

Assessment and Management of Multispecies,
Multigear Fisheries: A case study from
San Miguel Bay, the Philippines.

by

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Abstract

This work uses ecological theory to explore the interactions between fishing and the ecosystem and examines the implications for fisheries assessment and management. The multispecies, multigear fishery of San Miguel Bay, the Philippines is used as a case-study. Three approaches were taken. The first was to descriptively and analytically assess the fishery. Catch rates in 1992-1994 and 1979-1982 were similar, but all other indications are that the fishery suffers from growth, recruitment and ecosystem overfishing. Large scale effort has decreased, but small scale effort has intensified and diversified. The second approach was to model the ecosystem using ECOPATH an equilibrium mass-balance model. The model described a relatively mature and resilient ecosystem, dependent on detrital and benthic flows. Different fishing gears have differential impacts on the ecosystem and these are modified by the interactive effects of predation and competition. In the third approach, a dynamic multispecies model, ECOSIM was used. The impacts of fishing by a multisector fishery on a multispecies resource were dynamically explored under top-down and bottom-up trophic hypotheses. The results demonstrated that the interplay of fishing mortality, species interactions and flow dynamics have profound implications for fisheries assessment and management. The uncertainties concerning the resource dynamics were explored using an adaptive management approach. Four ECOSIM models of the San Miguel Bay were used, top-down, bottom-up, immigration plus top-down and immigration plus bottom-up. Analysis of the EVPI showed that there was no value in learning more about the uncertainty or distinguishing between the different resource models. It was concluded that although an active experimental adaptive management was not worthwhile, adaptive management, using feedback information from the response of the resource to management actions as recommended. This thesis demonstrates the critical importance of an ecosystem-based approach to understanding fisheries dynamics.

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In memory of my mother,

Colleta Carolina Fransisca Leenhouts Bundy

Whose spirit lives on.

Chapter 1

Multispecies, Multigear Fisheries: Introduction and Overview

“...despite their importance, tropical fisheries are most often badly managed (if at all) - the resources are generally over-exploited and the fisheries overcapitalised..... Compounding the problem is the fact that, inspite of recent advances, the biological basis of tropical fisheries is only beginning to be understood; indeed the relative neglect of tropical fisheries research in the international literature appears very clearly when the relevant journals are surveyed. This problem is further complicated by the enormous diversity of life histories and adaptations among the organisms exploited, as well as the diversity of fisheries, gear types and social conditions, which represent serious (although not overwhelming) constraints on resources assessments and fisheries management.” Pauly (1994:15-16)

Introduction

About 60% of the world's fish catch is taken from fisheries in developing countries, yet the community dynamics and fishery dynamics of these predominantly tropical resources are little understood. These fisheries are typically multispecies, with over a hundred or more species landed for immediate consumption, trade, export, fishmeal, animal food or fish sauce. This is especially the case in Southeast Asia (Pauly 1994). These multispecies fisheries are usually exploited by a diverse small-scale sector, and, in many cases, a competing large-scale sector. Increased population growth, development of fishing technologies and demand for fish have placed enormous pressure on fish stocks. Patterns of exploitation have expanded and large-scale, capital intensive fisheries have long since moved into areas, and targeted species, traditionally exploited by small-scale fisheries (Pauly 1997, 1979a, Smith 1983, Marr 1982, Gulland 1982). The inevitable differences in interests and exploitative power between these

different sectors have led to over-exploitation and competition over resource use. At the same time, there has been little research into the ecosystem impacts of fishing on these multispecies, multigear fisheries.

The aim of this study is to use and explore a multispecies, ecological approach to the assessment and management of multispecies, multigear fisheries. In all fisheries, both inter-specific interactions, and interactions between the fishery and the ecosystem occur. Without doubt, fishing causes change in the ecosystem, but it does not do this in isolation. The impact of fishing on one or more species, has impacts on other species in the ecosystem. Fishing has been described as having a top-down effect on the food web (Pauly 1979a, Larkin 1996), although fishing occurs at all trophic levels, and thus at all levels of the food web, Christensen (1996). When examining the effects of fishing in a multispecies context, both the direct effect of fishing and the effect of inter-species interactions need to be taken into consideration.

Larkin describes ecosystem management, when applied to marine ecosystems as “scientific shorthand for the contemporary appreciation that fisheries management must take greater note of the multispecies interactions in a community of fish species and their dependence on underlying ecosystem dynamics” (1996:146-147). The problems of managing multispecies fisheries on an ecological basis are still being addressed in the developed world. Murawski (1991) amply demonstrated this in his description of issues and problems in the multispecies, multigear demersal fishery of the Gulf of Maine. Yet, there is arguably a greater demand and challenge to resolve these problems in tropical multispecies fisheries, where ecological and

fisheries data, are poor, but the fisheries catch many species and are fished simultaneously by very diverse fishing gear.

There has been very little research aimed specifically at the ecological assessment and management of multispecies, multigear fisheries, particularly for the fisheries of the developing world (Christensen (1996), Larkin (1996), Pauly (1994), Murawski (1991), Garcia (1989), Gulland and Garcia (1984)). Troadec (1983) suggests that historically, small-scale fisheries are regarded as unimportant and that emphasis in fisheries research is placed on large-scale, money making operations. West African governments for instance have a *laissez-faire* attitude to small-scale fisheries, regarding them as an activity of last resort, the situation also found in Southeast Asia (Smith 1983). However, the example of the Ivory Coast shrimp fishery, where small-scale fishers may have caused the economic collapse of the large-scale sector vividly demonstrates that small-scale fisheries should not be ignored (Griffin and Grant 1981). However, data from small-scale fisheries in developing countries are often sparse and uncertain. Data collection and analysis in many of these fisheries are challenged by dispersed landing sites, lack of trained personnel, lack of funds, lack of access, lack of facilities (Marr 1982) and, as suggested by Troadec (1983), lack of interest and management.

Much of the research that exists for the assessment and management of multispecies, multigear fisheries in developing countries concerns tropical penaeid shrimp fisheries, a resource with high value. Several studies have been conducted to examine the optimal fleet configuration for these fisheries (e.g., SWIOP/CNRO 1989, Willman and Garcia 1985, Jarrold and Everett 1991). Shrimp resources were traditionally exploited by small-scale fishers in countries such

as Mexico, West Africa, India and Southeast Asian countries, but since the 1950s, industrial shrimp fleets expanded into these areas (Willman and Garcia 1985). Penaeid shrimps are vulnerable to both gears at different stages of their life history. The small-scale fishers target the juveniles, the large-scale sector targets the adults.

Studies which examine the effect of fishing on the penaeids only are not truly multispecies studies, which are studies of systems composed of predator species, prey species and competing species. Other studies which have examined the effects of different sectors of a fishery on a single species or family conclude that a singular solution is best, that is, that the optimal solution is one where the fishery consists of only one gear. For example, Medley *et al.* (1991) used optimisation theory to demonstrate that optimal exploitation of the multifleet yellowfin tuna fishery occurred when only one gear existed. Clark and Kirkwood's (1979) bio-economic model of the Gulf of Carpentaria prawn fishery (Australia) also produced the result that the optimal fleet composition consists of only one of type of fishing gear. However, this was not considered to be an acceptable alternative since neither gear sector was likely to accept displacement by the other. Charles and Reed (1985) in their bio-economic analysis of sequential fisheries found that co-existence would only occur optimally (for the maximisation of total discounted rent) under a narrow range of conditions, related to cost and price ratios of the two fleets and the fraction of the offshore stock which spawns each season.

However, in multispecies, multigear fisheries, fishing gears do not all target the same species and singular solutions are highly unlikely to be optimal for small-scale, multispecies fisheries. Gulland and Garcia (1984) list a series of attributes which define multispecies, multigear

fisheries. These include (1) that the resource be multispecific, (2) that catch composition should depend on fishing strategy, (3) that the species composition of the resource changes with time and (4) that the various fisheries interact. For example, probably everywhere a large-scale shrimp fishery exists, the shrimp trawlers will interact and compete with finfish fishers. The shrimp trawlers, which use a small mesh size, catch the juveniles of the species which are targeted by the small-scale sector. This is particularly the case in Southeast Asia, where shrimp are found in the same shallow coastal areas as juvenile fish (Pauly 1994). In Australia, Haysom (1985) reported that conflict between shrimpers and sports fishers and crab fishers was caused by increased pressure by the shrimpers on whiting and sandcrabs. In West Africa, many small-scale fleets exploit the juveniles of other stocks, for example, *Sardinella*, which are exploited by large-scale sectors when mature (Troadek 1983). In Senegal in 1969, all trips made by large-scale shrimp trawlers were directed at shrimps. In 1978, only 25% of trips were targeted specifically at shrimp - the rest of the trips were primarily directed at high value species such as sole, kingclips and croakers (L'Homme and Garcia 1984).

There are many examples of species composition changes as a result of fishing. In West Africa the 'trash' fish *Balistes* has increased due to the exploitation, and consequent decrease in biomass, of predatory demersal stocks (Gulland and Garcia 1984, Pauly 1979a). In Northwest Africa, the sparids were replaced by cephalopods (Gulland and Garcia 1984). The Gulf of Thailand is the classic example of the effects of ecosystem overfishing (Pauly 1994). Huge declines occurred in the biomass of slow growing fish such as rays, small demersals such as the pony fish (*Leiognathidae*), medium sized and large predators as a result of the large-

scale trawl fishery. This biomass was replaced by generalists such as squid, prawns and the hairtails (Trichiuridae) (Pauly 1979a).

One multispecies approach to fisheries assessment is the study of technical in mixed-species fisheries (as opposed to multispecies fisheries). A mixed-species yield per recruit approach is used, for example, Brander (1983), Murawski (1984), Murawski *et al.* (1991), Pikitch (1989). These studies are premised on the fact that fishing gears vary in their selectivity patterns and exploit different age and size groups. However, although the mixed-species yield per recruit approach models the impact of fishing on a mix of species, by 'n' gears, it does not model biological interactions, that is multispecies interactions. Brander (1983) modelled the technical interactions between the *Nephrops norvegicus* and cod fisheries of the Irish sea, and the biological interactions between cod and *N. norvegicus*. He demonstrated that the yield of *N. norvegicus* was affected by the biomass of cod: when fishing pressure on cod was low, predation mortality on *N. norvegicus* was high, thus reducing their yield to the fishery. The maximum single species yields of cod and *N. norvegicus* were sub-optimal when the biological interactions between the two species were taken into account. As Brander (1983) suggested, the results "call into question the validity of management objectives based on single species yield-per-recruit criteria in a mixed fishery".

Yet, there is relatively little known about species interactions and the effects of fishing.

Sophisticated assessment methods used in temperate fisheries, such as multispecies Virtual Population Analysis (Magnusson 1995, Sparre 1991, see Larkin 1996, Hilborn and Walters 1992 for a review of multispecies approaches), are inappropriate for many fisheries due to their

large data requirements. There are several impressive volumes on multispecies methods and theory (Daan and Sissenwine 1991, Mercer 1982, and Pauly and Murphy 1982). Yet, there is no attempt to understand, in a holistic context, both the interactive dynamics between fish and the impact of fishing by diverse gears on a multispecies resource.

Research Objectives

The overall objective of this thesis is to use ecological theory to inform fisheries assessment and management. Its departure point is the knowledge that fishing takes place on an ecosystem, that fish do not live in isolation from other fish (and therefore they interact), that there are interactions between the fishery and the ecosystem, and that different fishing gears have different impacts. These interactions within the ecosystem and between the fishery and the ecosystem should be considered in fisheries management and assessment. A mass-balance, tropho-dynamic approach is used here to investigate their impact on fisheries assessment and management.

Specifically, the following aims were addressed in this thesis:

- to study species interactions in a multispecies fishery;
- to examine the biological and ecological impacts of a multigear fishery on a multispecies resource;
- to determine the effects of ecosystem considerations on fisheries assessment and management;

- To develop a systematic, integrated approach to the assessment of multispecies, multigear fisheries using relatively simple methods with wide applicability;
- to develop tractable sustainable management strategies for multispecies, multigear fisheries; and
- to focus on developing countries, where fisheries are frequently multispecies, multigear and data sparse.

The last three aims are also intended to address another concern. Many of the fisheries methods used to assess the multispecies fisheries of developing countries were developed for fisheries in temperate, developed countries, usually in the 'West', from a single species perspective. In many cases they are not particularly suitable for tropical multispecies fisheries (Pauly 1994). For many fisheries in the West, skilled fisheries scientists develop very specific, often sophisticated models to assess fisheries. Again, these approaches are not very useful for the fisheries in developing countries, for reasons discussed above, particularly the lack of trained personnel and lack of infrastructure. The main fisheries statistic collected in fisheries in the developing world, after catch and effort, is length of fish. Most of the methods developed in the West, use age to structure their models, not length. There is thus quite a degree of incompatibility between methods and approaches used to assess fisheries in the developed and developing world, or for temperate and tropical fisheries. For this reason, methods were used here that were designed for the type of data available in the fisheries of developing countries. One aim was to develop an overall approach which made maximum use of data that were available. Another aim was to use methods which were likely to increase knowledge about multispecies, multigear fisheries.

Methods

The fishery of San Miguel Bay, Philippines was used as a case study for this work. It is a multispecies, multigear fishery under stress due to excess fishing pressure from both large and small-scale sectors. It is typical of many fisheries in the developing world where small-scale fishers are dependent on the resource for their livelihood. Three types of trawlers operate regularly in San Miguel Bay, alongside a diverse range of small-scale gears, including handlines, gillnets, and fish corrals. There is a history of resource competition in the Bay dating from the 1940s, when there was already a trawl fishery for shrimp (Warfel and Manacop 1950). No regular annual fisheries statistics have been collected in San Miguel Bay. A study of the fishery conducted in the early 1980s concluded that the fishery was over-exploited and a series of management recommendations were made (Smith *et al.* 1983).

The data used in this work came from a multi-disciplinary study of San Miguel Bay conducted by the International Centre for Living and Aquatic Resource Management (ICLARM) from 1992-1994. ICLARM was contracted by The Fisheries Sector Programme of the Philippines (sponsored by the Department of Agriculture and the Asian Development Bank) and as part of a larger five year coastal resource management study in the Philippines, to conduct a 17 month "Resource and Ecological Assessment of San Miguel Bay".

This results of this intensive multi-disciplinary project have been published as a CD-ROM (ICLARM 1995). This complements an earlier study from the 1980s which was also conducted by ICLARM, in collaboration with the Institute of Fisheries Development and Research of the

University of the Philippines (Smith *et al.* 1983, Bailey 1982a, 1982b, Pauly and Mines 1982, Smith and Mines 1982).

The research presented in this thesis represents an independent analysis of the database from the 1992-1994 ICLARM research project¹, kindly made available to the author, and the use of published results where appropriate. The results of the 1979-1982 study of the Bay were used for comparative purposes. In addition, to fill in gaps in available data, information from literature sources was also used.

A large and diverse range of methods have been used in this thesis. Most were chosen for their applicability to the type of data that are available for San Miguel Bay, and to the limitations that these data place on the type of analyses that can be performed. For example, estimates of mortality and yield-per-recruit analyses are made using length-based methods because only length-based data are available for San Miguel Bay. Length-based approaches to fisheries assessment have been criticised (see Hilborn and Walters, 1992 for example). However, for many tropical fisheries, this is the only type of approach that is possible.

Whilst this research takes an ecological approach to fisheries management, it was not possible to include the entire ecosystem. For example, mangroves fringe the coast of San Miguel Bay, and are an important link between the terrestrial and marine ecosystems. Inorganic nutrients are imported from the land and organic nutrients are exported into the Bay. Mangroves also stabilise fresh water run-off from the land. In addition, forty two percent of the fish in Bay

¹ In some cases, because the initial analysis of the database occurred concurrently with ICLARM's analysis, the same parameters were calculated from the data, for example, total catch and biomass. Where this occurred, the results from my analysis were used.

spend part, or all, of their life cycle in mangrove areas (Vega *et al.* 1995a). Thus mangroves are important in the ecology of San Miguel Bay. However, there has been a trend of mangrove destruction in San Miguel Bay: the current area of mangroves is only 42 % of the area covered by mangroves in the 1950s (Vega *et al.* 1995a).

Several rivers discharge into the Bay, the largest of which is the Bicol river. The rivers introduce freshwater, silt, nutrients and pollution (from industrial, agricultural and domestic sources) into the Bay. Mendoza *et al.* (1995b) report that the Bay is not polluted although there is nutrient enrichment. Siltation of the Bay makes it shallower, and may also clog the gills of fish (Mendoza and Cinco 1995).

The Bay is, by definition, open to the ocean. Suspended materials maybe carried into the Bay by tidal currents, although Villanoy *et al.* (1995) report that there is no net transport into or out of the Bay. It is also likely that there is movement of fish in and out of the Bay. Pauly (1982b) speculated that fish spawn outside of the Bay and that larvae are carried into the Bay on tidal currents. Another feature which may influence the ecology of the Bay are coral reefs which are found at the mouth of the Bay. These are reported to be in fair to good condition (Garces *et al.* 1995c).

The biotic and abiotic inputs described above are clearly part of the overall ecosystem. However, whilst recognising that they will have a modifying effect on the Bay, it was not feasible to explicitly include them in the mass balance, tropho-dynamic approach used here.

Thesis Outline

The thesis begins with a description and assessment of the San Miguel Bay fishery (see Figure 1.1). This is a background chapter which sets the scene for the three subsequent chapters. In Chapter 3, the ecosystem is modelled using a static mass-balance model (ECOPATH) and in Chapter 4 it is modelled using a tropho-dynamic model, ECOSIM. The aim of both these chapters is to model the ecosystem, gain an understanding of species interactions, the impact of fishing by a multigear fishery on the ecosystem and the implications of the results for fisheries assessment. In Chapter 5, the management implications of the results in Chapters 2, 3 and 4 are examined using an adaptive management approach. Each chapter is described in more detail below.

The fishery of San Miguel Bay is described in Chapter 2. The background of the fishery is first assembled, then the current state of knowledge of the fishery described. The purpose of this chapter is to lay the foundations for the subsequent chapters. It is a basic fisheries assessment chapter, using methods which avoid the constraints that the data place on analysis. For example, the standard practice of analysing series of catch and effort data was not possible, because these data do not exist. Data from the trawl survey and catch survey are analysed using single species and length-based methodologies, including mortality estimation and yield-per-recruit analysis.

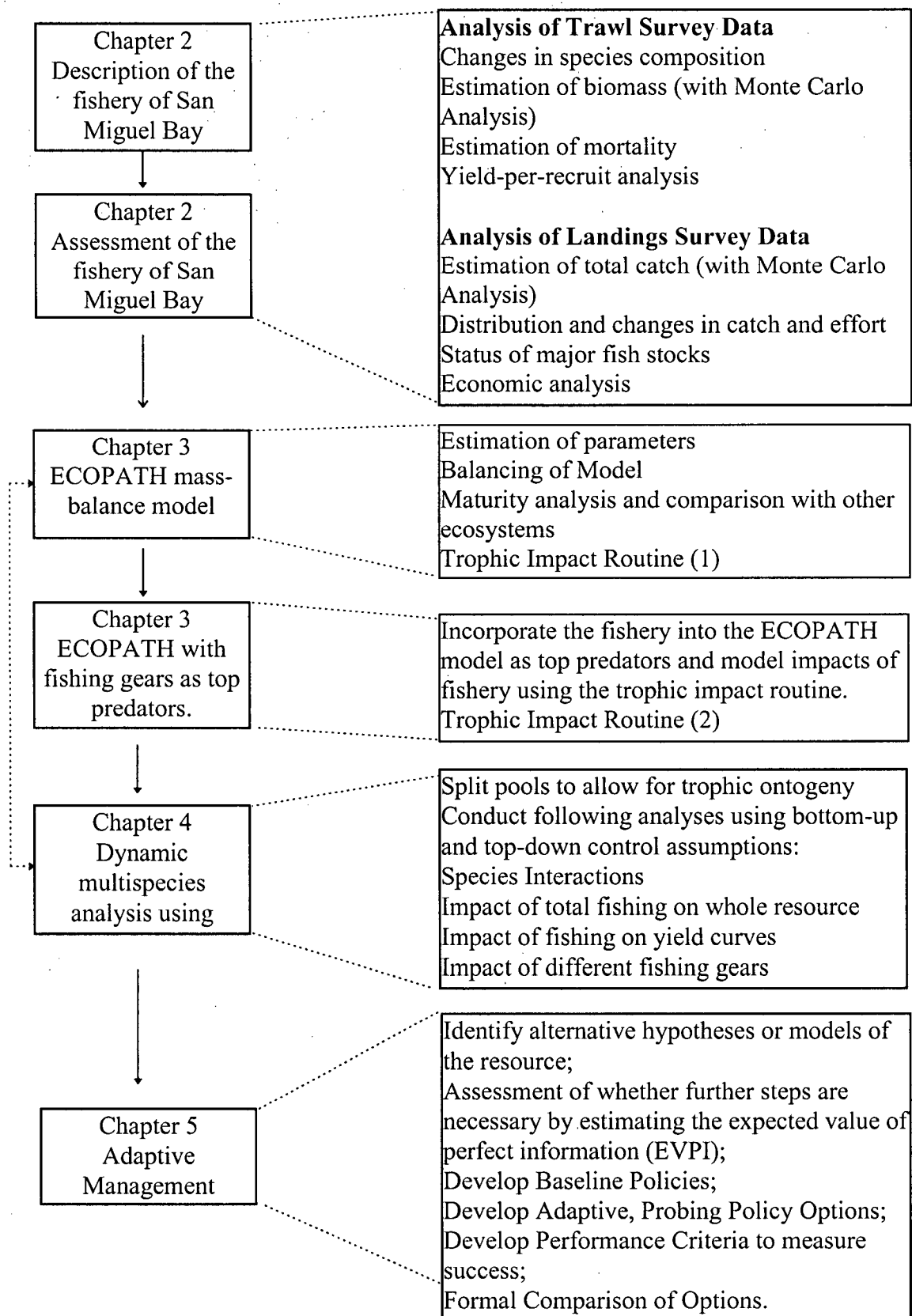


Figure 1.1 Flow chart showing thesis chapter layout and content.

A longitudinal analysis of the species composition data is conducted and compared to historical data from 1979-1982 and an earlier survey in the late 1940s (Warfel and Manacop 1950).

Chapter 3 takes the first step towards a multispecies species analysis of the fishery. The fishery is modelled using a mass-balance model, ECOPATH (Christensen and Pauly 1992a).

ECOPATH is a mass-balance description of trophic interactions. It is used to determine and describe the interactions between different components within the ecosystem. In this way, the major energy flows and pathways in the ecosystem, upon which the fishery is based, are identified. The state of development or maturity of the ecosystem is also examined using a series of indices outlined in Christensen and Pauly (1993c) and Christensen (1995, 1994). In addition, the key areas where information is poor are also highlighted. ECOPATH is a means to collate data about a system in a coherent form, enabling a better understanding of the entire system. Knowledge is increased. A second model was then built, incorporating the fishery, large-scale and small-scale, as predators within the model. In this way it was possible to directly examine the impacts of the fishing gears on the ecosystem.

In Chapter 4 the fishery of San Miguel Bay is modelled using ECOSIM, a dynamic multispecies model (Walters *et al.* 1997). This model requires only a few more parameters than ECOPATH, and is thus very appropriate for data poor fisheries. With ECOSIM it is possible to directly examine the impact of the fishery on multispecies resource, and, ask, “what if?” questions, that is, to make predictions. The model is used to examine the interactions between different components of the ecosystem and between the fishery and the ecosystem.

Furthermore, ECOSIM is used to simulate the two “views” of energy flow control in ecosystems: top-down control (Carpenter and Kitchell 1993) and bottom-up control (Hall *et al.* 1970, Hunter and Price 1992) are simulated and compared.

