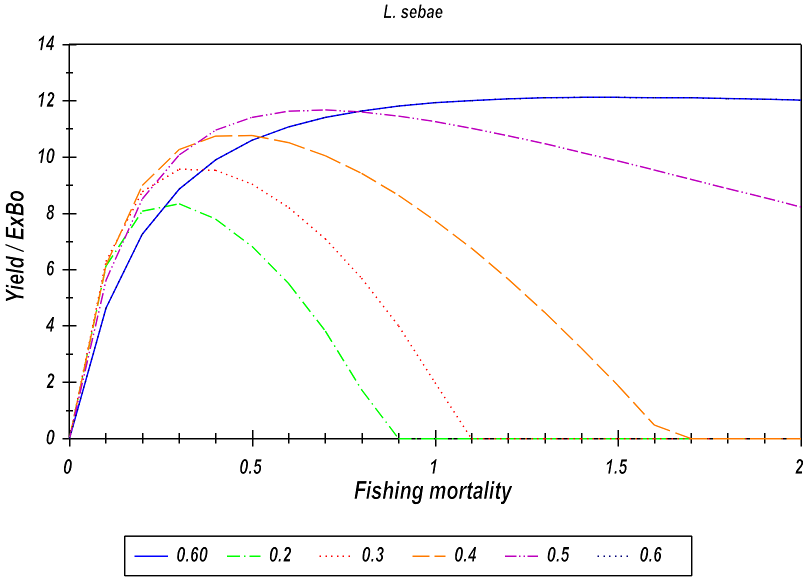

Figure 6.23 : Equilibrium curves for *L. sebae* with different lengths at first capture (Lc), including the current estimate (solid line). Estimated F on the Mahe plateau ranged between F=0.3 and F=0.7 during 1991-1993. For other areas, 0.2<F<0.7.

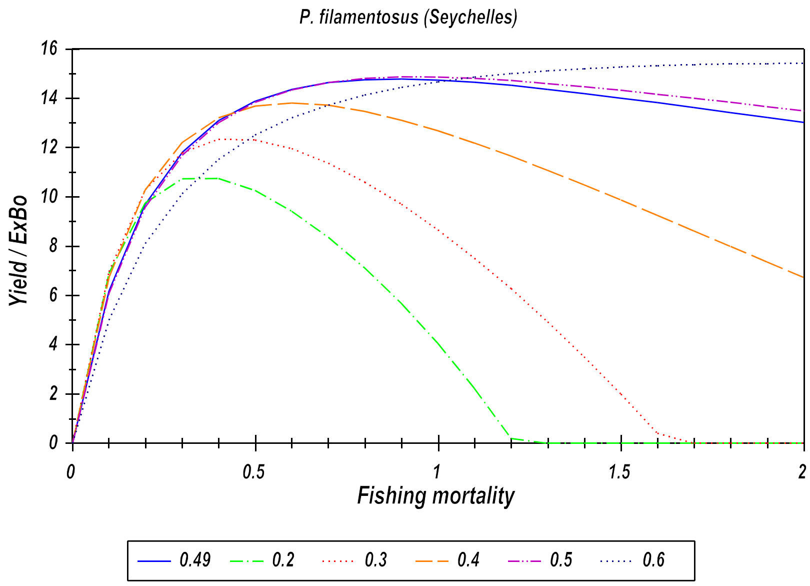

Figure 6.24 : Equilibrium curves for Seychelles banks stocks of *P. filamentosus* with different lengths at first capture (Lc), including the current estimate (solid line). Estimated F on the Mahe plateau ranged between F=0.2 and F=0.5 during 1990-1993. For other areas, 0.1<F<1.4.