There are two major uncertainties concerning the resource dynamics in San Miguel Bay: the relative strength of inter-species interactions and their response to fishing, and the importance of fish immigration in the fishery. These uncertainties are explicitly addressed in Chapter 5 using adaptive management (Walters 1986). Adaptive management is essentially a feedback system whereby empirical information generated from the resource in response to management is used to shape future management strategies. It is proposed that experimental adaptive management leading to an increase in empirical data could result in more constructive management policies, particularly for data sparse fisheries in the developing world such as San Miguel Bay.

In Chapter 6, the preceding chapters are revisited in the light of the aims of this thesis. Finally, the question is asked, “Is the ecological approach developed here, whereby ECOPATH and ECOSIM are used to model multispecies fisheries and adaptive management used as a means to resolve uncertainty and gain knowledge, a useful approach for other fisheries?”

This thesis may be read as a thorough assessment of a multispecies, multigear fishery from the Philippines. It may also be read as part of an attempt to develop and use methodological approaches appropriate for multispecies, multigear fisheries in the developing world. Most fundamentally, it is an attempt to understand the biological and ecological interactions

between a multigear fishery and a multispecies resource and to examine the implications for fisheries assessment and management.

Chapter 2

The San Miguel Bay Fishery

“We know that the assumption of a constant parameter system is never strictly fulfilled in real life. However, we are often in a situation which forces us to make assumptions, which are known to be crude approximations to reality. It often happens that only by making such assumptions we are able to carry out an analysis of available data, and it is better to do a crude analysis than no analysis at all.”

Sparre *et al.* 1989, p.139

Introduction to San Miguel Bay

San Miguel Bay is a large shallow estuary in the Bicol region of the Philippines (Figure 2.1). It supports an important multi-species fishery, traditionally exploited by small-scale fishers using a wide range of gear types. The Bay falls within the jurisdiction of two provinces, Camarines Norte and Camarines Sur. The two provinces have seven municipalities adjacent to the Bay, Basud and Mercedes in Camarines Norte and Cabusao, Calabanga, Sipocot, Siruma and Tinambac in Camarines Sur. The most densely populated municipality is Calabanga and the least, Siruma. Around the coasts of these municipalities are 74 coastal or fishing barangays (villages). The population of the 74 barangays was reported as 92, 716 in 1990. This represents 2.4% of the Bicol population of 3.9 million. Bicol, whose economy is dependent on agriculture, is one of the poorest provinces in the Philippines and small-scale fishers live at or below the national poverty level (Dalusung 1992).

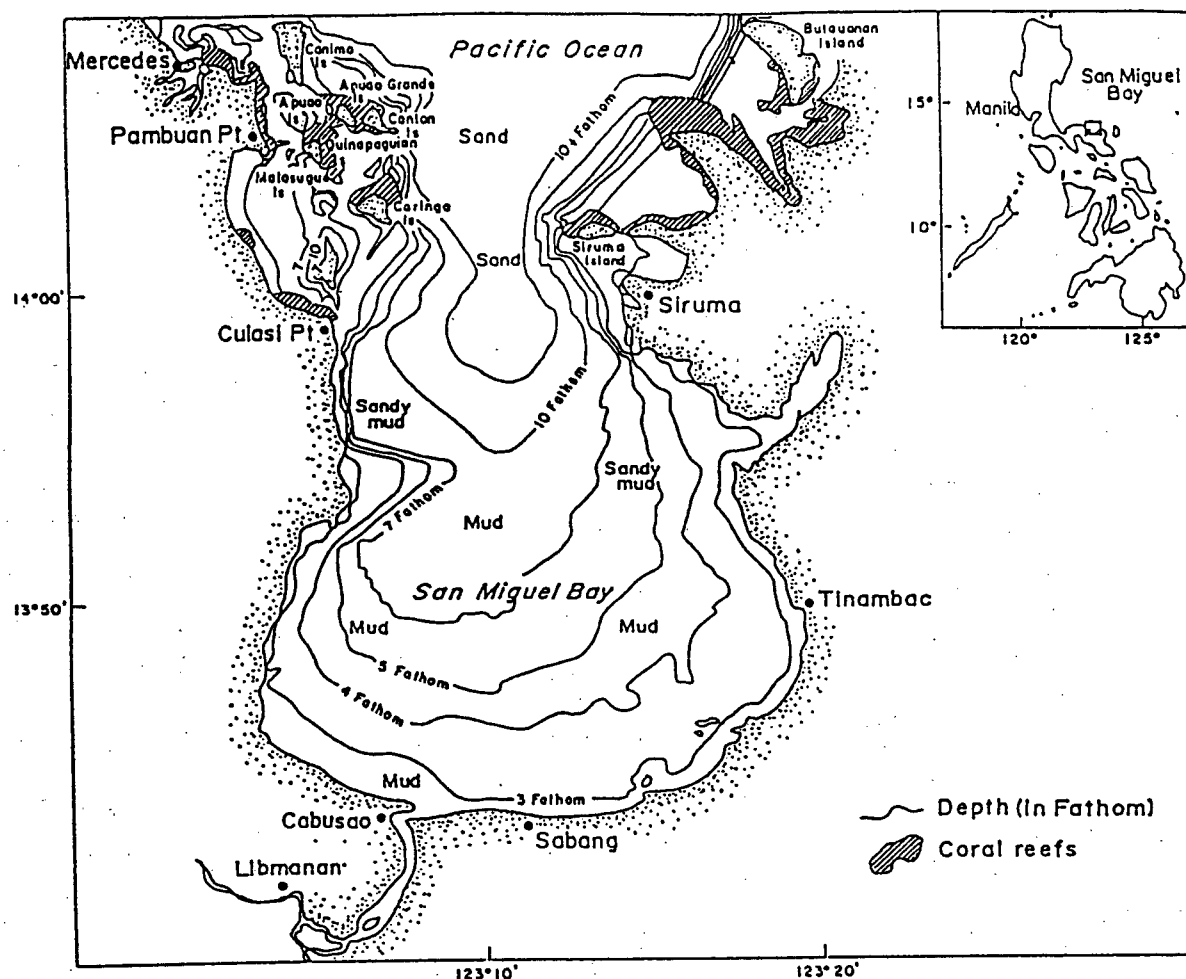


Figure 2.1 Map of San Miguel Bay, Philippines showing bathymetry, sediments and key locales. (ICLARM 1995)

Since the 1970s and initially as a result of government credit programmes, trawlers have exploited the Bay, mainly for shrimp. The Bay is ideally suited for trawling: it is wide, 1115 km², shallow and 95% of the Bay has a soft-bottom (Garces *et al.* 1995a). However, this was by no means the first instance of trawling in San Miguel Bay. In the 1930s, three Japanese beam trawlers fished there (Umali 1937). In 1950, as a result of a nationwide trawl survey, Warfel and Manacop (1950) suggested that San Miguel Bay, as one of the most productive trawling areas in the Philippines, could sustainably support four or five trawlers. By the late 1970s there were 88 commercial trawlers fishing in the Bay (Simpson 1978), and this number has since increased.

Catch and effort data are not routinely collected in San Miguel Bay. There are thus no systematically collected fishery statistics, nor time series of data. Management has been minimal over the years. Though mesh size regulations and closed areas for trawlers exist, enforcement has proven problematic. Conflict has developed between the small-scale fishers and large-scale trawl fishers who compete for the same, limited resource. This was noted in other areas of the Philippines as early as the 1950s. Rasalan (1957:53) reported that, in the opinion of the fishers, “the extensive operations of the otter trawl do not only destroy the eggs, larvae, food and the spawning ground of fishes, but also fish corrals and other fishing gear which are set at the sea bottom to catch demersal species”. At this time, the annual catch landed by the commercial fishery was 1,000-2,000 tonnes (Rasalan 1957).

During the late 1970s and early 1980s, an estimated annual catch of around 19,000 tons was caught by over 18 different types of small scale gear and four types of trawlers (Pauly and

Mines 1982). The small-scale gears include an array of passive gillnets, such as drift, bottom, and shrimp gillnets, some using motorised boats (*bancas*); fixed gears such as fish corrals, filter nets, fish weirs and stationary lift nets; simple active gears such as push and pull nets, which do not require either a boat or engine, and handlines, longlines and spear guns. The trawlers include large, medium, "baby" and "mini" trawlers (Vakily 1982). Large trawlers are 19 to 25 meters long, varying in capacity between 27 and 117 tonnes with engines of 275 to 555 horsepower. The medium trawlers are smaller vessels: their length is around 18 meters, their tonnage 3 to 6 tons and engine horsepower of 200. Baby trawlers, as the name suggests, are significantly smaller. Their capacity is around 1.6 to 3 tonnes, their horsepower 68 to 160 and their length about 12 meters. Mini trawlers are powered by a 10-16 horsepower engine, they are about 5 meters long and have a capacity of less than one tonne. The small-scale sector employed 5, 100 out of a total of 5, 600 fishers in San Miguel Bay (Smith and Mines 1983).

In the Philippines, fishing vessels are divided into commercial vessels and municipal vessels (in this thesis, the terms municipal sector and small-scale sector are equated). Commercial vessels are those with a capacity of over 3 gross tons and, in San Miguel Bay, include the large and medium trawlers. Accordingly, municipal or small-scale vessels are those that weigh less than 3 gross tons. This law dates back to 1932, when it was arbitrarily instituted for taxation and licensing purposes (Pauly 1982a). The unfortunate consequence of this arbitrariness is that small commercial trawling vessels are classified as municipal because their capacity is less than 3 gross tons. Thus, in addition to the traditional gears in San Miguel Bay, the so-called small-scale sector also formally includes the "mini-trawlers", which mainly exploit sergestid shrimp, and the small or "baby" trawlers, which target penaeid shrimp and demersal fish. This

classification impacts the fishery regulations, which have different rules for commercial fishing and municipal trawling.

The Fisheries Act of 1975 (or PD 704) encompasses all fisheries regulations in the Philippines (Luna 1992). The Bureau of Fisheries and Aquatic Resources (BFAR) have responsibility for commercial fisheries whilst municipal or small-scale fisheries are under the jurisdiction of the municipality. All vessels must be licensed, although this is more to earn revenue for the government than to control fishing. In fact, according to Luna (1992), the whole Fisheries Act can be considered to be primarily focused on the further development of the fisheries sector.

The Fisheries Act placed a 7 km or 7 fathom ban on all commercial vessels. This meant that commercial trawlers, for example, could not fish within either 7 km of the shoreline or in waters less than 7 fathoms deep. However, there were alternative rules for municipal trawlers, that is, trawlers less than 3 gross tons, the baby and mini-trawlers. These vessels were allowed to fish in waters as shallow as 4 fathoms, if given permission by the municipality and approval by the Department of Agriculture secretary (Luna 1992). In 1991 the New Local Government code was instituted and this increased the jurisdictional responsibility of the municipalities. Principally, The Code extended the municipal boundaries to 15 km from the shoreline. This placed San Miguel Bay in its entirety within the jurisdiction of the municipal authorities, and thus the decision on whether to allow trawling in the Bay was completely in their hands. In the early 1980s a multidisciplinary research project involving ICLARM (International Centre for Living and Aquatic Resource Management) and the Institute of Fisheries Development and Research of the University of the Philippines conducted a study of the

fisheries of San Miguel Bay. This was the first real attempt to assess the status of San Miguel Bay and to consider management options and realities. Data were collected on the biology, economic and socio-economics of the fishery. The study concluded that the Bay was over-fished, that there were reduced profits in the fishery, that catch and income were unequally distributed between the trawling and small scale sector (heavily in favour of the trawlers) and that there were too many people fishing. The overall conclusion was that the Bay was in dire need of management. ICLARM developed a series of management objectives, including the restriction of trawling activities and recommended the creation of a San Miguel Bay Fisheries Authority, an authoritative body to manage the entire Bay (Smith *et al.* 1983).

Despite the findings of this comprehensive study, and a subsequent 5 year ban on commercial trawling, little changed in San Miguel Bay: overfishing and lack of management continued. In 1986, Smith and Salon (1987) conducted a follow up study. They reported that the number of trawlers had increased by 50%, that the number of small-scale gears and fishers had increased, that out-migration of fishers had increased, and that demands to close the Bay to all trawling activities had grown to a clamour. So it appeared that not only did the results of the ICLARM study have no impact, the situation had, in fact, worsened.

More recently, a second comprehensive study of San Miguel Bay was conducted by ICLARM from 1992 to 1994. ICLARM was contracted by The Fisheries Sector Programme (FSP) of the Philippines (sponsored by the Department of Agriculture and the Asian Development Bank) as

part of a larger 5 year coastal resource management study in the Philippines, to conduct a 17 month "Resource and Ecological Assessment of San Miguel Bay"².

An assessment of the San Miguel Bay fishery is made in this chapter using the data from this study. As noted above, no regular fishery statistics are collected in San Miguel Bay. This means that there is no time series of catch and effort data, no regular, systematic trawl surveys and no annual census of boats or fishers. Thus, most methodological approaches commonly used in fisheries assessment are not applicable to this fishery. For example, even relatively simple methods such as surplus production models cannot be used because of the lack of a time series of catch and effort data with which to fit the model.

For the above reasons, the assessment made here is both analytical and descriptive. It uses the data from the recent ICLARM project in San Miguel Bay to assess the current state of the fishery. The results of the earlier 1979-1982 ICLARM project are used for comparative purposes.

² A CD-ROM "The San Miguel Bay Story" (ICLARM 1995a) has been published containing the results of this research.

Methods

The Data

The data for the assessment of the fisheries of San Miguel Bay came from three surveys conducted by the Capture Fisheries Assessment component of the ICLARM study (Silvestre *et al.* 1995, Cinco *et al.* 1995).

Trawl survey Data

The trawl survey was conducted from July 1992 to June 1993 (Cinco *et al.* 1995). A one hour drag, during daylight, was made monthly at each of nine stations, distributed around San Miguel Bay (Figure 2.2). The vessel used for the survey was a 1.93 ton, 10 m trawler with outriggers and a 65 HP inboard diesel engine. The trawl net used, a 4-seam bottom trawl with a 12 m headline and 0.9 cm cod-end mesh size, was typical of the trawl gear used in San Miguel Bay.

Fishery survey Data

The fishing gear inventory was conducted from January 1993 to June 1993. It covered all 74 fishing barangays in San Miguel Bay and thus the seven municipalities. Information on ownership of gear and seasonality of gear use were obtained from the local barangay officials, who are up-to-date on who is fishing, with what, and where.

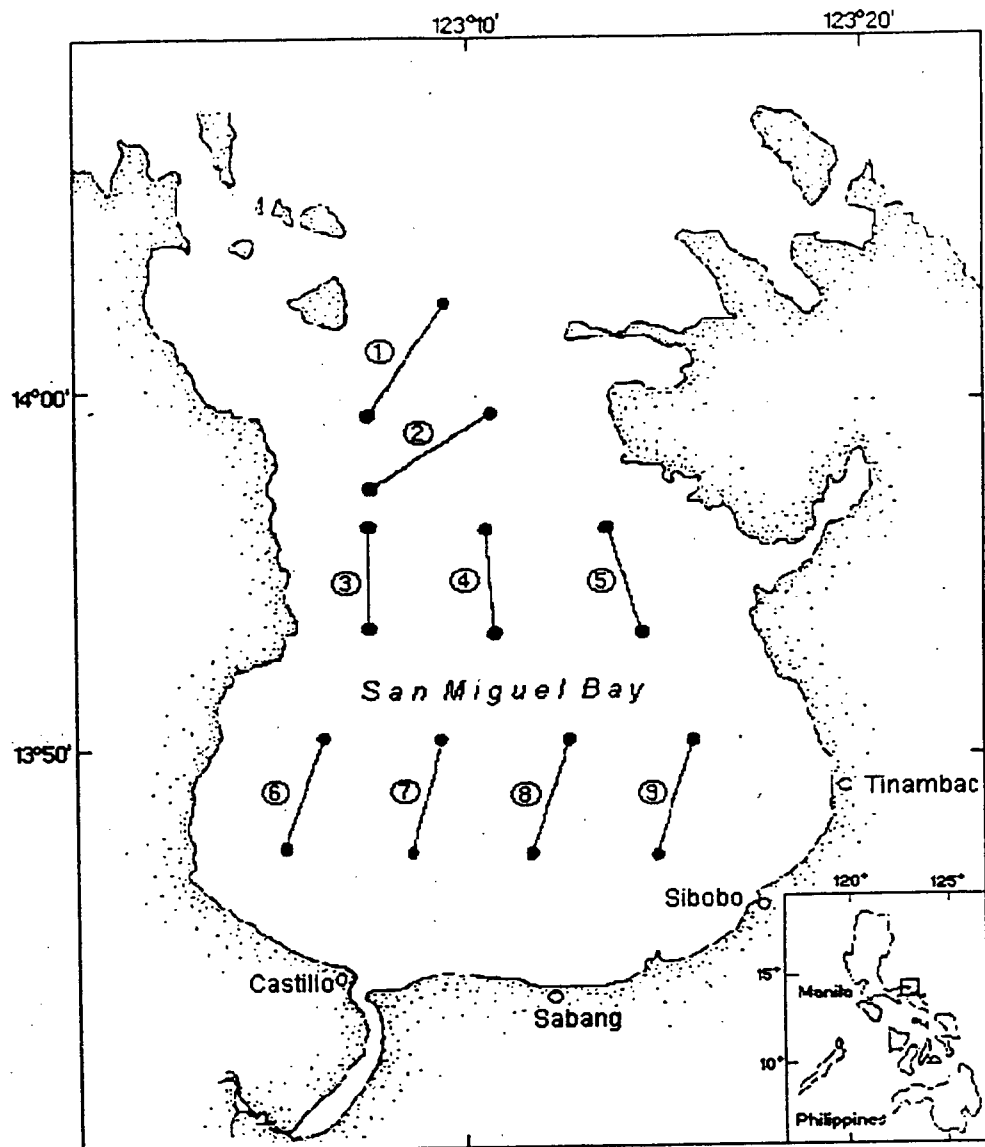


Figure 2.2 Map showing the trawl stations used during the 1992-1994 trawl survey of San Miguel Bay. (Cinco *et al.* 1995).

The landings survey took place from July 1992 to June 1993. It covered the large and the small-scale sector and the following data were collected for each gear type: catch landed per trip, species composition of the catch, length composition of the catch (by species or group).

The large-scale sector was surveyed at three landing sites in three municipalities Sabang (Calabanga), Castillo (Cabusao) and Padawan (Mercedes) and the small-scale sector at five landings sites in four municipalities, Castillo (Cabusao), Sabang and Sibobo (Calabanga), Filarca (Tinambac) and Padawan (Mercedes). The landings of the large scale sector were monitored once a week and the landings of the small-scale sector every other day. In both cases, monitoring was done by team members resident at the landing sites.

The comparative data from the 1979-1982 ICLARM study were taken from published reports³.

Analyses of Trawl Survey Data

The Species Composition of San Miguel Bay

The current species composition of San Miguel Bay was determined from the 1992-1994 trawl survey data. Catch per unit effort (CPUE) was used as a direct measure of abundance.

³ These are available on the San Miguel Bay CD-ROM (see footnote 1), or as a series of hard copy technical reports published by ICLARM (1995b).

The abundance of each species present was expressed as the relative contribution of the CPUE of that species to the total CPUE over the 10 month survey. Once the total species present were collated, they were grouped into families.

Estimating Density and Biomass from the Trawl survey Data

The density and total biomass of San Miguel Bay was estimated from the trawl survey data by the swept area method. Following the procedure in Sparre *et al.* (1989) and Cinco *et al.* (1995), the monthly density (t/km^2) was first calculated from the total monthly catch using equation (a) below. The mean annual density was then calculated as the mean of the estimated density per month (equation (b)). The mean biomass was calculated by multiplying the mean annual density by the effective area of the Bay (c).

$$\text{Density}_{\text{month } i} = \text{Catch Rate}_{\text{month } i} / (X_1 * X_2 * L * HL) \quad (a)$$

$$\text{Density} = \text{mean} (D_{\text{month } i}) \quad (b)$$

$$\text{Biomass} = \text{Density} * A \quad (c)$$

where X_1 = escapement factor

X_2 = effective width of the of the swept area

L = length of the path swept by the trawl

HL = headrope length

A = area of San Miguel Bay

The density and biomass estimates from this equation are not very precise due to uncertainties in the equation parameters (Sparre *et al.* 1989). In an attempt to include these uncertainties in the biomass estimate, Monte Carlo simulation (Crystal Ball Inc., Denver, Colorado) was used to estimate means and 95% confidence limits of the estimates. In this technique, the error distributions of the input parameters are specified. Then 2000 random samples are taken from the input parameter distributions and the model results calculated for each. Finally, the distribution of these result values provides the band of estimates and a sensitivity analysis. Error distributions were input for the parameters X_1 , X_2 and L .

The escapement factor, X_1 , refers to the proportion of fish in the path of the trawl net that are actually captured and retained in the net. Underwater film footage of trawl nets in operation clearly show that some fish can outswim the trawl net, at least for some time. The proportion of the fish caught varies with the speed of the trawl, the height of the fishing line from the seabed, the width of the trawl opening, the species of fish targeted and other species in the fishery, etc. Values for X_1 in the literature range from a recommended 0.5 for trawlers in southeast Asia (Saeger *et al.* 1980) to 1.0 Dickson (1974). Differentiating between these values is difficult (Sparre *et al.* 1989). Indeed, Hilborn and Walters note that 'the bottom line is always "What proportion of fish in the area swept were captured"' (Hilborn and Walters 1992:163). To incorporate this uncertainty into the biomass estimate, a triangular probability distribution, with a minimum of 0, and a maximum of 1.0 was used. Since there are no means

to discriminate between the two literature estimates above, the mean value of 0.75 was taken as the peak of the triangle and thus the most probable value.

Estimates of X_2 , the proportion of the headline length that describes the width of the trawl path, range from 0.4 (Shindo 1973) to 0.6 SCSP (1978). Pauly (1980a) recommended 0.5 as a compromise value. The uncertainty in this parameter originates in two areas, the variation of wingspread during the trawl and the differences in the way that the gear is rigged by fishers. This likely variation was represented by a triangular probability distribution, with a minimum value of 0.4, a maximum value of 0.6 and the "compromise" value of 0.5 as the midpoint.

The effective distance, L , travelled in one hour by the survey trawlers was 5.49 km (Cinco *et al.* 1995). To allow for variability in the distance travelled caused by environmental factors such as weather, tides, bottom topography and depth, a 10% error factor was introduced to the estimate. This was represented by a uniform probability distribution with a range of 4.94 to 6.04 and 5.49 as the midpoint.

The area of San Miguel Bay is 1,115 km² (Garces *et al.* 1995a) with a trawlable area of 95%. However, since the 1979-1982 study, and other earlier studies (see Pauly 1982a) used a value of 840 km² as the area of San Miguel Bay, this latter value was also used for comparative purposes⁴. The biomass and density results were then compared with the results from 1979-

⁴ The difference in these two estimates is due to different boundary definitions of San Miguel Bay. The estimate of 1115 km² is derived from a boundary drawn further north than the estimate of 840 km². The latter was drawn from Pambuan Point in Camarines Norte, eastwards to Siruma Island and then the mainland of Siruma in Camarines Sur (Mines *et al.* 1982, Figure 1). The former was drawn from Grove Point in Camarines Norte east to Butaanan Island and southeast to Quinabuscan Point in Camarines Sur (Garces *et al.* 1995a), see Figure 2.1.

1982 trawl data and with the results from 5 earlier trawl surveys dating back to 1947 (Pauly 1982a).