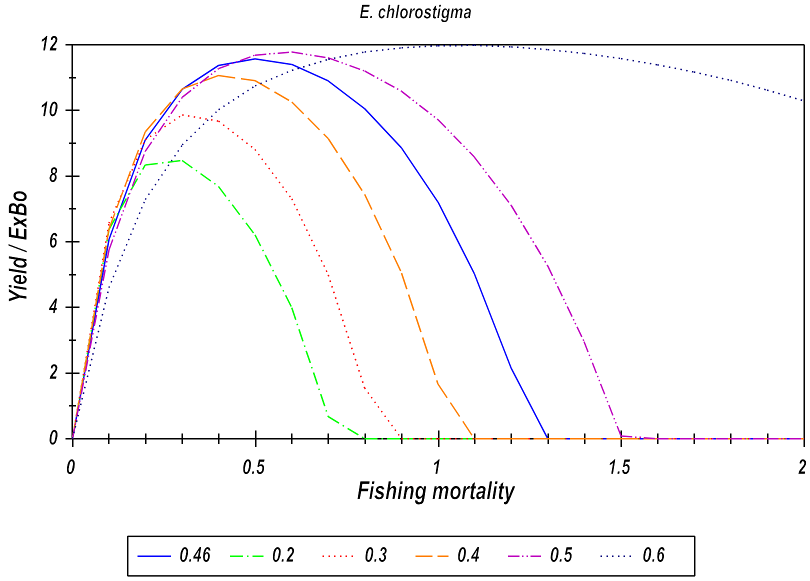

Figure 6.25 : Equilibrium curves for *E. chlorostigma* with different lengths at first capture (Lc), including the current estimate (solid line). Estimated F ranged between F=0.03 and F=0.7 during 1991-1994.

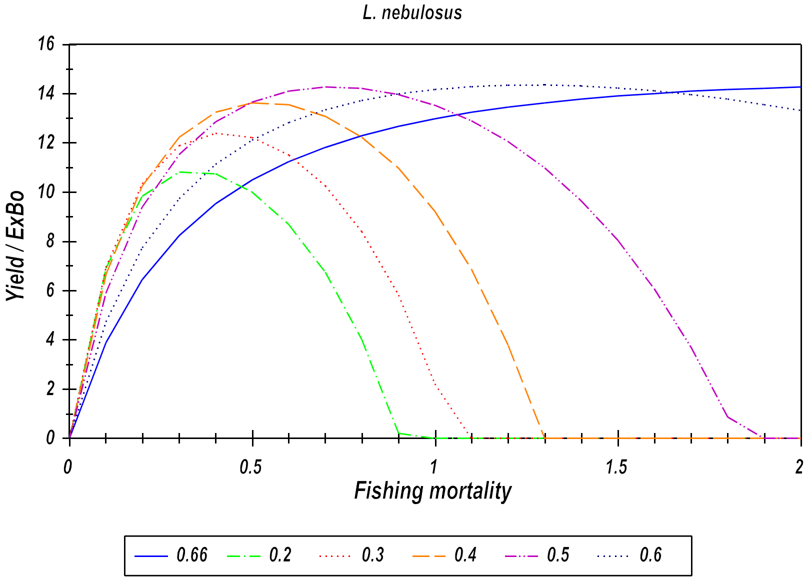

Figure 6.26 : Equilibrium curves for *L. nebulosus* with different lengths at first capture (Lc), including the current estimate (solid line). Estimates for current F are not available.

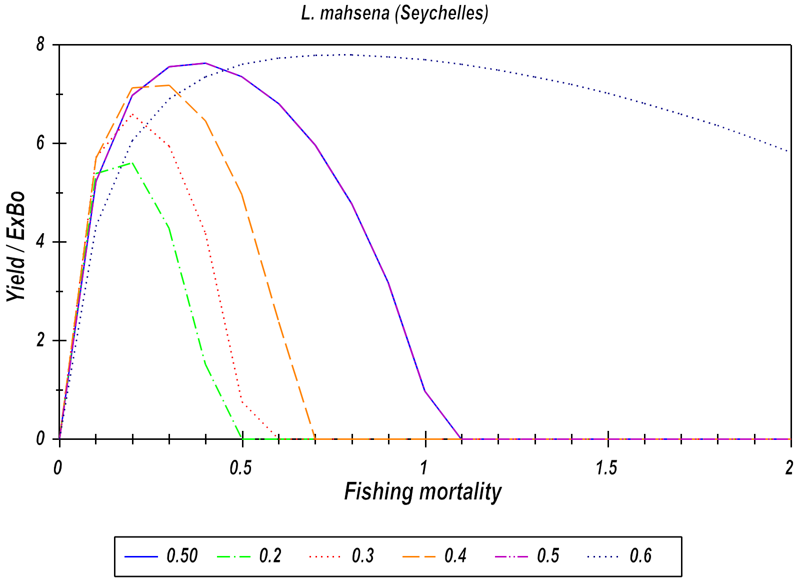

Figure 6.27 : Equilibrium curves for Seychelles banks stocks of *L. mahsena* with different lengths at first capture (Lc), including the current estimate (solid line). Estimates for current F are not available.

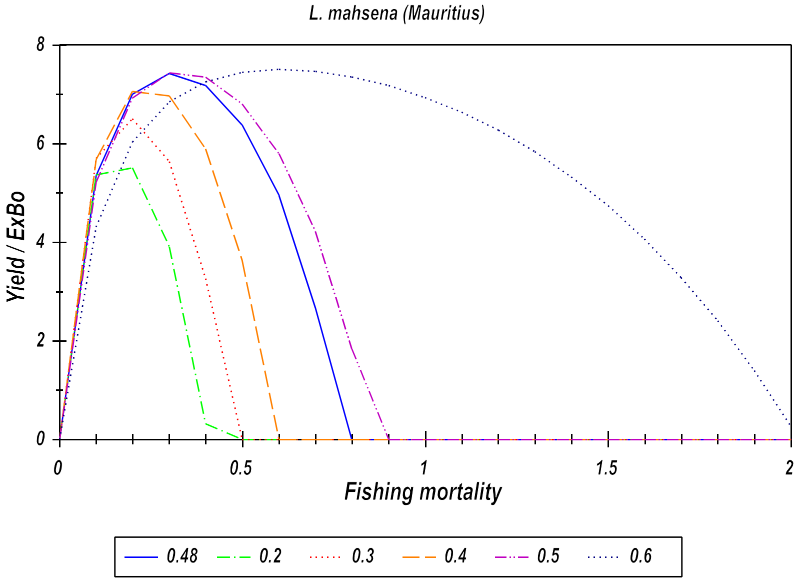

Figure 6.28 : Equilibrium curves for Mauritian banks stocks of *L. mahsena* with different lengths at first capture (Lc), including the current estimate (solid line). Estimates for current F are not available.

6.5 Minimum data requirements for management

It is not the intention in this section of the report to provide a detailed description of data requirements and the means of collecting them (which are available from fisheries texts, e.g. Gulland, 1977; 1983), but to highlight particularly important information. That relevant to fisheries management may broadly be classified as catch and effort data, biological data, social, economic and post harvest information, and climatic, environmental and hydrological data. Munro and Fakahau (1987) detail the role of each type of information and propose the frequency with which it should be collected. This project relied on historical information from established fisheries where the quality of data was believed to be good, described in Section 2.2. This 'commercial' data was used to both explore the effects of fishing (with certain limitations, see Section 5.2), and effects of different management strategies (see Section 6.3).

The results of this study indicated that single species and aggregate single species models were adequate to derive management advice for study demersal fisheries without the need for more complex ecosystem models accounting for all multispecies interactions. However, standardisation to account for technical interactions was essential. This indicated that :

- the present collection of catch and effort data for a few important species, and guilds of other species was adequate;
- to implement biological guidelines for management certain information is required and this was described (Section 6.3.5). It includes length at capture, growth and mortality parameters, catchability (q) of certain species, and virgin biomass (if possible). The estimation of catchability may prove to be one of the more problematic data requirements. It may be derived from depletion studies, or from an estimate of fishing mortality (F) and standardised effort (E), $F=qE$, but effort will need to be apportioned by species to account for targeting.
- Length frequency and biological data need only be collected for key species as is currently the case. The collection of length frequency information is essential, but biological data (eg length at maturity) was not so important.

Data collection and analysis implies costs in terms of manpower and financial resources. The cost of catch and effort data collection will vary depending upon the means of collection (logbooks to creel surveys or observer programmes), collection of length frequency data almost certainly requires manpower allocated to this task, whilst biological data collection often involves the purchase of fish specimens. The focus of the remaining discussion centres on these data requirements for management, but additionally the means and costs of other data collection need to be considered. Much information will already be available (eg. prices, exports), whilst some (eg. socio-economic status of the fisheries) may require incorporation into existing data collection formats or the execution of new surveys. Social, economic, post harvest, climatic, environmental and hydrological data tend only to be intermittently collected, and frequently not related to the 'fisheries' data.