Longitudinal Comparison of Species Compositions from Trawl survey Data

The theory of “fishing down an ecosystem” is now well known. When a fishery first develops, the larger, more valuable fish are targeted. As time proceeds and stocks dwindle, the fishery switches its attention to the next most valuable species and fishes this until another switch is forced and so on. In this way, a fishery tends to be fished at different trophic levels at different stages in the history of the fishery. This fishing practise becomes problematic when fishing pressure increases to the point where all extant trophic levels are targeted simultaneously and there is no reprieve for declining fish groups. Usually this point is not reached unless large-scale trawling activities, often for prawns, compete with the traditional small-scale sector for resources. The effect of this mass harvesting of trash fish and juvenile fish, in addition to prawns, can destabilise multispecies resources and cause massive changes in species composition (Pauly 1979a, Beddington and May 1982).

Pauly (1979a) examined the species composition changes that had occurred in the Gulf of Thailand and the Thai waters of the Malacca Strait Fishery as a result of the large scale trawling activity that has occurred since the 1960s. He proposed a general pattern of change which included the loss of slow growing fish such as rays, huge declines in small demersals such as the Leiognathidae, declines in medium sized and large predators and the rise of generalists such as squid, prawns and the Trichiuridae. When a fishery reaches this stage it

might suffer from “ecosystem overfishing” which is defined as “what takes place in an ecosystem when the decline (through fishing) of the originally abundant stocks is not fully compensated for by an increase of the biomass of other exploitable animals” (Pauly 1979b).

A longitudinal comparison was made with the species composition data from the 1992-1994 trawl survey, the trawl data from 1979-1982 (Vakily 1982) and the trawl survey data from 1947 (Warfel and Manacop 1950). The aim was to determine what species composition changes have occurred in San Miguel Bay, to determine whether they agree with Pauly’s general pattern, and to determine whether San Miguel Bay is ecosystem overfished.

Seasonal Analysis of the Trawl Survey Data

San Miguel Bay is subject to seasonal environmental conditions. Most significant is the northeast monsoon which occurs from October to March (Villanoy *et al.* 1995). During this time, the average wind speed is 3 metres per second but it can be as high as 7 metres per second. The northeast monsoon also marks the beginning of the rainy season, and the rate of fresh water discharge from the 12 rivers that flow into the Bay is correlated with this. It is also the period when the plankton density is highest.

The other main environmental force in San Miguel Bay is the southwest monsoon which occurs between June and September (Villanoy *et al.* 1995). This has much less effect than the northeast monsoon because the Bay is protected by the Bicol Peninsula. May to August is the

hottest period of the year, with temperatures in the 32°C to 37°C range and during this time, the plankton counts are low.

In order to examine the impact of these seasonal conditions on the San Miguel Bay fishery the CPUE and species composition of the 1992-1994 trawl survey data were examined for seasonal differences. They were then compared with the trawl data from 1979-1982, where appropriate, to determine whether there was any consistency between the two data sets. The 1979-1982 trawl data were obtained using commercial trawlers. A systematic trawl survey was not conducted.

Estimation of Mortality

The instantaneous rate of total mortality, Z , was estimated using three length based methods, the Length Converted Catch Curve, Beverton and Holt's method (1956), the Powell-Wetherall Plot (Gayanilo *et al.* 1996). A fourth method which estimated Z from the sum of fishing mortality and natural mortality was also used.

Estimation of Growth Parameters

The three length based methods all require the growth parameters, L_{∞} , the asymptotic length in the von Bertalanffy growth function and K , the von Bertalanffy growth constant. These were estimated using the ELEFAN I program contained within the FAO-ICLARM Stock Assessment Tools software package, FiSAT (Gayanilo *et al.* 1996). Growth curves are fitted by ELEFAN I through a time series of length frequency data, using the von Bertalanffy Growth Function. The goodness of fit of the curves is assessed by the number of peaks and troughs in the length frequency data that the growth curve passes through. Several options are available in ELEFAN I for estimating the growth parameters. Once representative values of L_{∞} and K are described for a given series of length frequency data, the effects of trawl selectivity are used to correct the length frequency distribution. L_{∞} and K are then re-estimated using the new distribution. Inclusion of the smaller size classes enables a more accurate estimate of L_{∞} and K (Pauly 1987).

Values of L_{∞} and K were also extracted from the literature for comparative purposes. FishBase (FishBase 1995), a global fish database with data on growth rates of different fish species, was used for this purpose. For each species analysed using the above method, mean L_{∞} and K values were calculated from comparable studies. These mean values, which can be considered “generic values” for these species, were calculated with the aid of the auximetric parameter ϕ' (phi prime), which expresses the growth performance of a species or family of species (Munro and Pauly 1983, Pauly and Munro 1984, Pauly 1991). The equation relating ϕ' to L_{∞} and K :

$$\phi' = \log_{10}K + 2 \log_{10} L_{\infty}$$

ϕ' and L_{∞} are normally distributed and K has a log normal distribution. In order to estimate the average K from a series of L_{∞} and K values, ϕ' is calculated for each combination and the mean of the ϕ' and L_{∞} values taken. The above formula for ϕ' is then solved for K , to find the mean K .

These “generic” values of L_{∞} and K were used to check the results of the length frequency estimation of growth parameters. In addition, they were used as a guide in the analysis if the parameter definition was confused (for example, no clear definition of K). Tandog *et al.* (1988) who also used this approach for Philippine fishes, concluded that values of ϕ' are normally and narrowly distributed within a species.

Mortality Estimation using the Length-Converted Catch Curve

Length-converted catch curves have become one of the standard methods of estimating mortality (Gulland and Rosenberg 1992). Catch curves originated with the use of length based data by T. Edser in 1908 and F. Heinke in 1913, who also incorporated growth rate information (Ricker 1975). However, the modern form is based on age data and derived from the catch equation:

$$C(t_1, t_2) = N(t) \cdot F/Z \cdot (1 - \exp(-Z(t_2 - t_1))) \quad (1)$$

where,

$C(t_1, t_2)$ = catch from time t_1 to time t_2 ,

$N(t)$ = numbers at time t ,

F = instantaneous fishing mortality rate

Z = instantaneous total mortality rate

The catch equation is log transformed and manipulated into a form called the “general linearised catch curve”.

$$\text{Ln}C(t_1, t_2) = \ln(N(t_r) \cdot F/Z) + Z(t_r) - Z(t_1) + \ln(1 - \exp(-Z(t_2 - t_1))) \quad (2)$$

where,

$N(t_r)$ = number at age of recruitment

$Z(t_r)$ = instantaneous total mortality rate at age of recruitment.

The first term on the right hand side, “ $\ln(N(t_r) \cdot F/Z) + Z(t_r)$ ” can be consolidated as a constant, leaving “ $-Z(t_1) + \ln(1 - \exp(-Z(t_2 - t_1)))$ ”. The second part of this term is non-linear unless the time interval is constant. However, the time to grow from one length interval to the next is not constant in fish and in order to make this non-linear term linear, an additional assumption is made. For small values of “ x ”, that is $x < 1$, the following holds true:

$$\ln(1 - \exp(-x)) \approx \ln(x) - x/2,$$

and therefore where $Z(t_2-t_1)$ is small, it follows that:

$$\ln(1-\exp(-Z(t_2-t_1))) = \ln[Z(t_2-t_1)] - Z(t_2-t_1)/2 \quad (3)$$

Substituting this into the general linearised catch curve, substituting delta "t" for (t_2-t_1) and incorporating $\ln Z$ into the constant gives:

$$\ln[C(t, t+\Delta t)/\Delta t] = c - Z(t + \Delta t/2) \quad (4)$$

or,

$$Y = c - Zx.$$

where "c" is the constant and "-Z" is the slope of the descending right arm of the curve.

Equation (4) is written in terms of age. In order to convert it into a length based catch curve equation, length is transformed to age using the inverse von Bertalanffy equation, that is:

$$t_L = t_0 - 1/K * \ln(1 - L/L_\infty),$$

where,

K = constant of the von Bertalanffy Growth Function (VBGF),

L_∞ = asymptotic length at infinite age.

Substituting this into equation (4) produces the Length Converted Catch Curve equation below.

$$\ln[C(L_1, L_2) / \Delta t(L_1, L_2)] = c - Z[(t(L_1) + t(L_2))/2] \quad (5)$$

where,

L_1 = length at time 1,

L_2 = length at time 2,

$\Delta t = t(L_1) - t(L_2) = 1/K * \ln[(L_\infty - L_1)/(L_\infty - L_2)]$

L_∞ = asymptotic length at infinite age.

The catch of fish of length L_1 to L_2 is thus divided by the time it takes to grow through this length class, and this is plotted against the age of the fish which is $(t(L_1) + t(L_2))/2$. The slope of the descending right limb gives the estimate of Z .

In the method just described, it is assumed that the ecosystem is in a steady state, that the sample represents the mean population structure, that Z is constant over all size classes, that recruitment fluctuations are small and random and that the gear used (in this case trawling gear) has a selection curve where only the smaller animals are selected against. In this case the trawl samples, taken over a period of 10 months, were pooled to produce the sample for analysis. This should enable the second assumption to be met. The third and fourth assumptions are usually met if the descending limb appears straight (Pauly *et al.* 1980). In the case of San Miguel Bay, the mean size of fish are sufficiently small and fishing pressure has been sufficiently intense to virtually guarantee that none of the fish included in the sample

were large enough to be selected against. The assumption of a steady-state is always precarious and should be used with caution.

The next three methods share the assumptions described for the Length Converted Catch Curve.

Mortality Estimation using Beverton and Holt's Mean Length Method

The Beverton and Holt Model (Beverton and Holt 1956) uses mean length, L_{∞} and K to estimate Z . This method is robust under conditions of variable recruitment (Wetherall *et al.* 1987).

$$Z = K * (L_{\infty} - \bar{L}) / (\bar{L} - L_c),$$

where,

K = constant of the von Bertalanffy Growth Function (VBGF),

L_c = the length for which all fish that length and longer in the catch are under full exploitation,

\bar{L} = mean length of fish in the sample from L_c to L_{\max} .

Mortality Estimation using the Powell-Wetherall Plot

Another method which uses mean length information, is the Powell-Wetherall Plot (Gayaniilo *et al.* 1996). It requires the whole sample to be pooled and assumes that this pooled sample represents the equilibrium state. It is a graphical method where the mean length of fish above L_c , \bar{L} minus L_c ($\bar{L} - L_c$) is plotted against L_c . The equation on which this method is based is:

$$\bar{L} - L_c = a + b * L_c$$

where,

$$\bar{L} = (L_{\infty} + L_c) / (1 + (Z/K))$$

$$Z/K = -(1+b)/b \text{ or } b = -K/(Z+K),$$

$$L_{\infty} = -a/b, \text{ or, } a = -b * L_{\infty}$$

The slope of the line gives an estimate of Z/K and the intersect with the x-axis, L_{∞} .

Mortality Estimation using Fishing Mortality = Catch / Biomass

Under equilibrium conditions, fishing mortality, F , is equal to the quotient of the catch and biomass. Total mortality, Z , is the sum of F and natural mortality, M . Natural mortality was estimated using Pauly's M equation (Pauly 1980b), and F was estimated from the catch and the

swept area estimates of biomass (see below). This method was included as an alternative to the length frequency approaches above. However, the Pauly estimate of M does require the growth parameters L_{∞} and K .

Yield-per-Recruit Analysis

The mortalities estimated above provide a snapshot of the current status of some of the main fish species in San Miguel Bay, but they do not give information on how these mortality rates relate to the optimal rates of fishing, nor how much change in yield can be expected as a result of adjusting fishing mortality. This is obtained using a length-based yield-per-recruit analysis. The analysis was carried out using the relative yield-per-recruit model contained within the FiSAT software (Gayanilo *et al.* 1996). This model is based on a modified version of the Beverton and Holt yield-per-recruit model (Pauly and Soriano 1986). The required input parameters are a selection curve, the asymptotic length, L_{∞} , the von Bertalanffy growth parameter, K and natural mortality, M . The growth and mortality parameters were calculated in the Mortality Section above.

The selection curve should ideally be estimated from empirical data collected during the trawl survey⁵. However, this type of data was not collected. Instead a routine within FiSAT, was used to estimate the selection curve. The routine first estimates the probability of capture from the Length Converted Catch Curve (see above). The model uses the Z and the M estimates to

⁵ In order to estimate the selectivity of a trawl net, a fine meshed net is used to cover the cod-end and thus catch any fish which escape the cod-end of the trawl net.

back-calculate the numbers of fish that would have been present in the sample if no selectivity had taken place. The probability of capture is calculated as the ratio of the number of fish present with selectivity to the number of fish that would be present with no selectivity. The selection curve is then estimated using a moving average of probabilities for the age corresponding to three adjacent length classes, i.e., Probability at time (t) is equal to the average of probability at time (t-1), time (t) and time (t+1). The method assumes that the estimate of Z is accurate and that the smallest fish caught are fully recruited to the fishery (Isaacs 1990). Otherwise, the curve thus estimated is a "resultant", that is, the product of a selection with a recruitment curve (Gayanilo *et al.* 1996).

Yield-per-recruit analyses were conducted for the species in San Miguel Bay for which there were sufficient data. Current exploitation rates were compared to the optimal rates generated by the yield-pre-recruit analysis. Current rates were calculated as $F=(Z-M)/Z$, where Z is the resultant overall mortality calculated above and M is calculated from the empirical formula of Pauly (1980b).

Analyses of Fishery Data

Estimating Catch and Effort

One characteristic of tropical fisheries is that catch, particularly by the small-scale sector, is landed at small, numerous and often widely distributed landing sites. Although convenient for fishers and local buyers, this complicates the collection of fisheries data. Indeed it can make it impossible to collect comprehensive data. This is the situation in San Miguel Bay, which has landing sites dispersed all along its coastline. Despite the detailed study conducted by ICLARM, it was not possible to numerically account for all effort and all landings at all landing sites. Instead, both are estimated from samples.

Effort

For each gear type, Silvestre *et al.* (1995) calculated the total number of gear in each municipality from the information gathered in the Fishing Gear Inventory. They then summed these totals to obtain the number of each gear type in San Miguel Bay. They estimated the average number of trips made per year from information on seasonal use of gear in the Fishing Gear Inventory. The total effort, by gear, was then calculated from the product of the number of gears operating per year and the average number of trips made per year, that is:

$$\text{Effort}_{\text{gear}} = \text{number of units of gear} * \text{mean annual number of trips}$$

Catch

The total catch was estimated from the annual average catch per unit effort (CPUE) per gear multiplied by the average annual effort of that gear, that is:

$$\text{Annual catch per gear} = \text{CPUE (kg/trip)} * \text{effort (No. of trips)} * \text{No. Vessels (gear)}$$

CPUE per gear, was calculated from the sum of all landings made per gear over the survey period, divided by the total number of trips made per gear.

The total catch is simply the sum over all gears, that is:

$$\text{Total annual catch} = \sum_{\text{gear}} \text{annual catch per gear}$$

When estimating catch from sub-samples in such a diverse fishery, there is inevitable uncertainty around the resultant catch estimates. In order to include and examine these uncertainties, Monte Carlo simulation (Crystal Ball Inc., Denver, Colorado) was used to estimate means and 95% confidence limits of the estimates, as described above.

Error distributions were entered for all three parameters, that is, CPUE, Effort and Vessels/Gear. Two approaches were used:

1. a 10 % uniform probability distribution was used for all of the parameters

2. a 20% uniform probability distribution was used for the CPUE parameters whilst a 10 % uniform probability distribution was used for the other two parameters.

In order to compare the 1992-1994 catch data to the 1979-1982 catch, the 1979-1982 catch estimate was also subjected to Monte Carlo simulation and analysis.

Distribution of Catch, Effort and CPUE in the Fishery

Once the catch and the effort were estimated, several aspects of the distribution of catch, effort and CPUE in the fishery were examined. Informative comparisons with similar data from the 1979-1982 study were then made. These fall under the following headings:

Distribution of catch, effort and CPUE across gears

In this section, the distribution of the total catch, effort and CPUE across different gears was examined. This was then related to the distribution between the large and small-scale sectors.

Comparative Analysis of Species Composition from Landings Data

The overall species composition of the total catch was estimated and compared to the 1979-1982 data. The species composition of selected gears was examined and comparisons were made with the earlier data. Where possible, the changing patterns of use of fishing gear since 1979-1982 were also determined.

Species Composition and Distribution of the Catch by Fishing Gear and Season

In the Fishing Gear Inventory, a survey was undertaken to determine the seasonal use of fishing gears in San Miguel Bay. The results of this are examined and compared to the seasonality of the catch, the species composition and CPUE. The aim is to gain an understanding of the seasonal nature and operation of the fishery.

Status of the Major Species in San Miguel Bay

The CPUEs of each of the top ten species were compared across the major gears that catch them and compared to the 1979-1982 figures. The modal or mean lengths of the fish in the catch were examined where possible. In addition, their seasonal abundance was examined.

Results

Analyses of Trawl Survey Data

The Species Composition of San Miguel Bay

In total, 55 valid trawls were made during the trawl survey over a period of 10 months (September 1992 to June 1993). Although there was an intended standard trawl time of 1 hour, (Cinco *et al* 1995) in practise there was considerable variability; the modal trawling time was

30-40 minutes (31 out of 55 hauls) with mean of 46 minutes and median of 35. The number of trawls made per month varied from 2 to 7 and trawling time per month varied from 2 hours to 6 hours.

A total of 98 species, from 46 families were recorded and identified during the course of the trawl survey. However, despite this large degree of diversity, the leiognathid, *Leiognathus splendens* was easily recognised as the most abundant species (Table 2.1). It comprised 16.5% of the total. Another leiognathid, *Secutor ruconius* was the second most abundant species and accounted for 11.1% of the total abundance. In fact, over 60% of the total CPUE was produced by only 10 of the 98 species recorded in the trawl survey, as shown in Table 2.1. Three leiognathids are included in these 10 species.

In Table 2.2 the top ten families in San Miguel Bay account for 79% of the trawl survey abundance. The families are the same as those represented in Table 2.1, with the addition of the Tetraodontidae and a group, called here the Benthic Invertebrates. It is clear from these figures that the Leiognathidae are, by far, the most abundant family in the trawlable biomass of San Miguel Bay. The Trichiuridae, Sciaenidae and Engraulidae each have about 20% of the abundance of the leiognathids, whilst the rest of the families shown in Table 2.2 are less than 14% as abundant as the leiognathids.

Table 2.1 The ten most abundant species in San Miguel Bay (1992-1994 Trawl Survey data).

| <i>Species</i> | <i>Family</i> | <i>% of total CPUE</i> | <i>Accumulative %</i> |
|--------------------------------|---------------|----------------------------|---------------------------|
| <i>Leiognathus splendens</i> | Leiognathidae | 16.5 | 16.5 |
| <i>Secutor ruconius</i> | Leiognathidae | 11.1 | 27.6 |
| <i>Trichiurus haumela</i> | Trichiuridae | 7.4 | 35.0 |
| Penaeid Shrimps* | Penaeidae | 4.9 | 39.9 |
| <i>Otolithes ruber</i> | Sciaenidae | 4.4 | 44.3 |
| <i>Scomberomorus commerson</i> | Scombridae | 4.3 | 48.6 |
| <i>Stolephorus commersonii</i> | Engraulidae | 3.7 | 52.4 |
| <i>Nemipterus japonicus</i> | Nemipteridae | 3.0 | 55.4 |
| <i>Drepane punctata</i> | Ephippidae | 3.0 | 58.4 |
| <i>Leiognathus bindus</i> | Leiognathidae | 2.8 | 61.2 |

* The Penaeid species are grouped together, following the procedure used in the 1979-1982 study.

Table 2.2 The ten most abundant families in San Miguel Bay (1992-1994 Trawl Survey data).

| <i>Family</i> | <i>% of total CPUE</i> | <i>Accumulative %</i> |
|------------------------|----------------------------|-----------------------|
| Leiognathidae | 35.2 | 35.2 |
| Trichiuridae | 7.4 | 42.6 |
| Sciaenidae | 6.8 | 49.4 |
| Engraulidae | 6.4 | 55.8 |
| Penaeidae | 4.9 | 60.7 |
| Scombridae | 4.4 | 65.1 |
| Tetraodontidae | 4.0 | 69.1 |
| Benthic Invertebrates* | 3.9 | 73.0 |
| Nemipteridae | 3.0 | 76.0 |
| Ephippidae | 3.0 | 79.0 |

* Benthic Invertebrates include Bivalves, Shellfish, Brittlestars, Jellyfish and Nudibranchs.

Estimating Density and Biomass from the Trawl survey Data

The monthly catch rates and the results of the density and trawlable biomass estimation without Monte Carlo simulation are given in Table 2.3. Using the most likely parameters, the mean annual density is 1.56 t/km^2 . There is considerable variation in the monthly estimates from which this annual figure is estimated. Mean monthly density estimates range from 0.55 t/km^2 in May to 3.34 t/km^2 in October.

The Monte Carlo simulation produced a mean density estimate of 2.72 t/km^2 , with a 95% certainty of 1.21 to 8.67 t/km^2 , and a standard deviation of 2.96 (Table 2.4). This corresponds to a mean biomass estimate of 3304 t if the San Miguel Bay area is taken as having a surface area of $1,115 \text{ km}^2$. The 95% certainty range was $1,351$ to $9,653 \text{ t}$ with a standard deviation of $3,303$. The corresponding mean biomass for an area of 840 km^2 was $2,281 \text{ t}$. Cinco *et al.* (1995) estimated a density of 1.96 t/km^2 and a biomass of $1,646$ tonnes (for an area of 840 km^2), using the swept area method described above. This falls within the 95% certainty range of the estimate produced here and can be reproduced by using a value of 0.6 for either X_1 or X_2 , the escapement factor and the effective width of the trawl respectively.

The results are very sensitive to the input parameters. The greatest sensitivity was to the value of X_1 (the escapement factor) (93.6% of variation), then to the value of X_2 (the effective width of the swept area) (3.5%) and the least sensitivity is to L , the distance travelled in a one hour trawl (2.9%), measured by rank correlation. The parameter to which

Table 2.3 Monthly Catch Rate (Trawl Survey), Density and Biomass Estimates for San Miguel Bay, September 1992 to June 1993

| <i>Month</i> | <i>Area of San Miguel Bay</i> | | | |
|--------------|-------------------------------|---------------------------|---------------------------|--------------------------|
| | <i>Catch Rate</i> | <i>Density</i> | <i>1115km²</i> | <i>840km²</i> |
| | <i>(kg/hr)</i> | <i>(t/km²)</i> | <i>Biomass (t)</i> | |
| September | 69.64 | 2.82 | 3142.9 | 2367.7 |
| October | 82.63 | 3.34 | 3729.5 | 2809.6 |
| November | 47.02 | 1.90 | 2121.9 | 1598.6 |
| December | 18.89 | 0.76 | 852.7 | 642.4 |
| January | 30.51 | 1.24 | 1377.1 | 1037.4 |
| February | 27.03 | 1.09 | 1219.9 | 919.0 |
| March | 31.58 | 1.28 | 1425.1 | 1073.6 |
| April | 14.94 | 0.60 | 674.4 | 508.0 |
| May | 13.48 | 0.55 | 608.3 | 458.3 |
| June | 49.34 | 2.00 | 2226.8 | 1677.6 |
| MEAN | 38.51 | 1.56 | 1737.8 | 1309.2 |

Table 2.4 Results of Density and Biomass Estimation in San Miguel Bay using Monte Carlo Simulation

| <i>Statistic</i> | <i>Density</i> | <i>Biomass</i> | |
|--------------------|---------------------------|------------------------------|-----------------------------|
| | <i>(t/km²)</i> | <i>A=1115 km²</i> | <i>A=840 km²</i> |
| Mean | 2.72 | 3,027 | 2,281 |
| Standard Deviation | 2.96 | 3,304 | 2,489 |
| Lower 95% Limit | 1.21 | 1,351 | 1,019 |
| Upper 95% Limit | 8.67 | 9,653 | 7,290 |
| Range Width | 51.2 | 57,092 | 43,011 |

the results were most sensitive, X_1 , is also the parameter which had the widest input range and the parameter for which there is least certainty. Indeed, given that the standard deviations produced by the Monte Carlo simulation are as large as the mean, there can be little certainty about any of the estimates given above.