It is important to prioritise data requirements in order that available resources are best utilised. With respect to fisheries and biological assessment for management, catch and effort data collection is a priority. Presently available information is adequate with respect to species information (although consideration should be given to the species to be grouped into guilds), but information in relation to technological changes in the fishery were limited. Principal deficiencies in the data, individually, or in common to the case study locations were :

- a lack of spatial and temporal heterogeneity in both catch, effort and biological data which limited their usefulness for assessing the effects of fishing and for deriving parameters for management;
- a lack of detail on technological changes within the fishery (eg addition of fish-finding equipment, gear changes);
- a lack of information on changes to fishing practices, e.g. longer time spent fishing (ie. need for more accurate effort data), or shift from day to night time fishing;
- a lack of information on gear sizes - particularly hook size- this is most relevant in relation to length frequency data, but can easily be collected with catch and effort information and related to the biological data collected from a particular vessel;
- insufficiently fine spatial detail, ie. inadequate reporting of fishing location;
- small sample sizes for lightly fished locations with the result that it was not possible to adequately standardise catch rates to account for the numerous variables (climatic and seasonal events, technological changes, depth, boat and gear differences, environmental differences - and lack of adequate information on all these variables);

For length/age based assessments, length frequency data collection is essential and where possible independent means of assessing growth appear to be advisable. Russ et al (1995) indicated that age structure was more useful than size structure in detecting effects of fishing. Biological data not available from length frequency information (eg. reproductive parameters, gut contents) whilst important, appear to be the least influential (see below). Deficiencies in data collection identified were :

- length frequency data, whilst commonly available, did not in the case of Seychelles relate to the most heavily fished location;
- growth parameter estimates were only available from length frequency data leading to uncertainty in estimates of a number of biological parameters (see Section 4.11);
- a lack of information on gear sizes - particularly hook size - in order to determine gear selectivity;
- a lack of spatial and temporal heterogeneity in length frequency (and biological) data limited their usefulness for deriving parameters for management;
- biological data was only available for certain species from Seychelles and not from other case study locations. Considerable geographical variation occurs in the estimates of biological parameters available in the literature suggesting that location specific data are required. Certain parameters required for management could not be derived;

As a generalisation, accurate collection of as much data as possible, addressing also the deficiencies above, is required to be of use for management. In particular, greater detail on the technological changes in the fishery are required to permit standardisation for them. However, as indicated, costs and priorities must be considered :

- Information lacking on the catch and effort database may be collected simply by adding the requirements to existing data collection procedures at no additional financial cost (although an additional analytical burden occurs).

- The means of defining analytical guilds and for reducing species information content on catch and effort data forms requires consideration, and could form the basis of further study. Guilds should be determined according to appropriate criteria such as depth and habitat association. Cluster analysis may be useful in defining these guilds. Species within study families are similar and within a depth or habitat type it may be appropriate to aggregate by family if further sub division of guilds is required. It may also be appropriate to aggregate within guilds species from different families which share similar biological characteristics, length at first capture, mortality, or trophic relations, for example. Suggestions for appropriate classification of guilds are given in Section 6.3.5.

- Improved spatial and temporal catch/effort, length and biological information may also be obtained at no additional cost by implementing a targeted sampling strategy. However, the lack of spatial and temporal heterogeneity in the data reflect fleet movements and little improvement in the data collected may, in fact, be possible. Length frequency and biological data which are the most costly, should therefore be targeted to relate to the most fished locations. However, where resources are available, length frequency (age) data in particular should also be collected from lightly fished locations in order that as the fishery progresses any change in the size structure of exploited resources may be observed.

- Increasing sample size implies additional costs and needs to be carefully considered. For the lightly fished locations this may not be practical anyway.

- Collection of biological data is the lowest priority due to cost and benefits but where resources are available length at maturity and spawning seasons should be obtained from successive one or two year biological data collection programmes aimed at the various key species. As occurs in Seychelles, costs may be minimised by reselling fish after analysis.

- Improved assessment of age and growth through techniques such as reading hard parts (otoliths) would appear to be important. However the costs of doing so are high, and another project (R.6465) will assess the costs and benefits to management of this. Growth estimates are used in determining other parameters such as mortality. If fishing mortality and effort are to be used to estimate catchability in order to apply the guidelines of this report, then accurate growth assessment is required.

- Better use of available information would be achieved by integrating all data, presently held in unrelated files, in a Fisheries Information and Management System (FIMS -database).

- More use should be made of techniques such as informal interviews of fishermen to gain additional information useful for management, such as the location and timing of spawning aggregations, at considerably cheaper cost than research surveys.

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