The density estimate of 2.13 t/km^2 from the 1979-1982 study was also estimated using the swept area method (Vakily 1982). This value falls within the 95% certainty range of the 1992-1994 estimate. However, given the uncertainty surrounding these estimates, it is not possible to comment on whether there has been a change in density or biomass since 1979-1982. Monte Carlo analysis of the 1979-1982 trawl data would give a similarly wide distribution of density estimates since the parameter to which the results is most sensitive, X_1 would remain the same.

Pauly (1982a) collated a table of historical trawl surveys conducted in San Miguel Bay and calculated the density and biomass using the swept area method described above. The first trawl survey was conducted in 1948 (Warfel and Manacop 1950) in the month of July. The survey consisted of five drags of approximately 1 hour duration using a 30 m, 400 HP trawler with a 10 cm meshed cod end. Allowing for differences in gear and mesh sizes, Pauly estimated a density of 10.6 t/km^2 and stock biomass of 8,900 tonnes, figures he described as conservative. Although these figures should only be used as an indicator of likely density (since the survey was limited to one month and there were differences in the gears used) the 1948 estimates do fall outside the confidence limits of the 1992-1994 density and biomass estimates. This indicates that the density and biomass in San Miguel Bay have decreased since

1948. Using the mean density estimate of 2.72 t/km^2 for 1992-1994 as a guide, the density in San Miguel Bay can be said to have decreased to around 26% of the 1948 density.

The density estimates from the other trawl surveys described in Pauly (1982a) ranged from 5.2 t/km^2 in 1957-58 to 3.49 t/km^2 in 1977. These values fall within the 95% certainty range of the 1992-1994 density estimate. They therefore cannot be said to be different from the 1992-1994 estimate. In addition, most of these surveys, like the 1948 survey, only took place over one month and should be interpreted with similar caution. The results suggest that there has been a downward trend in density with time. However, given the uncertainty of density estimation using the swept area method shown above, the inconsistent sampling methodology between surveys and the sparsity of seasonal coverage for the 1960s and 1970s surveys, it is not possible to make any conclusions about trends in density or biomass over the time period of these surveys. On the basis of the data presented here, it is as likely as not that the total density and biomass in San Miguel Bay have not significantly changed since the late 1950s.

Given the inaccuracy of the swept area method for density estimation, another, less direct means of comparing biomass over time would be to use CPUE as a measure of abundance. However, even this is fraught with difficulty since recent work concludes that for many fish, especially schooling fish, there is no direct relationship between CPUE and abundance (Hilborn and Walters 1992), that is, CPUE does not accurately track abundance. A dramatic decline in abundance can occur while CPUE remains stable (Pitcher 1997, Mackinson *et al. in press*). A decline in CPUE is a clear signal that abundance has decreased.

The CPUE of the 1948 trawl survey was 636 lb/hr (Warfel and Manacop 1959), that is, 289 kg/hr. This is considerably higher than the largest CPUE of 82.63 kg/hr in Table 2.3 above. The ratio of the largest 1992-1994 CPUE to the 1948 CPUE is 28%; using the mean CPUE for 1992-1994, the ratio falls to 13%. So, using the CPUE data for comparative purposes, it appears that the 1992-1994 CPUE rate is about 13-28% of the 1948 CPUE.

The mean CPUE rates in the 1979-1982 trawl data were 30.77 kg/hr for the period from March 1979 to February 1980 and 36.17 kg/hr for March 1980 to February 1981. These figures are both in close agreement with the 1992-1994 mean CPUE of 38.51 kg/hr. These results indicate that there has been little change in the total CPUE since the early 1980s.

Longitudinal Comparison of Species Compositions from Trawl Survey Data

A more telling analysis of changes in biomass over time can be had by examining the changes in species composition over time. The seminal work of Pauly (1979a) on the Gulf of Thailand and Andaman Sea fisheries demonstrated that changes in the species composition of a fishery can reflect changes in an ecosystem caused by fishing, potentially leading to ecosystem overfishing.

A comparison to the 1947 trawl survey data (Warfel and Manacop 1950; see also Cinco *et al.* 1995) and the 1979-1982 trawl data is made in Tables 2.5 and 2.6. The original species

recorded in the 1947 survey are used, plus those families which have substantially increased since 1947.

The relative abundance of the leiognathids was twice as high in San Miguel Bay in 1947 as in 1992-1994 (Table 2.5). Furthermore, the catch rate has substantially decreased from 173 kg/hr in 1947 to 13.4 kg/hr in 1992-1994. These results correspond with Pauly's (1979a) conclusions about the species changes that occurred in the Gulf of Thailand as a result of a trawl fishery, that is, that the bulk of small prey fish diminish at a much greater pace than the other species in the ecosystem. In San Miguel Bay, the Pristidae⁶, sharks and rays, Pomadasyidae, Sphyrinae, Ephippidae and Ariidae have all also markedly decreased in relative abundance and/or catch rate. Again, this is as predicted by Pauly, that is the rays and the medium predator fish, dependent on the small prey fish for food, decrease.

The relative abundance of other groups, such as the Trichiuridae, Scombridae, and squids, have increased since 1947 (although the CPUE of the Scombridae has decreased). These correspond to the pelagic fish group noted by Pauly as unlikely to be reduced by bottom trawl. Pauly also identified a group consisting of species likely to increase as a result of demersal trawl fishing. These include "r" strategists such as flat fish and benthivores. There are many species present in the 1992-1994 trawl survey that fall into this category, including the Benthic Invertebrates, Psettodidae, crabs and Tetraodontidae. Many of these were not recorded in either the 1947 or the 1979-1982 surveys. It is not clear whether these species

⁶ The Pristidae in the 1947 trawl survey consisted of one specimen weighing 400 pounds. Since it was the only sawfish caught during the 5 drags conducted in the survey, the abundance of sawfish may be less than suggested by the figure in Table 2.8

Table 2.5 Comparison of Species Composition of Trawl Survey data from 1947, 1979-1982 and 1992-1994.

| <i>Taxon</i> | <i>Species Composition</i> % | | | | <i>Direction of Change</i> |
|-----------------|---------------------------------|----------------|----------------|----------------|--------------------------------|
| | <i>1947</i> | <i>1979/80</i> | <i>1980/81</i> | <i>1992/94</i> | |
| Leiognathidae | 60.0 | 31.1 | 31.8 | 35.2 | ↓↑ |
| Pristidae | 15.4 | - | - | - | ↓ |
| Sharks and Rays | 6.3 | 0.7 | 0.5 | 0.0 | ↓ |
| Pomadasyidae | 2.3 | 0.7 | 0.3 | 0.2 | ↓ |
| Sphyrinaeidae | 1.6 | - | - | 0.8 | ↓ |
| Sciaenidae | 0.9 | 12.2 | 10.8 | 6.8 | ↑↓ |
| Ephippidae | 0.9 | - | - | 3.0 | ↑? |
| Ariidae | 0.8 | 0.7 | 0.1 | 0.2 | ↓ |
| Synodontidae | 0.8 | - | - | 1.6 | ↑? |
| Mullidae | 0.7 | - | - | 2.4 | ↑? |
| Scombridae | 0.7 | 0.4 | 0.4 | 4.4 | ↑? |
| Carangidae | 0.6 | 2.7 | 0.9 | 2.8 | ↑ |
| Trichiuridae | 0.1 | 2.1 | 3.9 | 7.4 | ↑ |
| Engraulidae | - | 19.5 | 21.7 | 6.4 | ↓ |
| Penaeidae | - | 5.9 | 6.6 | 4.9 | ↓ |
| Mugilidae | - | 7.1 | 4.7 | 1.6 | ↓ |
| Squids | - | 2.8 | 3.3 | 1.5 | ↓ |
| Clupeidae | - | 4.4 | 3.0 | 0.3 | ↓ |
| Others | 8.8 | 9.7 | 12.0 | 20.5 | ↑ |
| SUM | 100.00 | 100.00 | 100.00 | 100.00 | - |

Sources for historical data: Warfel and Manacop (1950), Vakily (1982).

Table 2.6 Comparison of Catch Rate from Trawl Survey data from 1947, 1979-1982 and 1992-1994.

| <i>Taxon</i> | <i>Catch Rate kg/hr</i> | | | | <i>Change of Direction</i> |
|-----------------|-----------------------------|---------|---------|---------|--------------------------------|
| | 1947 | 1979/80 | 1980/81 | 1992/94 | |
| Leiognathidae | 173.34 | 9.58 | 11.50 | 13.40 | ↓↑ |
| Pristidae | 44.38 | - | - | - | ↓ |
| Sharks and Rays | 18.31 | 0.22 | 0.19 | 0.02 | ↓ |
| Pomadasyidae | 6.66 | 0.22 | 0.12 | 0.08 | ↓ |
| Sphyraenidae | 4.62 | - | - | 0.30 | ↓ |
| Sciaenidae | 2.68 | 3.73 | 3.91 | 2.70 | ↑↓ |
| Ephippidae | 2.59 | - | - | 1.49 | ↓? |
| Ariidae | 2.40 | 0.20 | 0.03 | 0.06 | ↓ |
| Synodontidae | 2.31 | - | - | 0.60 | ↓? |
| Mullidae | 2.13 | - | - | 0.91 | ↓? |
| Scombridae | 1.94 | 0.11 | 0.15 | 1.65 | ↓↑ |
| Carangidae | 1.85 | 0.84 | 0.32 | 1.07 | ↓↑ |
| Trichiuridae | 0.28 | 0.66 | 1.42 | 2.81 | ↑ |
| Engraulidae | - | 5.99 | 7.84 | 2.42 | ↓ |
| Penaeidae | - | 1.83 | 2.40 | 1.94 | ↔ |
| Mugilidae | - | 2.18 | 1.68 | 0.16 | ↓ |
| Squids | - | 0.86 | 1.20 | 0.58 | ↓ |
| Clupeidae | - | 1.34 | 1.09 | 0.11 | ↓ |
| Others | 25.42 | 3.01 | 4.32 | 7.66 | ↓↑ |
| SUM | 288.91 | 30.77 | 36.17 | 37.97 | - |

Historical data sources: Warfel and Manacop (1950), Vakily (1982)

were not recorded because they were not present in the previous surveys or because they were not identified and therefore classified in the "others" category.

In general then, the species composition changes seen in San Miguel Bay confirm the pattern outlined by Pauly (1979a). Certainly by the time of the 1979-1982 survey, San Miguel Bay could be said to be suffering from ecosystem overfishing. However, the results do not indicate that the situation has necessarily worsened since then. The changes in species composition and in catch rates that have occurred between 1979-1982 and 1992-1994 are less clear. However, a note of caution should be sounded here regarding the 1979-1982 trawl data. These data came from the commercial fishery, not a systematic trawl survey such as was carried out in the 1992-1994 study, or indeed in 1947. Thus, there is a likely bias in the species composition of the 1979-1982 trawl data. With this in mind, some groups, such as the Ehippidae, Synodontidae, Mullidae, carangidae and Trichiuridae follow a clear upwards trend from 1947 to 1992-1994. However, the relative biomass and catch rate of the leiognathids, for example, have marginally increased since 1979-1982. The relative biomass and catch rate of others species which were not recorded in the 1947 survey⁷, such as the Penaeidae, Engraulidae, Clupeidae and squids, have decreased since 1979-1982. This suggests that those species which succeeded the traditional species outlined above are now themselves being succeeded. This mixed picture which may be due to differences in sampling methodology rather than real differences.

⁷ These species may have been present in the Bay, but not caught in the 1947 trawl survey because of the large mesh size used in the trawl net.

The Sciaenidae are worth a separate note. The relative biomass of the Sciaenidae increased from 0.9% in 1947 to 10.8-12.2 % in 1979-1982, then decreased by almost 50% to 6.8% in 1992-1994. However, the catch rate of 2.7 kg/hr in 1992-1994 is approximately equal to the catch rate of 2.68 kg/hr in 1947. The catch rate in 1979-1982 was higher, at 3.73-3.91 kg/hr.

As noted above many species are recorded in the 1992-1994 trawl survey⁸ which were not previously recorded. Does this mean that there is a greater diversity in 1992-1994 than there was in earlier years? Without definitive data for the earlier years it is not possible to answer this question categorically. However, if the species/families are matched one for one in 1947, 1979-1982 and 1992-1994, the “Others” category has the highest abundance in 1992-1994 and the highest catch rate in 1947 (Tables 2.5 and 2.6). So, although there has been an overall reduction in CPUE since 1947, the abundance of the “Others” groups has increased and so has, therefore, the abundance of “Other” groups which are now identified in the 1992-1994 surveys (see Appendix for a list of species present in San Miguel Bay).

Seasonal Analysis of the Trawl Survey Data

Table 2.3 above gives the monthly catch rate from the trawl survey. It is unfortunate that the survey did not cover the full 12 months of the year, and even more so that the 2 months that are missing are reputed to be the most productive (Villanoy *et al.* 1995). However, despite

⁸ This is also seen in the landings survey.

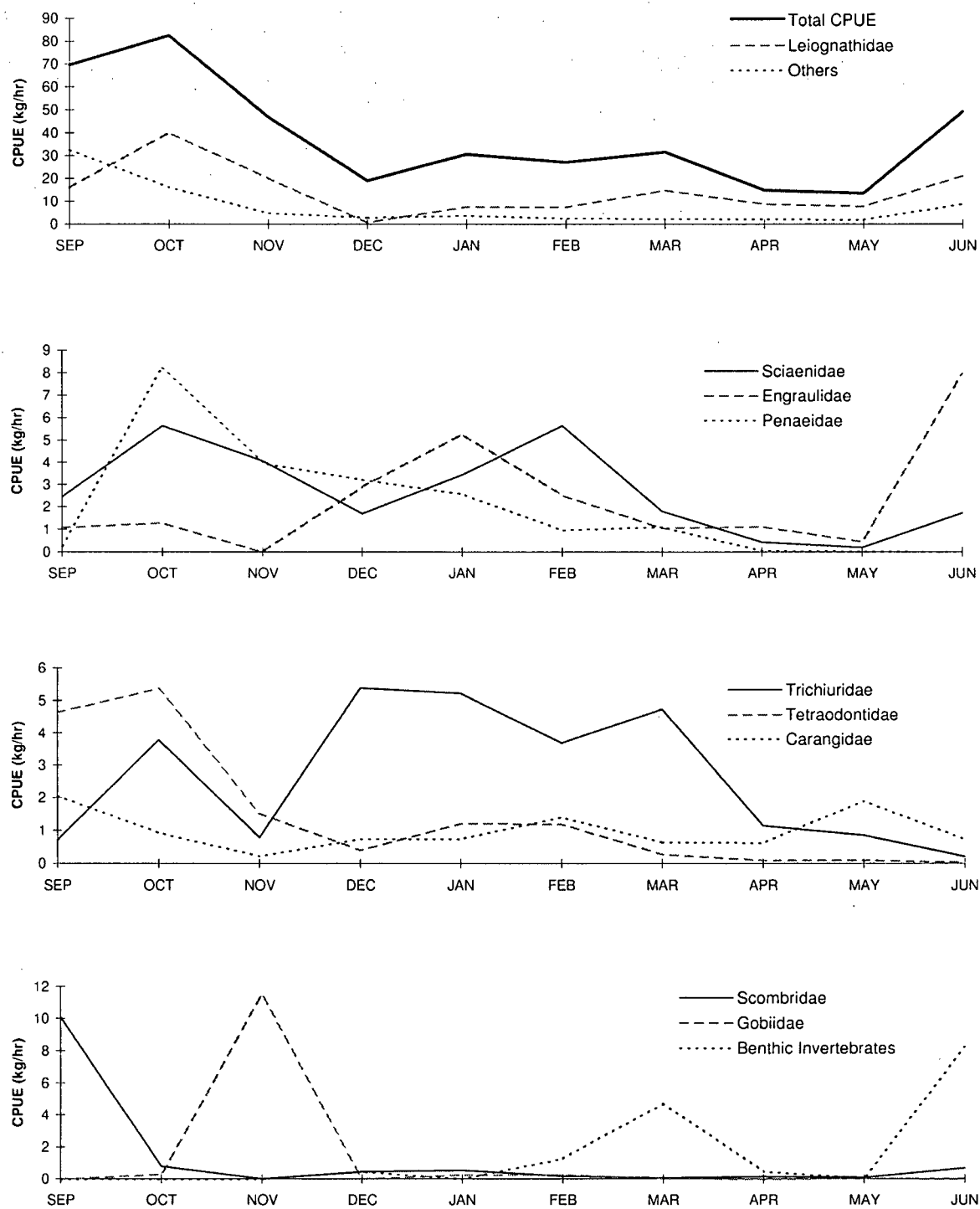


Figure 2.3 Seasonal variation in the CPUE of the major groups in the 1992-1994 Trawl Survey.

the missing data, there are clear trends through the year (Figure 2.3).

The total CPUE is high in September and October, but begins a decline (beginning of the northeast monsoon) to reach a low in December. This low CPUE prevails until May, creating a low CPUE plateau. In May, the CPUE rises again to June and the trend indicates that it would continue to rise to meet the September high. The trawl CPUE data from 1979-1982 (figure 3 in Vakily 1982) follows a similar pattern and supports this assumption. In September/October the CPUE decreased until November/December when it began to increase. In this case there is a CPUE high plateau from February/March until September/October.

The trend described above is determined by the leiognathids, and, to a lesser extent, the "Others" group. (Figure 2.3). The seasonal pattern for the other top ten Families is more variable. The Sciaenidae CPUE is low during March to June and variable between September and February. The Penaeidae CPUE peaks in October and then steadily declines through to April, May and June when it is very low. The Engraulidae CPUE peaks in January and June. The Trichiuridae CPUE basically increases from September to December (with a dip in November) and remains high until March when it decreases to a low rate in June. In contrast to the other groups represented in Figure 2.3, the Trichiuridae are most abundant during the northeast monsoon. The carangidae, the Scombridae, the Gobiidae and the Benthic Invertebrates show no clear trend. The CPUE of these groups is sporadic.

Unfortunately there is no monthly trawl CPUE by species from the 1979-1982 study available in the literature with which to compare these results. The results indicate that for most of the

more abundant species in San Miguel Bay, CPUE decreases during the northeast monsoon and rise to a higher level during the summer months.

Estimation of Mortality

Of the 98 species in the trawl survey, it was only possible to estimate the growth parameters and mortality for 6 species. These were the most abundant species in the trawl survey, that is, *Leiognathus splendens*, *L. bindus*, *Secutor ruconius*, *Scomberomorus commerson*, *Otolithes ruber* and *Trichiurus haumela*. The length frequency data for the other species in the trawl survey were not sufficient to define their growth parameters, even with the aid of generic values from the literature as a guide.

Leiognathus splendens

The length frequency analysis gave a range of combinations of L_{∞} and K for *Leiognathus splendens*. Using the surface scan option, a “banana” of correlated L_{∞} and K values, with high goodness of fit resulted, stretching from an L_{∞} of 13 - 16 cm and K values from 1.5-1.0 year⁻¹. These are in agreement with values in the literature. The average values from the literature were approximately $L_{\infty} = 14$ cm and $K = 1.0$ year⁻¹. Further analysis of the data using the automatic search routine and the K -scan produced estimates of $L_{\infty} = 13.2$ cm and $K = 1.45$ year⁻¹. Both combinations of growth parameters were used to refine the estimates by allowing for selectivity producing, for each combination two further estimates of L_{∞} and K plus the

average values from the literature. Table 2.7 below lists the L_{∞} and K estimates. Z estimates from the pooled original length-frequency sample were then calculated from the growth parameters and 95% confidence limits generated.

Using the Beverton and Holt mean length method it was not possible to definitively judge which of two points were the cut-off length above which all fish could be considered to be fully selected. For this reason, the calculations were made using both points. The results in Table 2.7 for the Beverton and Holt method are similar to those of the Length Converted Catch Curve, although using the slightly higher L_c of 6.5 cm produces a Z estimate which is 2.6 times higher than that obtained by using a L_c of 6.0 cm. This indicates that this method is sensitive to the value used as L_c .

The catch/biomass estimate of Z is considerably lower than the Z estimates just described: $M = 2.61 \text{ year}^{-1}$ and $F = 2.60 \text{ year}^{-1}$. The Wetherall *et al.* method did not produce realistic values of L_{∞} and is thus not included with the results. Since the Length Converted Catch Curve and the Beverton and Holt's methods give similar results, the average of these, $Z = 11.3 \text{ year}^{-1}$, was assumed to be a representative, if high, estimate of total mortality for *L. splendens*.

Table 2.7 Results of the length frequency analysis and mortality estimation for *Leiognathus splendens*

| Growth Parameters | | Length Converted Catch Curve | | | Beverton and Holt (1956) | |
|------------------------|----------------------------|------------------------------|-------------|-------------|----------------------------|----------------------------|
| | | | | | L _c = 6.5 cm | L _c = 6.0 cm |
| L _∞ (cm) | K (year ⁻¹) | Z (year ⁻¹) | Lower CI | Upper CI | Z (year ⁻¹) | Z (year ⁻¹) |
| 13.0 | 1.5 | 11.5 | 8.9 | 14.0 | 14.1 | 11.3 |
| 13.2 | 1.45 | 11.7 | 9.1 | 14.2 | 14.1 | 11.2 |
| 13.6 | 1.0 | 8.8 | 6.9 | 10.7 | 10.4 | 8.2 |
| 14.0 | 1.3 | 12.5 | 9.9 | 15.0 | 14.3 | 11.3 |
| 14.0 | 1.0 | 9.6 | 7.6 | 11.6 | 11.0 | 8.7 |
| Mean Estimate of Z | | 10.8 | — | — | 12.8 | 10.2 |

Table 2.8 Results of the length frequency analysis and mortality estimation of *Secutor ruconius*.

| Growth Parameters | | Length Converted Catch Curve | | | Beverton and Holt (1956) | Powell Wetherall Plot |
|------------------------|----------------------------|------------------------------|-------------|-------------|----------------------------|----------------------------|
| | | | | | L _c = 4.5 cm | L _c = 4.5 cm |
| L _∞ (cm) | K (year ⁻¹) | Z (year ⁻¹) | Lower CI | Upper CI | Z (year ⁻¹) | Z (year ⁻¹) |
| 8.4 | 1.25 | 7.83 | 6.75 | 8.91 | 6.44 | 6.78 |

Table 2.9 Results of the length frequency analysis and mortality estimation of *Leiognathus bindus*

| Growth Parameters | | Length Converted Catch Curve | | | Beverton and Holt (1956) | |
|------------------------|----------------------------|------------------------------|-------------|-------------|----------------------------|----------------------------|
| | | | | | L _c = 3.5 cm | L _c = 4.5 cm |
| L _∞ (cm) | K (year ⁻¹) | Z (year ⁻¹) | Lower CI | Upper CI | Z (year ⁻¹) | Z (year ⁻¹) |
| 12.2 | 1.2 | 7.96 | 6.56 | 9.37 | 7.39 | 8.4 |

Secutor ruconius

The length frequency analysis indicated that the growth parameters of *S. ruconius* lie in the range of $L_{\infty} = 8.3 - 8.9$ cm and $K = 1.0 - 1.3$ year⁻¹. Further analysis produced the optimal combination of $L_{\infty} = 8.3$ cm and $K = 1.3$ year⁻¹, and, after allowing for selectivity, $L_{\infty} = 8.4$ cm and $K = 1.25$ year⁻¹. The latter is in close agreement with the average growth parameter values for *S. ruconius* in the literature, $L_{\infty}=8.4$ cm and $K=1.41$ year⁻¹. The Length Converted Catch Curve gives a Z estimate of 7.83 year⁻¹.

The Beverton and Holt method gives a lower estimate of Z than the estimate by Length Converted Catch Curve and indeed does not fall within the confidence bounds (Table 2.8). However the Z estimate of 6.78 year⁻¹ produced by the Powell-Wetherall plot does lie within the bounds of the confidence limits of the Length Converted Catch Curve estimate. All three estimates could be said to be comparable, in the range of 6.5 - 7.8 year⁻¹. The catch / biomass Z estimate of 3.29 year⁻¹ is much lower than this. This is because the estimated fishing mortality of 0.14 year⁻¹ is low, due to the small catch recorded in the landing survey⁹. For this reason, the results from this method were not included in the Table 2.8. The overall total mortality derived for *S. ruconius* was $Z=7.02$ year⁻¹, the average of the first three estimates.

⁹ This is something of an anomaly since *S. ruconius* is well represented in the trawl survey. Compared to the high Z estimates from the other 3 methods, a higher catch would be expected.

Leiognathus bindus

The analysis of the *Leiognathus bindus* length frequency data produced an estimate of $Z = 7.93 \text{ year}^{-1}$. It was not possible to discriminate any one set of growth parameters from the length/frequency data alone, although a “banana” of values were identified using the surface scan routine. The average values from the literature were included however in the optimal range of parameter combinations and an $L_{\infty} = 12.2 \text{ cm}$ and $K = 1.2 \text{ year}^{-1}$ were used for the Length Converted Catch Curve and the Beverton and Holt Method (Table 2.9). Two cut-off points were used for the Beverton and Holt method, giving an average Z of 7.9 year^{-1} . Neither of the two other methods produced credible results. The catch/biomass method gave no result because the recorded catch of *L. bindus* in the Bay is minimal. The Powell-Wetherall Plot produced a higher L_{∞} of 14.9 cm , but it was not possible to estimate K .

Scomberomorus commerson

There were very few data for *Scomberomorus commerson* (123 observations over the 10 month period, with some months having as few as two records). However, the average growth parameters values from the literature fitted the length frequency data well. These generic values were used as the growth parameters for *S. commerson* for the mortality estimation. The results are given in Table 2.10.

The results are not very consistent. The Length Converted Catch Curve gives a Z value of 5.27 year^{-1} , intermediate between the two other results. The catch / biomass Z estimate is low, and

outside the confidence limits of the Length Converted Catch Curve. However, the Beverton and Holt total mortality estimate of 9.64 year^{-1} is high. It is dependent on the maximum length in the sample used, L_{\max} . The longer length groups are poorly represented in this sample and were not included in the analysis. Reducing L_{\max} to the longest well represented length group, (TL = 24 cm) produces a high estimate of Z . The mortality estimate for *S. commerson* is thus very uncertain. The length frequency data give widely different results when different methods are used.

Otolithes ruber

It was not possible to discriminate a set of growth parameters with the length frequency data from the trawl survey. Growth parameters from the 1979-1982 study of San Miguel Bay (Navaluna 1982), were fitted to the *Otolithes ruber* length frequency data. However, the fit was not good and the L_{∞} of 35.5 cm seemed to be too low. In the 1979-1982 length frequency data, relative age classes up to 6 years old were represented (Navaluna 1982). In the current data set, the oldest fish in the sample had a relative age, $(t-t_0)$ of 2 years.

Other values for L_{∞} and K were taken from the literature and averages calculated as described above. A combination of L_{∞} of 44.8 cm and $K=0.40 \text{ year}^{-1}$ gave the best fit to the data, after allowing for selectivity. The K-scan indicated that there was a reasonable estimate of K , given L_{∞} . Scanning the data for other fits of L_{∞} and K did not improve the fit so the averages from the literature were used for mortality estimation.

Table 2.10 Results of the length frequency analysis and mortality estimation of *Scomberomorus commerson*.

| Growth Parameters | | Length Converted Catch Curve | | | Beverton and Holt (1956) $L_c = 18$ cm |
|--------------------|----------------------------|------------------------------|----------|----------|---|
| L_∞ (cm) | K (year ⁻¹) | Z (year ⁻¹) | Lower CI | Upper CI | Z (year ⁻¹) |
| 160 | 0.2 | 5.27 | 2.97 | 7.56 | 9.64 |

Table 2.11 Results of the length frequency analysis and mortality estimation of *Otolithes ruber*.

| Growth Parameters | | Length Converted Catch Curve | | | Beverton and Holt (1956) $L_c = 6$ cm |
|--------------------|----------------------------|------------------------------|----------|----------|--|
| L_∞ (cm) | K (year ⁻¹) | Z (year ⁻¹) | Lower CI | Upper CI | Z (year ⁻¹) |
| 44.8 | 0.398 | 4.27 | 3.69 | 4.85 | 4.5 |

Table 2.12 Results of the length frequency analysis and mortality estimation of *Trichiurus haumela*.

| Growth Parameters | | Length Converted Catch Curve | | | Beverton and Holt (1956) $L_c = 37.5$ cm | $Z = M + F$ |
|--------------------|----------------------------|------------------------------|----------|----------|---|----------------------------|
| L_∞ (cm) | K (year ⁻¹) | Z (year ⁻¹) | Lower CI | Upper CI | Z (year ⁻¹) | Z (year ⁻¹) |
| 65.3 | 0.43 | 1.63 | 1.34 | 1.94 | 1.90 | 3.84 (0.84+3) |
| 135 | 0.26 | 4.40 | 3.63 | 5.18 | 4.68 | 3.45 (0.45+3) |

The Length Converted Catch Curve and the Beverton and Holt method were the only two methods which produced credible results for the *O. ruber* data and these are give in Table 2.11. The catch / biomass gave a total Z of over 18 year⁻¹. Whilst the estimates using the other two methods are high, this is too high to accept as a reasonable estimate. The Powell-Wetherall Plot produced unrealistic values for L_{∞} , and it was not possible to determine K from these values. The overall estimate of Z is taken as the average of $Z = 4.27$ year⁻¹ and $Z = 4.5$ year⁻¹. This value is considerably higher than the 1979-1982 total mortality values of 1.89-2.67 year⁻¹ (Navaluna 1982).

Trichiurus haumela

The analysis of the length frequency data of *T. haumela* did not enable discrimination of any one set of parameter values. Growth parameter values for *T. haumela* in the literature are variable. L_{∞} estimates range from 34.4 to 154 cm and estimates of K from 0.2 to 0.7 year⁻¹. However, although variable, the estimates from the literature fell into three categories: those where the L_{∞} was less than the L_{max} of *T. haumela* - these were not used; two sets of data from Manila Bay, the Philippines, with low L_{∞} and high K and six estimates with high L_{∞} and low K from other geographic areas. Their ϕ' values are 3.3 and 3.7 respectively. The average growth parameters for the latter two groups are given in Table 2.12. Essentially, the data suggest that *T. haumela* in Manila Bay are smaller, but faster growing fish than those elsewhere. The mortality analysis was conducted for both sets of parameters and the results are given in Table 2.12.

No results were obtained with the Powell-Wetherall Plot because it was not possible to select any single K-value for a given L_{∞} . However, the Length Converted Catch Curve and the Beverton and Holt method gave similar results for both sets of growth parameters, in each case the Beverton and Holt estimate falling inside the confidence intervals of the Length Converted Catch Curve. The catch / biomass method gave estimates very similar to one another. These results are also similar to the Length Converted Catch Curve and Beverton and Holt results for the slow growing, larger growth parameter set.

The results using the growth parameters which were not from the Philippines produce more consistent results than those from Manila Bay. However, since both Manila Bay and San Miguel Bay are shallow, heavily exploited bays in the same geographical area, the Manila Bay parameters were assumed to be representative of *T. haumela* in San Miguel Bay. The Manila Bay growth parameters produce lower Z estimates. These were checked against available estimates of Z in the literature and were comparable. The mean Z is 2.5 year⁻¹.

Yield per Recruit Analysis

Yield-per-recruit analyses were carried out for the six species above. Where there was more than one set of growth parameters and no conclusion about which was best, generic values were used. The Zs from the Length Converted Catch Curve were used in the analysis since the selection curves were generated from the Length Converted Catch Curves. The results are shown in Figure 2.4.

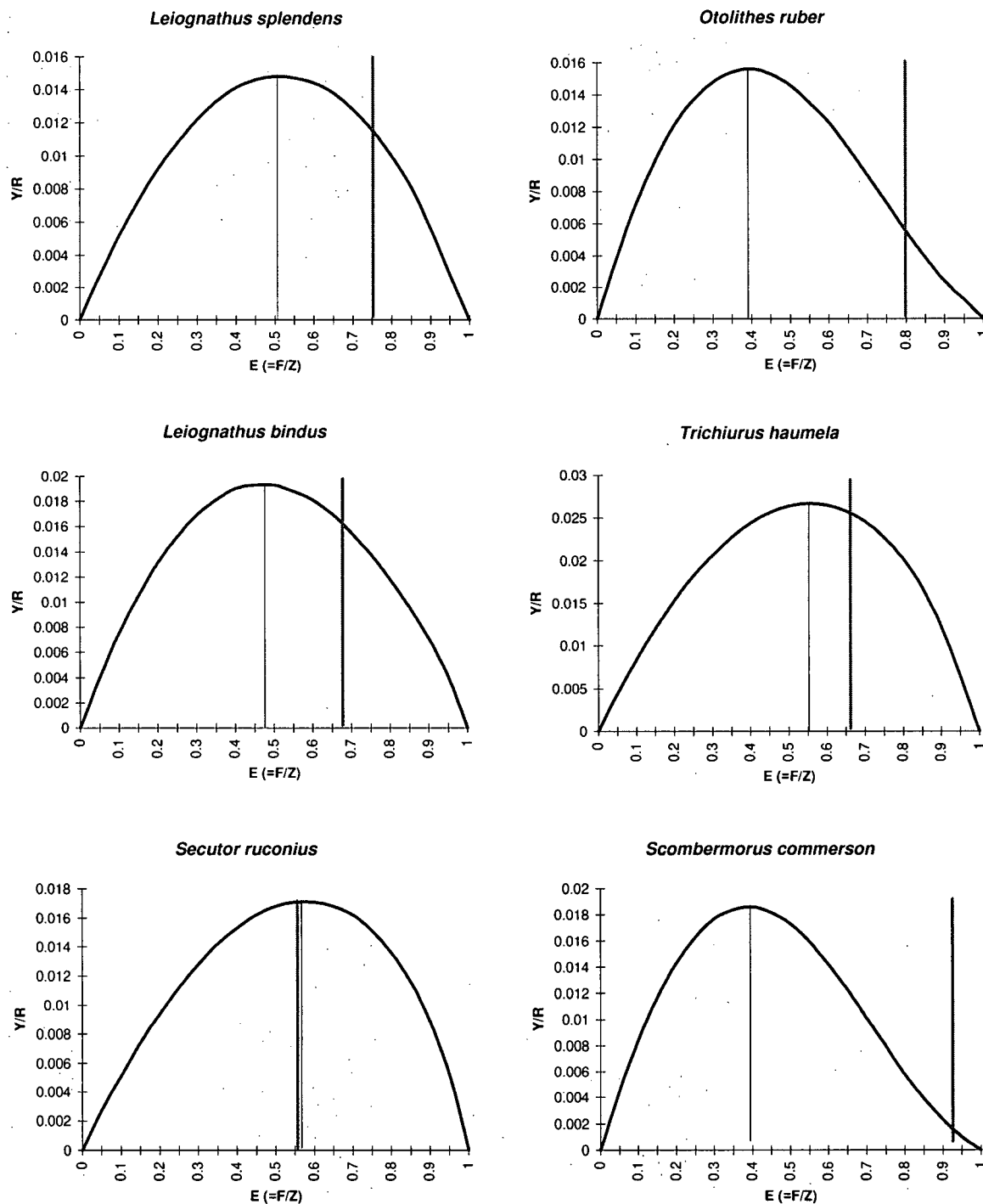


Figure 2.4 Length-based yield-per-recruit curves, showing yield-per-recruit against exploitation. The fine vertical line represents the optimal exploitation rate and the thick line is the current exploitation rate.

The calculated mortality rates above are high and this is reflected in the yield-per-recruit curves and the current state exploitation. The results indicate that five of the six species are overexploited. Optimal exploitation rates are around 0.5 (Gulland 1971) or below 0.5 (Pauly 1984). Actual exploitation rates range from 0.66 (*T. haumela*) to 0.92 (*S. commerson*). The sole species studied which is not overexploited is *Secutor ruconius*, which is currently exploited at the optimal rate. Since these species make up a large part of the catch, this indicates that much of the fishery is overexploited. The exploitation rates of *O. ruber*, *L. splendens*, *L. bindus* and *S. commerson* are all more than 50% greater than the optimal rate. The results confirm the earlier results that there could be great increases in yield if fishing mortality were reduced, particularly for the leiognathids and *O. ruber*, principal components of the trawl survey and catch¹⁰.

Clearly then, fishing is having a large impact on San Miguel Bay. The results of the fishery analysis are now examined in detail.

¹⁰ The conclusions drawn from the mortality and yield-per-recruit analyses are very similar to those of Cinco and Silvestre (1995). They conducted a similar analysis although they managed to obtain results for a total of 15 species, although in less detail than presented here.

Analyses of Fishery Data

Estimating Catch and Effort

Effort

The results of Silvestre *et al.* (1995) are shown in Table 2.13. For each gear type, the number of gears, number of trips per year and total effort are given. The comparative figures for 1979-1982 are also given and are shown graphically in Figure 2.5. Three things are immediately clear:

1. The large, medium and baby trawlers have all substantially decreased, both in absolute numbers and time spent at sea since the early 1980s¹¹;
2. The number of gillnet gears has increased by almost 100%, and the effort per unit has increased;
3. Total effort has increased in the fishery.

Since total effort (in terms of total number of trips) has increased and large-scale effort has decreased, the conclusion must be that there has been a substantial increase in effort by the

¹¹ These results contrast with the findings of Smith and Salon (1987) who conducted a survey of key informants in San Miguel Bay in the mid-1980s. They reported that effort by the medium and baby trawlers had increased by 50% since 1981. They also reported that the number of gillnetters had increased, particularly the number of non-motorised gillnetters.

Table 2.13 List of Fishing Gear in San Miguel Bay and a comparison of Gear Number and Effort in 1979-1982 and 1992-1994.

| Gear Type | Bicol Name | No. of units | | No. of trips per year | | Effort (trips per unit*units) | |
|-----------------------|--------------------------------|--------------|---------|-----------------------|---------|-------------------------------|---------|
| | | 1979/82 | 1992/94 | 1979/82 | 1992/94 | 1979/82 | 1992/94 |
| Large Trawl | — | 30 | 1 | 6 | 8 | 180 | 8 |
| Medium Trawl | — | 17 | 4 | 128 | 57 | 2176 | 228 |
| Baby Trawl | — | 72 | 50 | 128 | 133 | 9216 | 6650 |
| Mini Trawl | <i>Itik-itik</i> | 188 | 260 | 191 | 202 | 35908 | 52520 |
| Shrimp Gillnet | <i>Lait</i> | | 651 | | 120 | | 78120 |
| Surface Gillnet | <i>Palataw</i> | 470 | 99 | 115 | 182 | 54050 | 18018 |
| Bottom-set Gillnet | <i>Palubog/ Patundag</i> | 288 | 50 | 162 | 208 | 46656 | 10400 |
| Shark Gillnet | <i>Pamating/ Pandaracol</i> | 30 | 25 | 94 | 107 | 2820 | 2675 |
| Crab Gillnet | <i>Pangasag</i> | 257 | 343 | 174 | 190 | 44718 | 65170 |
| Ordinary Gillnet | <i>Panke</i> | 300 | 538 | 234 | 247 | 70200 | 132886 |
| Hunting Gillnet | <i>Timbog</i> | — | 288 | — | 220 | — | 63360 |
| Other Gillnet | — | — | 676 | — | 94 | — | 63544 |
| Stationary Lift Net | <i>Bukatot</i> | 171 | 60 | 53 | 115 | 9063 | 6900 |
| Crab Lift Net | <i>Bintol</i> | 71 | 164 | 132 | 192 | 9372 | 31488 |
| Set Longline | <i>Kitang</i> | 103 | 236 | 120 | 156 | 12360 | 36816 |
| Handline | <i>Banwit</i> | 424 | 316 | 120 | 96 | 50880 | 30336 |
| Filter Net | <i>Biacus</i> | 60 | 260 | 225 | 240 | 13500 | 62400 |
| Fish Corral | <i>Baklad/ Sagkad</i> | 89 | 123 | 209 | 106 | 18601 | 13038 |
| Scissor Net | <i>Hud-hud/ Kalicot/ Sakag</i> | 634 | 245 | 150 | 168 | 95100 | 41160 |
| Fish Trap | <i>Bubo</i> | 106 | 225 | 120 | 120 | 12720 | 27000 |
| Ring Net | <i>Kalansisi</i> | — | 2 | — | 120 | — | 240 |
| Pullnet | <i>Bitana /Padbit</i> | — | 3 | — | 224 | — | 672 |
| Fish Weir | <i>Sabay</i> | 5 | | 168 | | 840 | — |
| Stationary Tidal Weir | <i>Ambak</i> | 2 | 1 | 144 | 144 | 288 | 144 |
| Beach Seine | <i>Sinsoro</i> | 11 | 24 | 308 | 308 | 3388 | 7392 |
| Spear Gun | <i>Antipara</i> | 51 | 95 | 156 | 156 | 7956 | 14820 |
| TOTAL | — | 3379 | 4739 | 3137 | 3913 | 499992 | 765985 |

Table compiled from Pauly *et. al* (1982), Silvestre and Cinco (1992) and Silvestre *et. al* (1995).

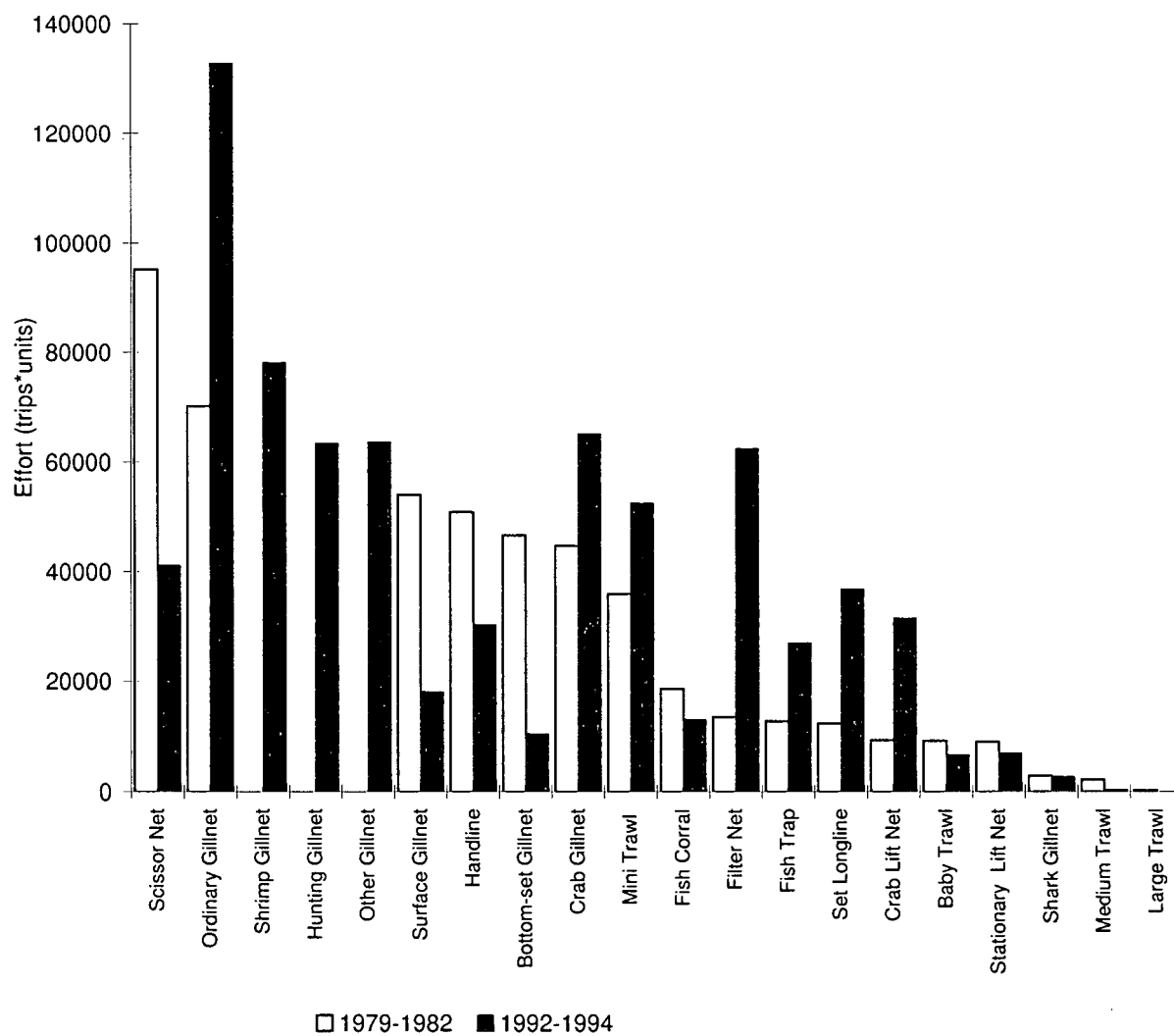


Figure 2.5 Comparison of effort between 1979-1982 and 1992-1994 for 20 types of fishing gear in San Miguel Bay. Effort is estimated from the number of trips made by each gear type per year times the number of units of each gear type (Silvestre *et al.* 1995).

small-scale sector of the fishery. This is seen in the increase in gillnet gears, crab gear, filter nets and “others” gears.

In addition, it is also apparent that there is an increase in the diversity of small-scale gears. In particular, the hunting gillnet (*timbog*) was not listed in the 1979-1982 study, nor were pullnets, ringnets or some of the other gillnets. It is possible that these gears were present earlier, but were subsumed into larger categories. However, the hunting gillnet was not previously described (Pauly *et al.* 1982, Silvestre *et al.* 1995).

Catch

The estimated annual catch for 1992-1994 was 15, 871 tonnes. This is 3, 262 tonnes lower than the 1979-1982 figure of 19,133 tonnes (Pauly and Mines 1982), and 984 tonnes lower than the estimate of 16,855 tonnes by Silvestre *et al.* (1995)¹². These figures indicate that the total catch in San Miguel Bay has decreased since 1979-1982.

The results of the Monte Carlo catch analysis using Crystal Ball are presented in Table 2.14 for 1992-1994 and 1979-1982. Using a 10% uniform input probability distribution for CPUE, vessels and trips, the 95% certainty range for the catch estimate does not include the estimate of 16, 855 (Silvestre *et al.* 1995), or the 1979-1982 mean catch. Similarly, the 95%

¹² The difference between this estimate of 15,871 and that of Silvestre *et al.* (1995) is due to different CPUE values, particularly for the baby trawlers: despite various data manipulations, it was not possible to reconcile these figures.

Table 2.14 Results of the Monte Carlo Simulation of the catch estimate (tonnes) for 1992-1994 and 1979-1982.

| <i>Year</i> | <i>Probability Distribution</i> | <i>95% Certainty Range</i> | | | | |
|------------------|---|----------------------------|--------------------|--------------------|---------------------------|--------------------|
| | | <i>Mean</i> | <i>Lower Limit</i> | <i>Upper Limit</i> | <i>Standard Deviation</i> | <i>Range Width</i> |
| 1992-1994 | 10% uniform probability distribution | 15, 872 | 15, 022 | 16, 806 | 452.1 | 2, 994 |
| | 20% uniform probability distribution CPUE | 15, 836 | 14, 556 | 17, 267 | 691.7 | 4, 408 |
| 1979-1982 | 10% uniform probability distribution | 19, 078 | 17, 660 | 20, 605 | 770 | 5, 086 |
| | 20% uniform probability distribution CPUE | 19, 080 | 17, 040 | 21, 360 | 1, 090 | 7, 186 |

certainty bounds of the 1979-1982 catch estimate (10% uniform distribution for all parameters) does not include the 1992-1994 mean catch estimate. The 95% certainty ranges of the two estimates do not overlap at all.

Greater uncertainty was introduced to the analysis by increasing the variation in the CPUE probability distribution to 20%. This had the effect of widening the range of estimates. In this case, the 95% certainty ranges just overlapped, but as above, the mean catch estimates for the two time periods still lie outwith the other's 95% certainty range.

The Monte Carlo analysis does not reveal how much uncertainty there is in the estimate of catch: it demonstrates the impact that including uncertainty has on the estimate. The uncertainty of the CPUE parameter was increased because it was considered to be the most uncertain parameter.

CPUE was calculated from catch and effort data in the landings survey which took place from July 1992 to June 1993. However, not all of the gears, nor all of the specified landing sites, were systematically covered over the whole of this period. Some months were only represented by one or two samples (see below). However, although increasing the uncertainty of this parameter does increase the range of possible values for the catch estimates, it is clear that there has been a decrease in total catch since the early 1980s (see also Silvestre *et al.* 1995).

The Distribution of Catch, CPUE and Effort Across Gears

The distribution of the catch in the fishery in 1992-1994 and 1979-1982 is shown in Figure 2.6. and the distribution of CPUE is given in Figure 2.7. Despite quite wide 95% confidence limits (using the results from run 2 in Table 2.14), there are very few cases of overlap between the catch in 1992-1994 and 1979-1982. The only gears where there is some overlap are the ordinary gillnet (*panke*), the shark gillnet (*pamating*), the crab lift net (*bintol*) and the lift net (*bukatot*).

Some considerable changes have occurred in the distribution of catch since the early 1980s. For instance, the catch of the trawling sector has decreased for all gears. In the case of the large and medium trawlers, the catch declined dramatically to less than 20% of the 1979-1982 estimate, whilst the catch of the baby trawlers and the mini-trawlers declined to about 50% of their previous value. The decrease in catch of the large and medium trawlers mirrors the decrease in effort seen in Figure 2.5. However, a comparison of CPUE (Figure 2.7) indicates that there was little change in CPUE for the large trawlers and that it increased for the medium trawlers¹³. In the case of the baby trawlers, there was a decrease in catch, in effort and in CPUE. However, despite a large increase in effort by the mini-trawlers, there was a marked decrease in their catch and in CPUE.

¹³ The CPUE estimate is based on one sample per month from July to April, excluding November and January. The estimated catches for July to September were higher than those in the subsequent months by a factor of 10. Since the whole year was not represented in the sample, and one sample was taken per month, both the CPUE figure and the catch figure for the medium trawlers should be treated as quite uncertain. A CPUE estimate based on the samples from October to April produce a CPUE of 615 kg per trip, a figure more in line with that from 1980/81.

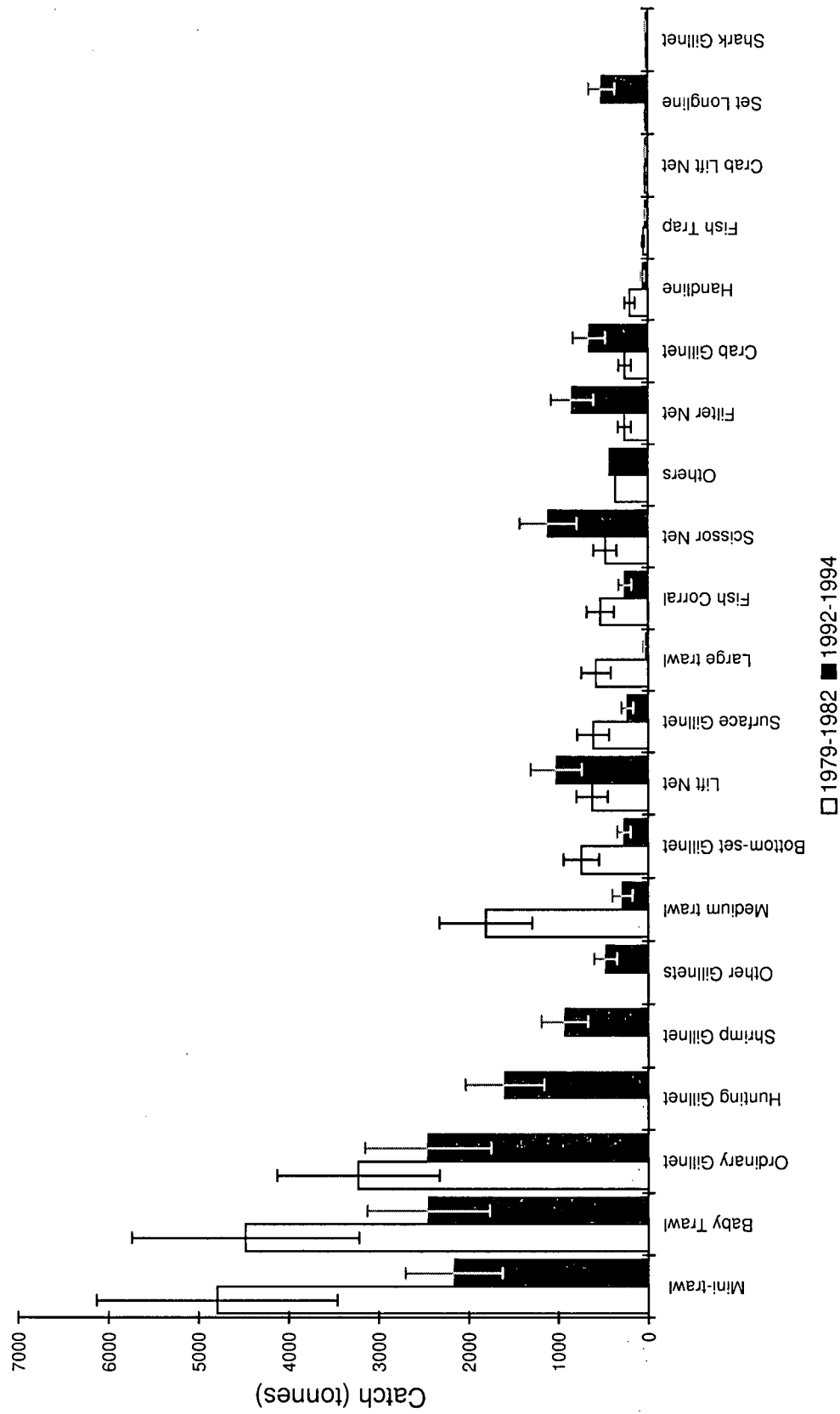


Figure 2.6 Comparison of Catch between 1979-1982 and 1992-1994 for 20 types of fishing gear in San Miguel Bay. Catch is estimated from the CPUE and effort. The error bars represent the 95% confidence limits on the estimates. See text for further details.

The total catch by the gillnet sector increased. This was not due to an increase in the catch of the ordinary gillnet (*panke*), the main type of gillnet used in San Miguel Bay. In 1979-1982 the ordinary gillnet represented 66% of the gillnet catch; in 1992-1994 it represents 37%. There has been an absolute reduction in the catch and CPUE of the ordinary gillnet, despite an increase in overall effort since 1979-1982¹⁴. The increase in total gillnet catch noted is largely due to the use of gears which were not recorded in 1979-1982. For example, there is a large catch by the hunting gillnet (*timbog*), the shrimp gillnet (*lait*) and other gillnets. The catch of the gillnets recorded in 1979-1982 all decreased by 1992-1994, with the exception of the crab gillnet (*pangasag*) and the shark gillnet (*pamating*). The increase in the catch by the crab gillnet is due to an increase in effort and an increase in CPUE. Shark gillnet effort decreased slightly, but an increase in CPUE appears to have enabled an increase in annual catch. The catch of the bottom-set gillnet (*palubog*) and surface gillnet (*palataw*) decreased: however, effort was also severely reduced (Table 2.13) and CPUE in fact increased (Figure 2.7).

There are three main types of fixed gear in San Miguel Bay, the lift net (*bukatot*), the fish corral (*sagkad*) and the filter net (*biacus*). As noted above, the filter net effort has increased since 1979-1982, whilst fish corral and lift net effort have both decreased by about 33%. The catch of the lift net has also increased, but this figure is based on only one sample. CPUE of the filter net and the fish corral both decreased: an increase in effort by the former produced an overall increase in the filter net catch, but the catch of the fish corral decreased.

¹⁴ There is however, an overlap in the 95% certainty ranges for the two time periods.

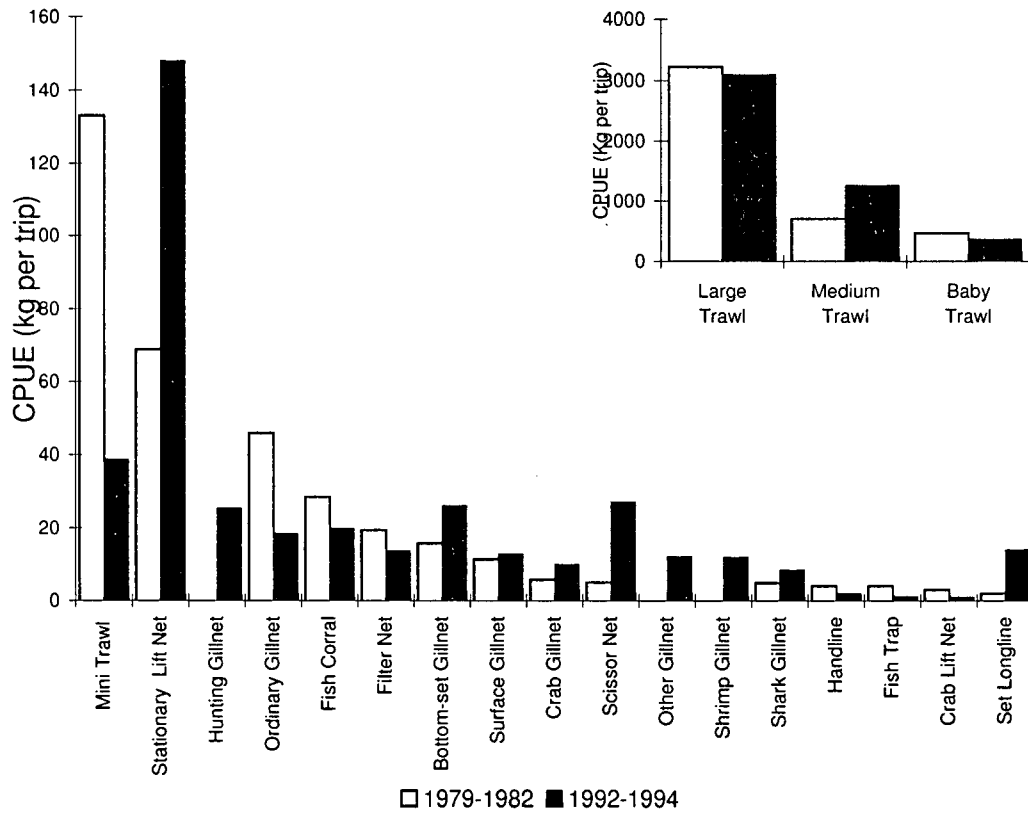


Figure 2.7 Comparison of CPUE between 1979-1982 and 1992-1994 for 20 types of fishing gear in San Miguel Bay. CPUE is estimated from the catch and effort recorded in the landings survey. See text for further details.

The crab lift net (*bintol*) catch remained relatively constant between 1979-1982 and 1992-1994. Effort increased by 300% and CPUE decreased. The catch of the set longline (*kitang*) increased from 25 tonnes in 1979-1982 to 514 tonnes in 1992-1994. Effort increased by some 300% and CPUE increased from 2 kg per trip to 14 kg per trip. At the same time, the catch of the handline (*Banwit*) decreased to 25% of its 1979-1982 value, effort decreased by 40% and CPUE decreased by 50%. The push net (*kalicot, huh-hud*) catch increased by over 100%¹⁵. The fish trap (*bubo*) catch decreased by 40%, the effort increased and the CPUE decreased. However, as with the lift net, these catch and CPUE estimates are based on one sample.

In sum then, the general trend in the distribution of catch in San Miguel Bay, since 1979-1982, has been a decrease in the share of the total catch by the large-scale sector and an increase by the small-scale sector. This is shown more clearly in Table 2.15. The gillnetters almost doubled their share of the catch whilst the large scale sector, or trawlers' share decreased by more than 50%. If the mini-trawlers are included in the large scale sector, the decrease remains as great. With the exception of the mini-trawlers, the catch share of all small scale gears, as grouped below, has increased.

Meanwhile the CPUE, and in many cases effort, of the gears recorded in the 1979-1982 study has decreased, whilst gear diversification appears to have led to increased CPUE for some non-trawl gears in the small-scale sector.

¹⁵ The catch estimate was based on only one sample however.

Table 2.15 Catch distribution by Gear Group in 1979-1982 and 1992-1994.

| <i>Gear</i> | <i>Catch (t)</i> | | <i>% of Catch</i> | |
|---------------|------------------|------------------|-------------------|------------------|
| | <i>1979-1982</i> | <i>1992-1994</i> | <i>1979-1982</i> | <i>1992-1994</i> |
| Trawlers | 6,897 | 2766 | 36.1 | 17.4 |
| Mini Trawlers | 4,779 | 2,165 | 25.0 | 13.6 |
| Gillnets | 4,855 | 6,639 | 25.4 | 41.8 |
| Fixed Gear | 1,416 | 2,125 | 7.4 | 13.4 |
| Line Gear | 228 | 571 | 1.2 | 3.6 |
| Others | 920 | 1,605 | 4.8 | 10.1 |
| Totals | 19,095 | 15,871 | 100 | 100 |

Table 2.16 The ten most abundant species in the Total Catch of San Miguel Bay 1992-1994.

| <i>Species</i> | <i>Catch (t)</i> | <i>% of Catch</i> | <i>Accumulative %</i> |
|--------------------------------|------------------|-------------------|-----------------------|
| <i>Sergestid shrimp</i> | 2736.52 | 17.61 | 17.61 |
| <i>Otolithes ruber</i> | 2349.03 | 15.12 | 32.73 |
| Penaeid Shrimp* | 1466.77 | 9.44 | 42.17 |
| Trash fish** | 991.82 | 6.38 | 48.56 |
| <i>Leiognathus equulus</i> | 925.54 | 5.96 | 54.52 |
| <i>Dendrophysa russelli</i> | 777.14 | 5.00 | 59.52 |
| <i>Portunus pelagicus</i> | 774.47 | 4.98 | 64.50 |
| <i>Trichiurus haumela</i> | 664.60 | 4.28 | 68.78 |
| <i>Stolephorus commersonii</i> | 454.02 | 2.92 | 71.70 |
| <i>Stolephorus indicus</i> | 452.25 | 2.91 | 74.61 |

* The Penaeid species are grouped together, following the procedure used in the 1979-1982 study.

** The Trash Fish are unidentified.

Table 2.17 The ten most abundant families in the Total Catch of San Miguel Bay 1992-1994.

| <i>Family</i> | <i>Catch (t)</i> | <i>% of Catch</i> | <i>Accumulative %</i> |
|------------------|------------------|-------------------|-----------------------|
| Sciaenidae | 3332.7 | 21.5 | 21.5 |
| Sergestid Shrimp | 2736.5 | 17.6 | 39.1 |
| Penaeid Shrimp | 1466.8 | 9.4 | 48.5 |
| Leiognathidae | 1452.5 | 9.3 | 57.9 |
| Engraulidae | 1089.1 | 7.0 | 64.9 |
| Trash Fish* | 991.8 | 6.4 | 71.2 |
| Portunidae | 774.5 | 5.0 | 76.2 |
| Trichiuridae | 664.6 | 4.3 | 80.5 |
| Mugilidae | 574.0 | 3.7 | 84.2 |
| Carangidae | 484.3 | 3.1 | 87.3 |

* The trash fish are unidentified.

Comparative Analysis of Catch Composition

The species composition of the total catch in San Miguel Bay is similar to the species composition of the trawl survey data. Table 2.16 lists the ten most abundant species, which comprise 75% of the total catch and in Table 2.17, the ten most abundant families, comprising 87% of the catch are listed. The sciaenids and the sergestids are the most abundant of all groups in the catch, each accounting for approximately 20% of the total catch. The leiognathids, trichiurids, engraulids and penaeids are again some of the most abundant families.

The main differences between the trawl and catch composition are due to species which are not selected by the trawl gear. The sergestid shrimp have the greatest abundance of any single species represented in the catch. The crab, *Portunus pelagicus*, is the sixth most abundant species. The sergestids however, were not represented in the trawl survey and *P. pelagicus* accounted for less than 1% of the total abundance. There was no "trash fish" category in the trawl survey, since all fish were identified¹⁶.

The catch composition is similar to the 1979-1982 catch composition. The catch figures from 1979-1982 and 1992-1994 are compared in Figure 2.8. The categories used for the 1979-1982 data were used for ease of comparison. The main decreases in catch occur in the sergestids, the engraulids, the clupeids, the mugilids, the squids, the leiognathids and the

¹⁶ There is another curious difference. *Leiognathus bindus* was quite abundant in the trawl survey but only a very minute quantity was recorded in the landings survey. *L. equulus* was very poorly represented in the trawl survey (less than 1% of the leiognathid abundance) and yet is apparently caught in large quantities. These differences may be due to misidentification.

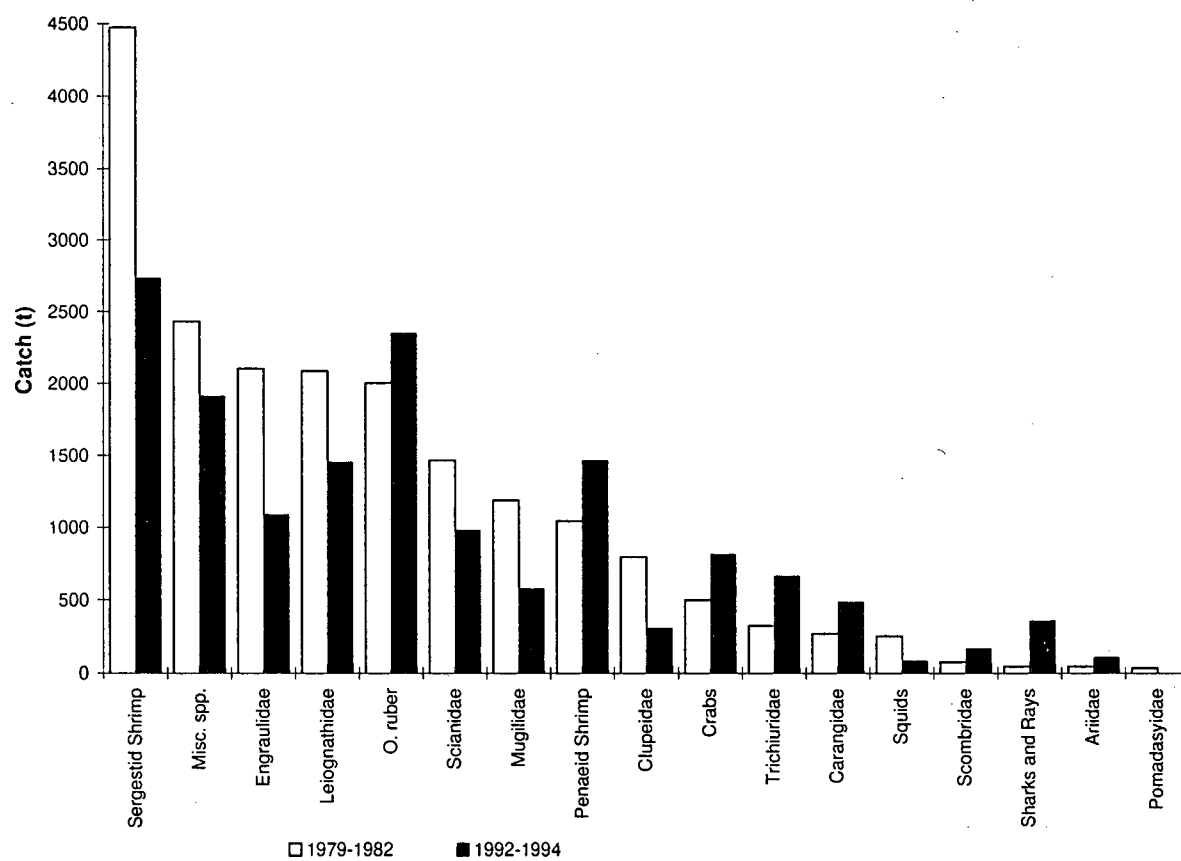


Figure 2.8 Comparison of the catch composition in 1979-1982 and 1992-1994

sciaenids. A few groups such as the crabs, carangids, trichiurids, sharks and rays, and Ariidae increased in abundance.

The miscellaneous/other group in Figure 2.8 decreased by around 20% of the 1979-1982 catch level. However, the miscellaneous/other groups in the 1979-1982 and 1992-1994 data are not necessarily the same. In the 1979-1982 survey, many species were not identified, but simply labeled as "miscellaneous". Pauly (1982a) noted that the large size of the category of miscellaneous species in the 1979-1982 study made it virtually impossible to do species-by-species assessments. He recommended that in future studies a greater attempt should be made to identify the unidentified species. This has been largely achieved in the 1992-1994 study. Because of this, many species appear in the 1992-1994 data that were not recorded in the 1979-1982 data. In Figure 2.8, the 1992-1994 miscellaneous/other group is comprised of species present in 1992-1994 which were not recorded in 1979-1982 plus "trash fish"¹⁷.

Of the species recorded in 1992-1994 which were not recorded in 1979-1982, the greatest proportion consists of demersal feeders such as the Tetraodontidae and Sillaganidae, and predators such as eels and *Lates calcarifer*. Because these species were not recorded in the 1979-1982 data, it is not possible to compare them directly. Some information can be gleaned from the historical trawl survey data, however. This data indicated that the abundance of some groups, such as the Sphyraenidae, the Synodontidae and the Mullidae has decreased since 1947 (Table 2.6). The abundance of others, such as the Sillaginidae, the eels and the Tetraodontidae may have increased. The latter three groups now constitute a significant

¹⁷ Trash fish include four categories: assorted fish, mixed species, fish meal and trash fish. All other species were identified in the survey, at least to the generic level.

proportion of the total catch with landings of 200 t, 144 t and 84 t respectively. These groups may thus have increased. Alternatively, the increase in catch of these groups could be explained by the fact that these species are now sought and marketed, whereas they were not previously.

Species Composition and Distribution of the Catch by Fishing Gear and Season

It was noted above that the large-scale share of the total catch has decreased since 1979-1982. On a species or group by group basis, the same story is told. Table 2.18 shows the distribution of catch between the large and small-scale sectors. Regardless of whether the total catch for each species or group in the table increased or decreased, the small-scale sector share increased, with one exception. Only the catch of the carangids, which increased in both sectors, increased proportionately more in the large-scale sector. The small-scale catch of the penaeids increased by 83%, the leiognathids to 66%, *Otolithes ruber* to 98%, the other sciaenids to 90%, the Scombridae to 78% and the sharks and rays to 74%. The small-scale share of the Miscellaneous/other species also increased.

The Large-Scale Sector

The large-scale sector lands 17.4% of the total catch and is less selective than most of the small-scale gears. The medium trawl and baby trawl catch around 70 species of fish, elasmobranchs and crustaceans. Of these, 6% and 19% respectively are trash fish.

Table 2.18 Distribution of species between the large-scale Sector and the Small-Scale Sector.

| Taxon | Catch (t) | | Catch (t) | | % | | % | |
|------------------|-----------|-------|-----------|--------|---------|-------|---------|-------|
| | 1979/82 | | 1992/94 | | 1979/82 | | 1992/94 | |
| | LS | SS | LS | SS | LS | SS | LS | SS |
| Sergestid Shrimp | 0 | 4473 | 0 | 2736.5 | 0 | 100.0 | 0 | 100.0 |
| Penaeid Shrimp | 462 | 582 | 248.9 | 1217.9 | 44.2 | 55.8 | 17.0 | 83.0 |
| Crabs | 120 | 380 | 55.1 | 758.7 | 24.0 | 76.0 | 6.8 | 93.2 |
| Squids | 235 | 15 | 57.3 | 21.8 | 93.9 | 6.1 | 72.5 | 27.5 |
| Clupeidae | 202 | 593 | 62.7 | 242.4 | 25.3 | 74.7 | 20.6 | 79.4 |
| Engraulidae | 1369 | 731 | 472.9 | 616.3 | 65.2 | 34.8 | 43.4 | 56.6 |
| Carangidae | 57 | 212 | 170.6 | 313.7 | 21.3 | 78.6 | 35.2 | 64.8 |
| Leiognathidae | 2013 | 74 | 490.8 | 961.7 | 96.4 | 3.6 | 33.8 | 66.2 |
| Mugilidae | 329 | 860 | 97.5 | 476.5 | 27.7 | 72.3 | 17.0 | 83.0 |
| <i>O. ruber</i> | 410 | 1594 | 50.9 | 2298.2 | 20.4 | 79.6 | 2.2 | 97.8 |
| Sciaenidae | 313 | 1155 | 100.1 | 883.5 | 21.3 | 78.7 | 10.2 | 89.8 |
| Trichiuridae | 254 | 70 | 181.9 | 482.7 | 78.5 | 21.5 | 27.4 | 72.6 |
| Ariidae | 6 | 39 | 8.4 | 99.0 | 13.2 | 88.0 | 7.9 | 92.1 |
| Pomadasyidae | 21 | 13 | 0 | 1.3 | 61.5 | 38.5 | 0 | 100.0 |
| Scombridae | 28 | 47 | 34.9 | 129.0 | 37.9 | 62.1 | 21.3 | 78.7 |
| Sharks and Rays | 36 | 9 | 92.1 | 263.6 | 79.9 | 20.1 | 25.9 | 74.1 |
| Misc. spp. | 1041 | 1389 | 742.9 | 1166.4 | 42.9 | 57.1 | 38.9 | 61.1 |
| SUM | 6897 | 12138 | 2867 | 12669 | 36.1 | 63.4 | 18.5 | 81.5 |

LS = Large-Scale Sector (Large, Medium and Baby Trawls plus the Ring Net)

SS = Small-Scale Gears(all other gears)

Collectively, this is about the same as the amount of trash fish landed by the entire small-scale sector, but in the small-scale sector, the trash fish only account for 4% of their total catch. The baby trawl landing of trash fish is also an increase from 1979-1982 when 10% of landings were trash fish.

The top ten species/groups landed by the trawlers account for 80%-85 % of the catch and include many of the families listed in Table 2.17. As in the 1979-1982 survey, the main groups caught are trash fish, leiognathids and engraulids. Figure 2.9 compares the CPUE between 1979-1982 and 1992-1994 for the baby trawlers. The top ten species/groups from the 1992-1994 data are shown plus any species/groups which were important in the 1979-1982 data. The CPUE of the leiognathids, engraulids, sciaenids, mugilids, clupeids and squid all decreased. Only the trash fish, the trichiurids, the carangids and sharks and rays increased. There was little change in the CPUE of the penaeids and the crab, *Portunus pelagicus*.

The trawl gears operate throughout the year. There is seasonal variation in the catch composition and the CPUE. The leiognathids, engraulids, carangids and clupeids are most abundant in the trawl catch during July, August and September, the period of the Southwest monsoon. During these months, the total CPUE of the baby trawler is an order of magnitude higher than during the rest of the year¹⁸. The greatest diversity in the catch of the baby trawler was from December to February, although this could be due to a higher number of samples. During these months, the trash fish make up the greatest component of the catch (22%-41%).

¹⁸ Months included in the landings survey for the baby trawler were: August, September, October, December (1992) and January, February, March and June (1993). Landings data for July to October and December (1992), February, March and April (1993) were collected for the medium trawl, and July and August 1992 for the large trawl.

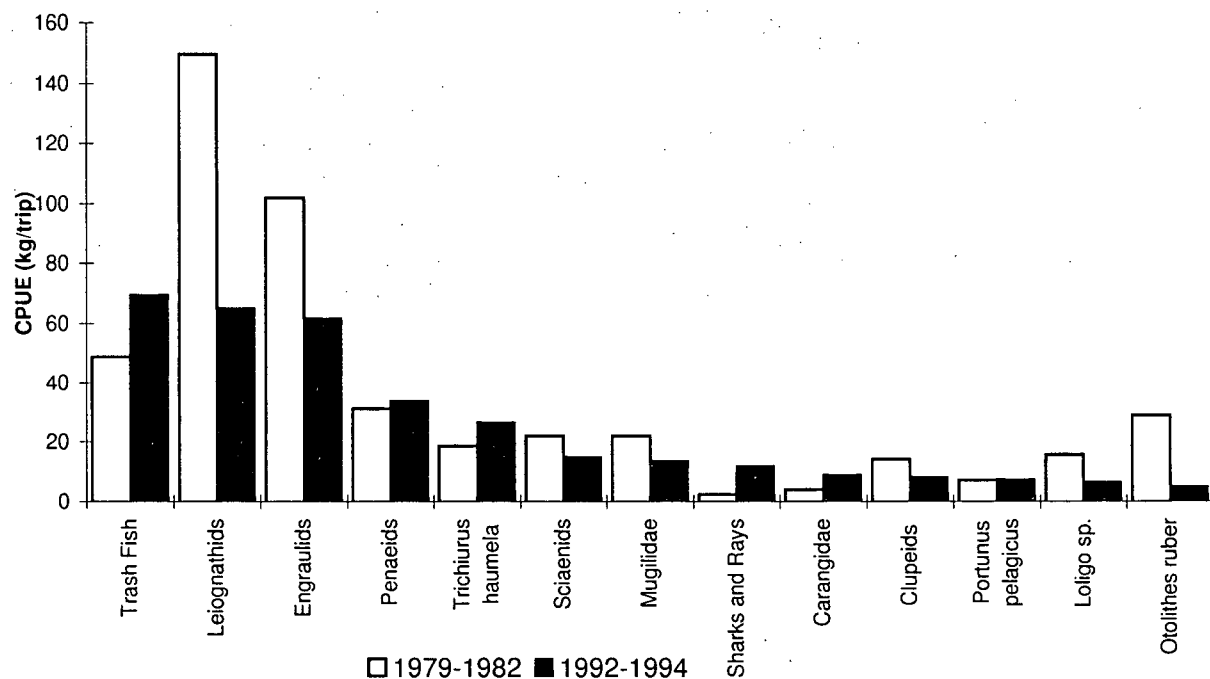


Figure 2.9 CPUE of the top species and groups in the Baby Trawl catch from 1979-1982 and 1992-1994. The sharks and rays are combined following the procedure used in the 1979-1982 data.

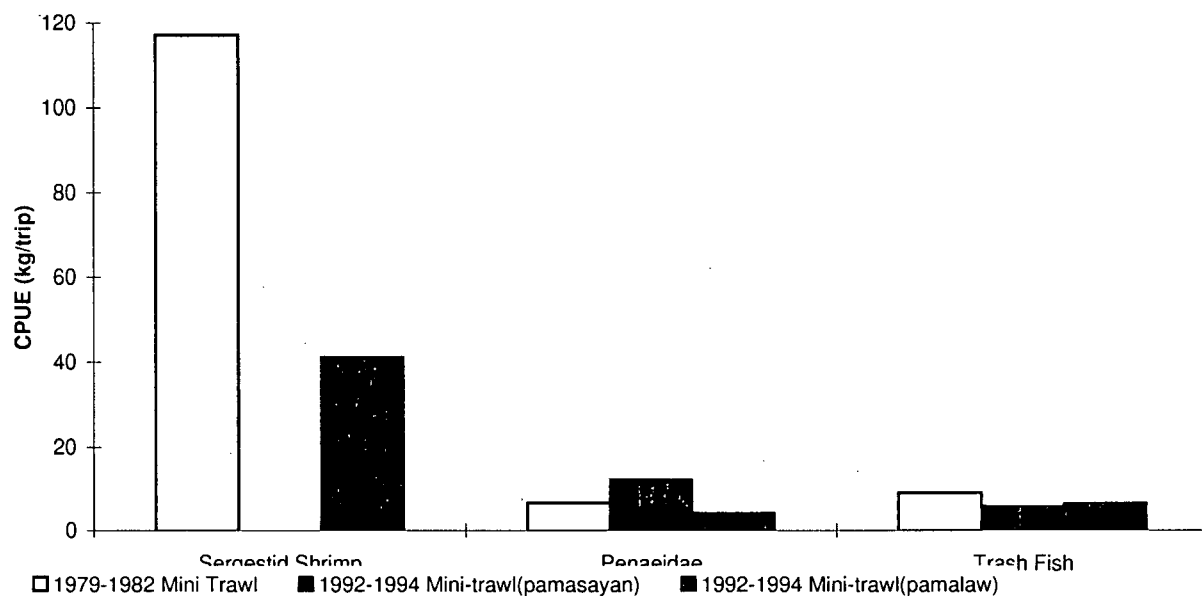


Figure 2.10 CPUE of the top species and groups in the Mini Trawl catch from 1979-1982 and 1992-1994. The two types of Mini Trawler, the Pamalaw, which targets Sergestid Shrimps and the Pamasayan, which targets Penaeid Shrimps are shown for the 1992-1994 data.

The medium trawl catch was also more diverse during these months, although the sample size did not change.

The Mini-Trawlers

The mini-trawlers land mainly sergestid and penaeid shrimps and they account for almost 14% of the total catch. There is a strong seasonal aspect to the operation of this gear. Two different operations of the mini-trawler exist, the *pamalaw* which targets the sergestid shrimp from October to May (during the months of the Northeast Monsoon) and the *pamasayan* which targets penaeid shrimp during April to October¹⁹. Penaeids comprise 52% of the catch of the *pamasayan*, 25% of the catch are trash fish, and 67 species are caught in total. The *pamalaw* is a little more selective. It catches 45 species in total, 77% of which are sergestid shrimp and 12%, trash fish.

Figure 2.10 compares the CPUE in 1979-1982 and 1992-1994. The data for 1979-1982 are collated, while the data from 1992-1994 are compared separately. The total CPUE of the mini trawl decreased in 1992-1994 (Figure 2.7) and the CPUE of the sergestids and the trash fish decreased too. The CPUE of the penaeids increased from the 1979-1982 value when compared to the *pamalaw*, which targets the penaeids, and decreased for the *pamasayan*.

¹⁹ The mini-trawler *pamalaw* and *pamasayan* differ in the material and mesh size used in the trawl net (Tulay and Smith 1982). It appears that the catch and effort of the two forms were combined in the 1979-1982 survey. For comparative purposes, their catch and effort are combined here too. The effort statistics (Sylvestre *et al.* 1995) refer only to the mini-trawlers collectively. Their effort was differentiated here on the basis of the number of months of operation. The *pamalaw* is the mini-trawler main mode of operation for 7/12 months and the *pamasayan* the main mode of operation for 5/12 months.

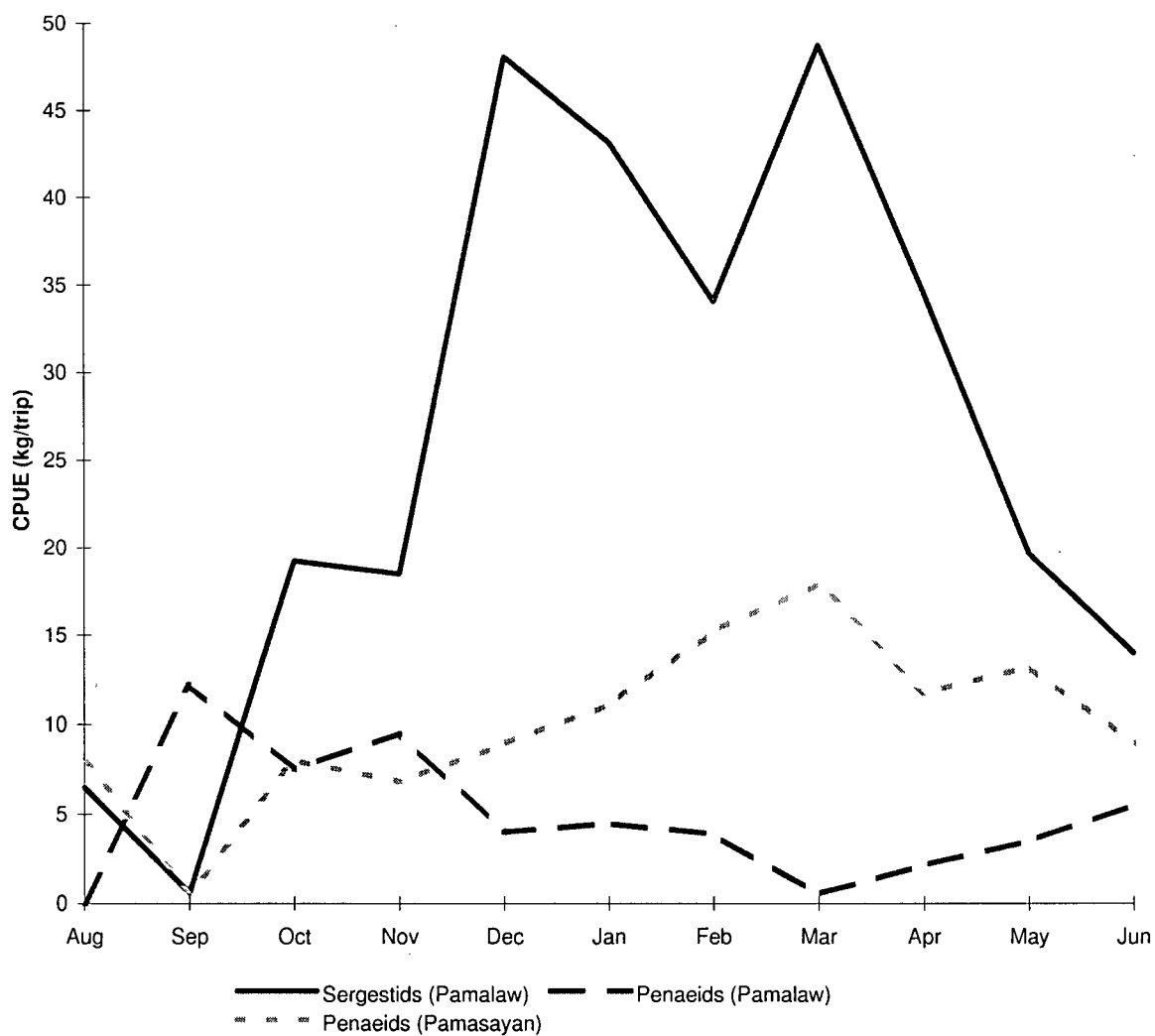


Figure 2.11 CPUE per month for the selected species in the Mini Trawler catch of 1992-1994.

Both the mini-trawler *pamalaw* and *pamasayan* were sampled from August 1992 to June 1993. The CPUE from these data confirm that the mini trawl *pamalaw* operate during the period of the year when the sergestid CPUE is highest (Figure 2.11), that is, during the Northeast Monsoon. However, the 1979-1982 CPUE value of 117 kg/trip was not achieved in any month in 1992-1994. The other species/groups caught by this gear are incidental and their CPUE is lower from October to May than during the rest of the year. The CPUE of the penaeids is more variable. It peaks in the mini-trawler *pamasayan* catch in March and in the *pamalaw* catch in September. There is no clear indication that the CPUE or abundance of the penaeids is higher during April to October, the period when they are targeted by the mini-trawler *pamasayan*.

The Gillnets

In terms of magnitude of catch and effort, gillnets are the dominant sector of the fishery. They are diverse in their operation and land 42% of the total catch. Table 2.13 lists eight types of gillnet, but the actual total is higher, since some are combined. Between them all, they catch virtually all species in the catch and operate all year.

The ordinary gillnet, the hunting gillnet and the shrimp gillnet all catch sciaenids, engraulids and penaeids, identified in the list of the top ten species/groups in the total San Miguel Bay catch. For each gear, *Otolithes ruber* is the most abundant in the catch. The shrimp gillnet is the most selective. Only six species are recorded in its catch. The ordinary gillnet catches 93

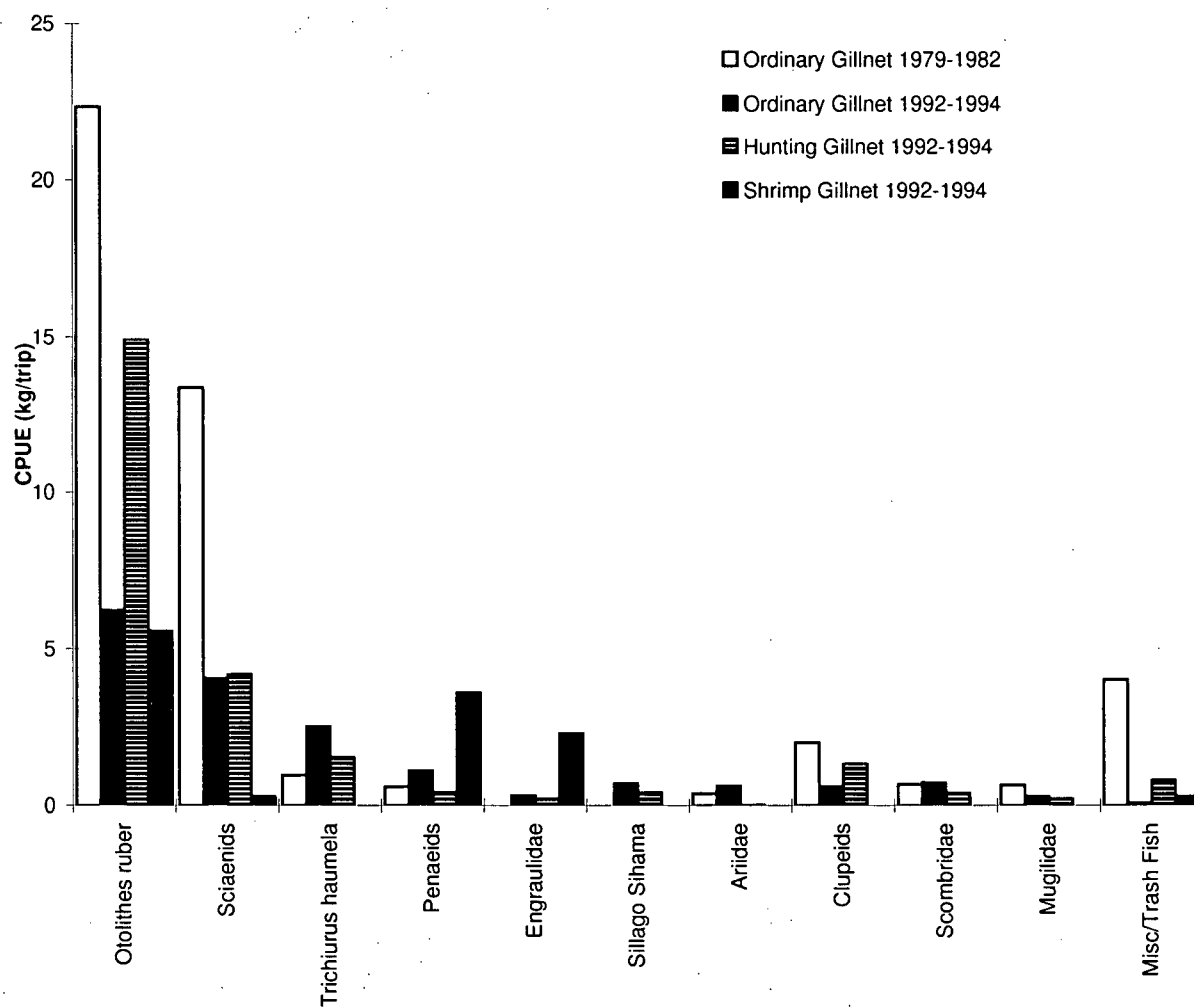


Figure 2.12 CPUE of the top species and groups in the Ordinary Gillnet catch from 1979-1982 and 1992-1994. Also shown are the Hunting Gillnet and Shrimp Gillnet CPUE for 1992-1994.

species and the hunting gillnet 68 species. This highly diverse catch is comparable to the trawler catch. The top ten species/groups in the ordinary gillnet catch account for 94% of the catch and the top ten species in the hunting gillnet catch account for 97%. The CPUE of these top ten species/groups are shown in Figure 2.12, together with the shrimp gillnet data. They are compared to the 1979-1982 CPUE for the ordinary gillnet.

It was noted above that the CPUE of the ordinary gillnet has decreased since 1979-1982. This was due to a decrease in the CPUE of the sciaenids, the clupeids and the mugilids. Some groups increased, such as the trichiurids, the Ariidae and the penaeids. Others, for example, *Sillago sihama* were not recorded in the 1979-1982 data, but may have been included in the miscellaneous group.

The ordinary gillnet is used throughout the year, though effort is higher during the months of October to March. Figure 2.13 shows the total monthly CPUE and the monthly CPUE of the sciaenids and *Trichiurus haumela*. The CPUE of the sciaenids is relatively stable, especially from December to April. There are two peaks in the total monthly CPUE. One is caused by the sciaenids in October and the other by a large increase in the CPUE of *Trichiurus haumela* in March²⁰. Without the latter, the monthly CPUE of the ordinary gillnet would be relatively flat. In almost every month, the sciaenids are the most abundant species in the catch. However, the average sciaenid CPUE of 1979-1982 was 36 kg/trip and this value is not reached during any month in 1992-1994 data. The highest sciaenid CPUE is 15 kg/trip in

²⁰ This data comes from the landings from 10 vessels, mostly taken towards the end of March. The catch of two vessels, recorded on the same day, accounts for 83% of the March catch.

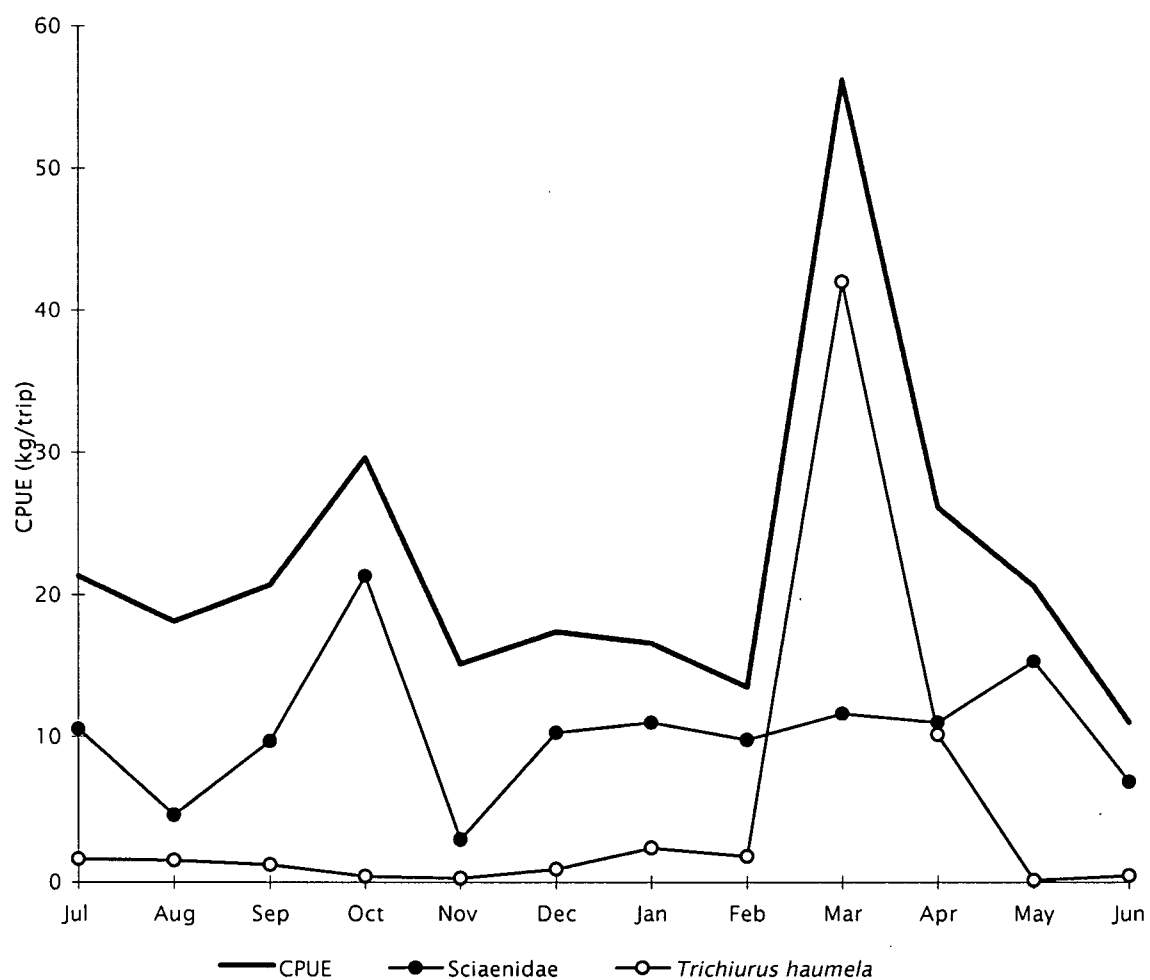


Figure 2.13 CPUE per month for selected species and groups in the Ordinary Gillnet catch of 1992-1994.

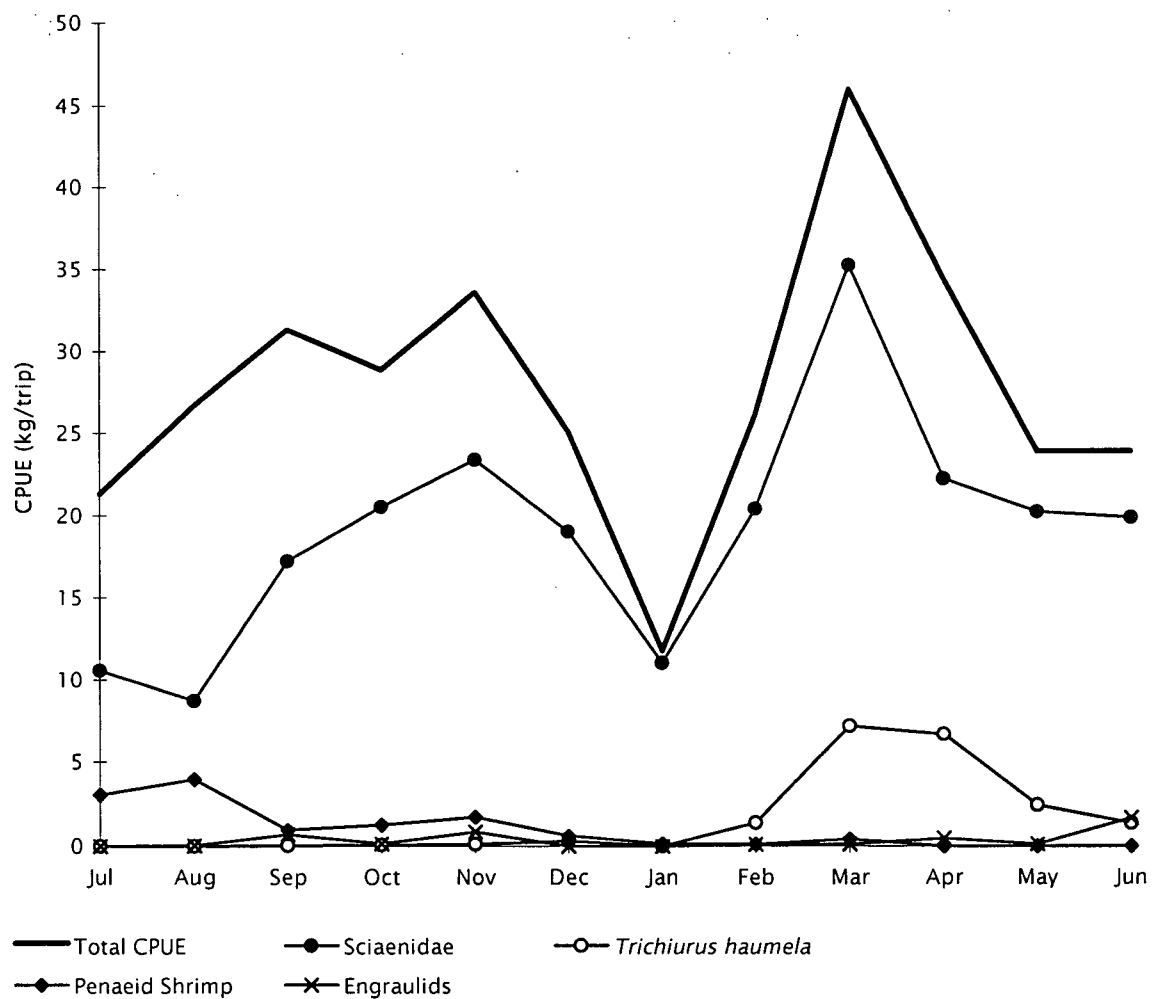


Figure 2.14 CPUE per month for selected species and groups in the Hunting Gillnet catch of 1992-1994.

May. Indeed the total CPUE exceeded the 1979-1982 sciaenid CPUE value only once, due to the large catch of *Trichiurus haumela* in March.

The hunting gillnet also operates all year round. The CPUE of the sciaenids is both higher and more variable than for the ordinary gillnet (Figure 2.14). Here the highest CPUE is reached in March, almost reaching the 1979-1982 average value²¹. The CPUE of *Trichiurus haumela* increases during March and April. Otherwise the total CPUE is shaped by the sciaenid CPUE. It is possible that the hunting gillnet was adapted in response to declining catches of the sciaenids.

The main use of the shrimp gillnet occurs from December to March, although 40-50% of respondents in the Fishing Inventory also use the gear during the other months. The landings of the shrimp gillnet were sampled during December, February, March and May²². Although described as a shrimp gillnet, in December and February over 70% of the catch were sciaenids, mostly *Otolithes ruber*. In addition, the total CPUE of the sciaenids is almost twice that of the penaeids. The CPUE of the sciaenids declined from December to May, while the CPUE of the penaeids was between 2-3 kg/trip for all months sampled except March when it peaked to 10 kg/trip. The total CPUE of the penaeids and the engraulids is higher than for the ordinary gillnet or the hunting gillnet (Figure 2.12). This is because of the high CPUEs recorded in the March sample.

²¹ However, this value came from one sample only.

²² Sampling was rather sparse, with four samples in December and one sample in the other months.

The surface gillnet and the bottom-set gillnet have surprisingly similar catch compositions. Both gears catch 67 species in total. The mugilids account for 76% of the surface gillnet catch and 67% of the bottom-set gillnet catch. The top ten species/groups in the catch account for 97.5% and 99.5% respectively. Other species in the top ten caught by both gears include *Trichiurus haumela*, engraulids, clupeids, penaeids, sciaenids, and trash fish. The CPUEs of these gears are compared to the 1979-1982 values in Figure 2.15. Although the catch and effort of these gears have decreased, the CPUE of the mugilids increased by around 40% for both. The bottom-set gillnet mugilids CPUE is twice the CPUE of the mugilids caught by the surface gillnet. A greater diversity of species is recorded by both gears in 1992-1994 than in 1979-1982.

The surface gillnet and bottom-set gillnet are basically used year round, although effort by the bottom-set gillnet is concentrated during the months of May to September. The landings were only sampled over 6 months though, and the bottom-set gillnet was not sampled from July to November, part of the period of its main use. There is a decline in the total CPUE of the surface gillnet from December to May, and an increase in the CPUE of the rays from March to May. The CPUE of the other species/groups in the catch is variable with no clear pattern. The monthly CPUE pattern of the bottom-set gillnet is dominated by a large peak in the mugilid catch in April²³. Unfortunately there is no data for March, and the CPUE falls rapidly again in May. There is thus considerable variability in the CPUE of the mugilids caught by the bottom-set gillnet. During January and February, the engraulids and the sciaenids comprise the greatest part of the catch. But this is the "off-season" for the bottom-set gillnets.

²³ Four vessels, on four separate days spanning the whole month, landed the catch which caused this large CPUE.

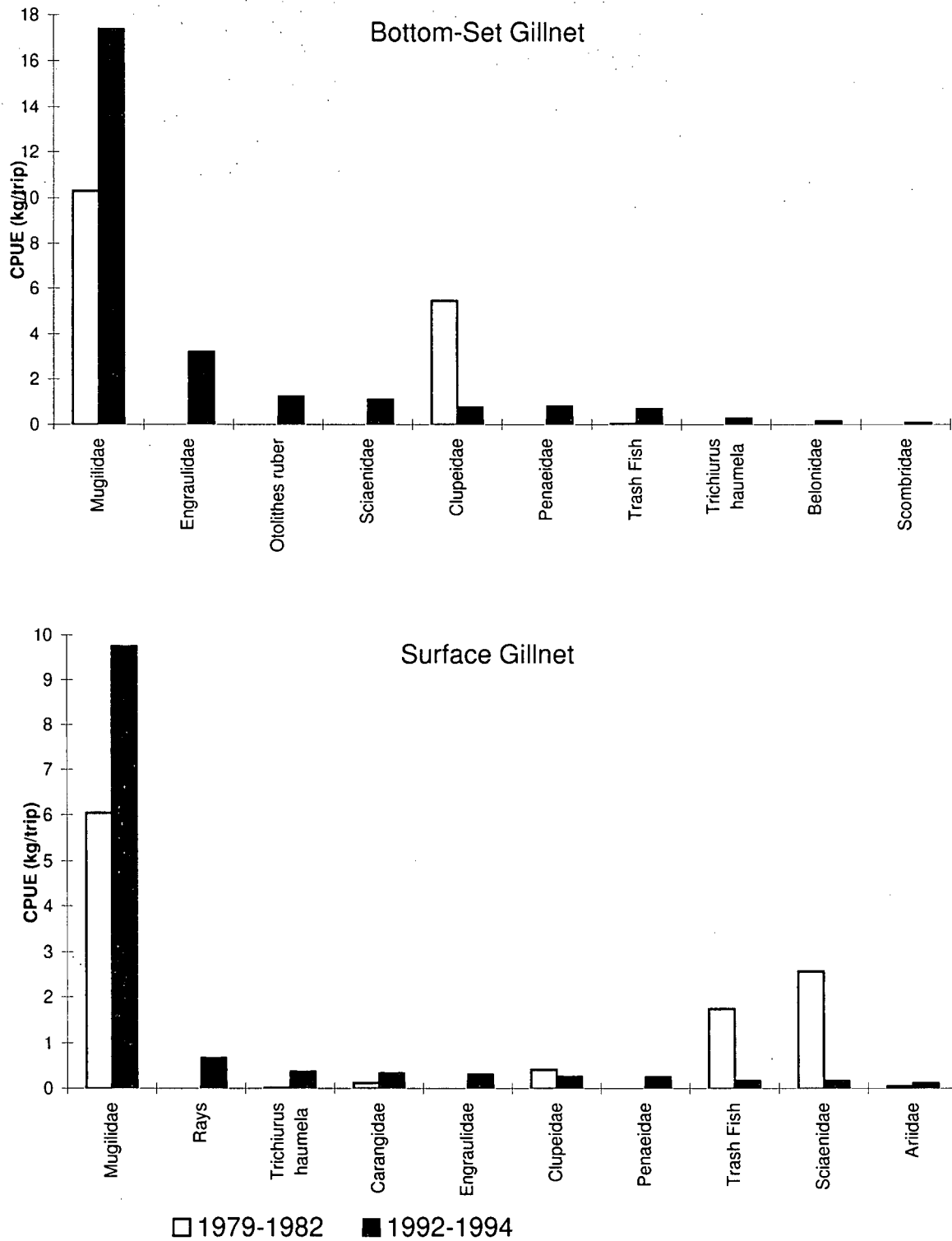


Figure 2.15 CPUE of the top species and groups in (a) the Bottom-Set Gillnet catch and (b) Surface Gillnet catch from 1979-1982 and 1992-1994.

The catch and CPUE of the last two gillnets to be discussed, the crab gillnet and the shark gillnet increased from 1979-1982 to 1992-1994. For the crab gillnet, this was due to an almost 100% increase in the CPUE of the portunid crab, *Portunus pelagicus*, which accounts for 96.5% of the total catch²⁴. Although some crab gillnets are in operation all year, most operate from May to October-December. Unfortunately, the 5 months during which this gear were sampled were February to June and so only 2 months of its main operation are covered. This does provide contrast however. From February to May the CPUE ranges from 0.3 kg/trip to 8.6 kg/trip. The June CPUE is 43.3 kg/trip, taken from an average of 29 samples²⁵. The catch, effort and CPUE of this gear have increased since 1979-1982, indicating that this has been one of the few areas of expansion in the fishery.

The shark gillnet catch is composed of the centropomid *Lates calcarifer* (50%), the shark, *Carcharhinus melanopterus* (24%), the ehippid, *Drepane punctata* (17%) and the catfish, *Arius leitocephalus* (3%). The effort of this gear has decreased slightly since 1979-1982, but the catch and CPUE increased. This gear was sampled from December to May, excluding April and there are no clear trends in the CPUE. There is not much basis for comparison with the 1979-1982 species data either. The CPUE of the shark group and the Ariidae decreased, but *L. calcarifer* and *D. punctata* were not recorded in the 1979-1982 data.

²⁴ In the 1979-1982 data, the crabs were not differentiated. Pauly (1982a) notes that the San Miguel Bay crabs were named *Neptunus pelagicus* (= *Portunus pelagicus*) by Umali (1937). It is assumed here that the crabs in the 1979-1982 catch are predominantly *P. pelagicus*.

²⁵ The implication is however, that the average annual CPUE of the crab gillnet would be higher if it were calculated from 12 months of CPUE data instead of the 5 months of data presented here. Consequently, the crab gillnet catch may be higher than presented in Figure 2.6. This case could also be made for several other gears. The confidence limits in Figure 2.6 are intended to represent this uncertainty.

The Fixed Gears

The fixed gears, the filter net, fish corral and lift net, land 13.4% of the total catch (Table 2.15). They all operate throughout the year, although twice the number of filter nets are used during January and February than in the other months. The filter net is a fine-meshed gear (1-2 cm, Silvestre *et al.* 1995) and 42.3% of its catch is the sergestid shrimp. The engraulids, penaeids and trash fish account for another 45.5% of the catch. Thirty three species/groups are recorded in the catch in total. The top ten species/groups in the filter net catch include eight of the top ten species/groups in the total catch of San Miguel Bay. In Figure 2.16, the CPUE for the top ten species/groups is compared to the 1979-1982 CPUE values. Most have decreased, especially the engraulids and the leiognathids. The exceptions are the sergestid shrimp and the penaeid shrimp, which increased. Note that when the sergestid CPUE was examined for the mini-trawler above, the sergestid CPUE had decreased.

The total monthly CPUE is shaped largely by the sergestid CPUE (Figure 2.17). The CPUE of the sergestids fluctuated between 5 and 10 kg/trip from December to May. The CPUE of the engraulids peak in December and then decrease to zero through March, April and May. In December and February, the total CPUE is higher than the 1979-1982 average CPUE. In November, it is much lower.

The catch of the fish corral is more diverse than the filter net. However, the fish corral catch, effort and CPUE decreased between 1979-1982 and 1992-1994. There are 86 species recorded in the catch and the top ten species/groups account for 80%. When compared to

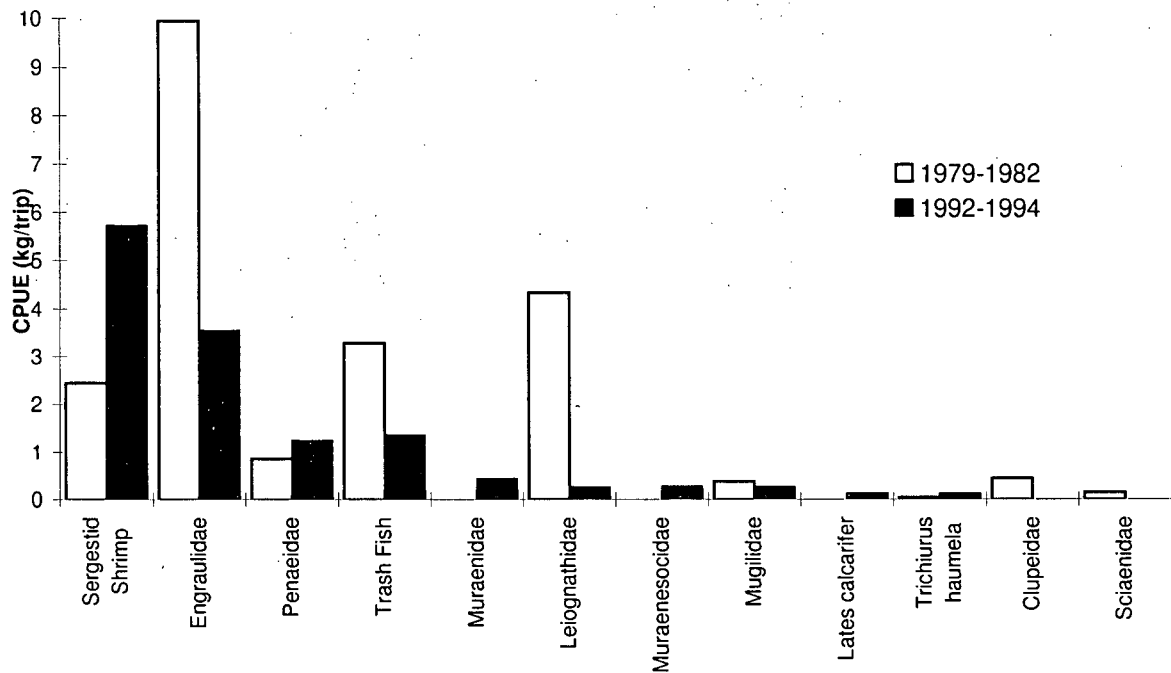


Figure 2.16 CPUE of the top species/groups in the Filter Net catch from 1979-1982 and 1992-1994.

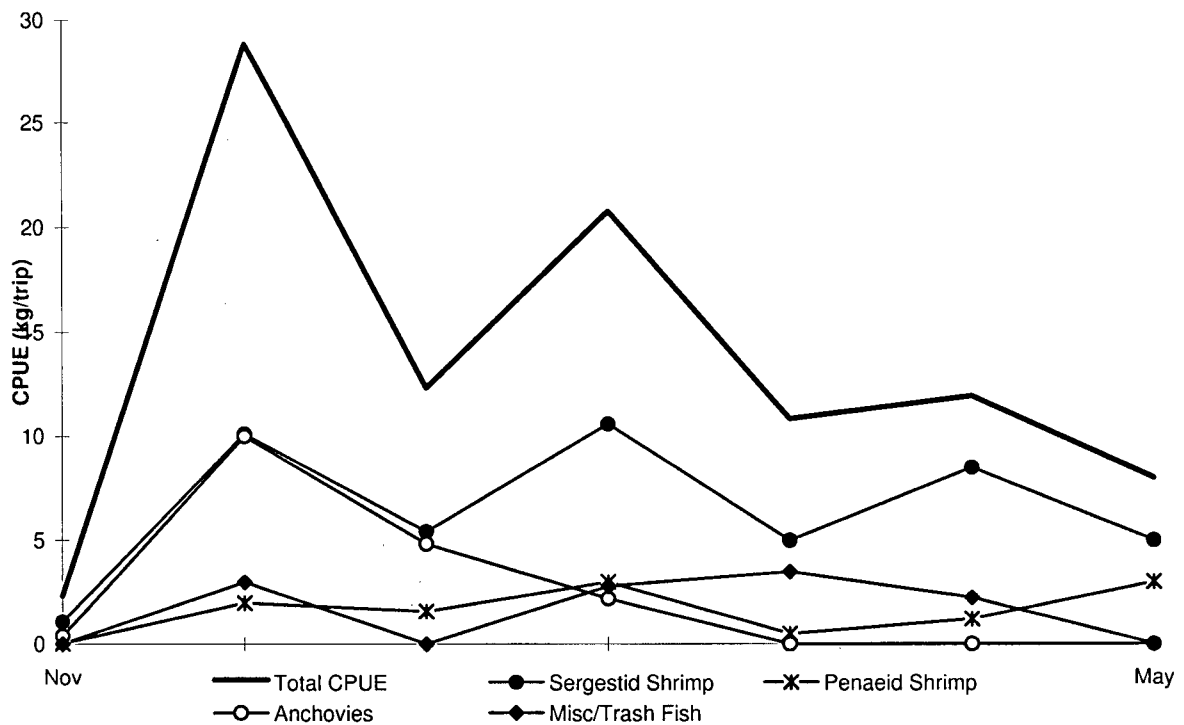


Figure 2.17 CPUE per month for selected species/groups in the Filter Net catch of 1992-1994.

the CPUE of the 1979-1982 catch in Figure 2.18, the most notable change is in the trash fish, which decrease from 12 kg/trip to 1.4 kg/trip. This decrease and the decrease in the crab, clupeid and sciaenid CPUE account for the large decrease in total CPUE since 1979-1982. The CPUE of the carangids, penaeids, engraulids, leiognathids, *Trichiurus haumela* and mugilids increased.

There is considerable variability in the monthly CPUE of the fish corral (Figure 2.19). About 28 fish corral fishers in the Fishing Gear Inventory use their gear all year around. However, around three times this number are in use in any one month. The two peaks in total CPUE occur in November and June, but there are no data for the months between June and November²⁶. The pattern of gear usage does not reflect these two peaks. The peaks are caused by the carangids, leiognathids and sciaenids in November and the carangids and penaeids in June. The sciaenids and leiognathids peak again between March and May and the CPUE of all species/groups is variable.

The lift net is also used all year around by almost all operators. The effort of this gear decreased considerably since 1979-1982. Unfortunately, only one landing sample was taken for the lift net during the landing survey. Its composition contrasts sharply with the 1979-1982 catch composition. In 1979-1982, 80% of the catch were engraulids and 8% clupeids. In 1992-1994, 88% of the catch are leiognathids, 7% engraulids and 5% clupeids. Given the seasonality of the catch seen in the gears above, little confidence can be placed in the 1992-

²⁶ The sample size for October to February was 1, 2 in April, 3 in March and May and 4 in June.

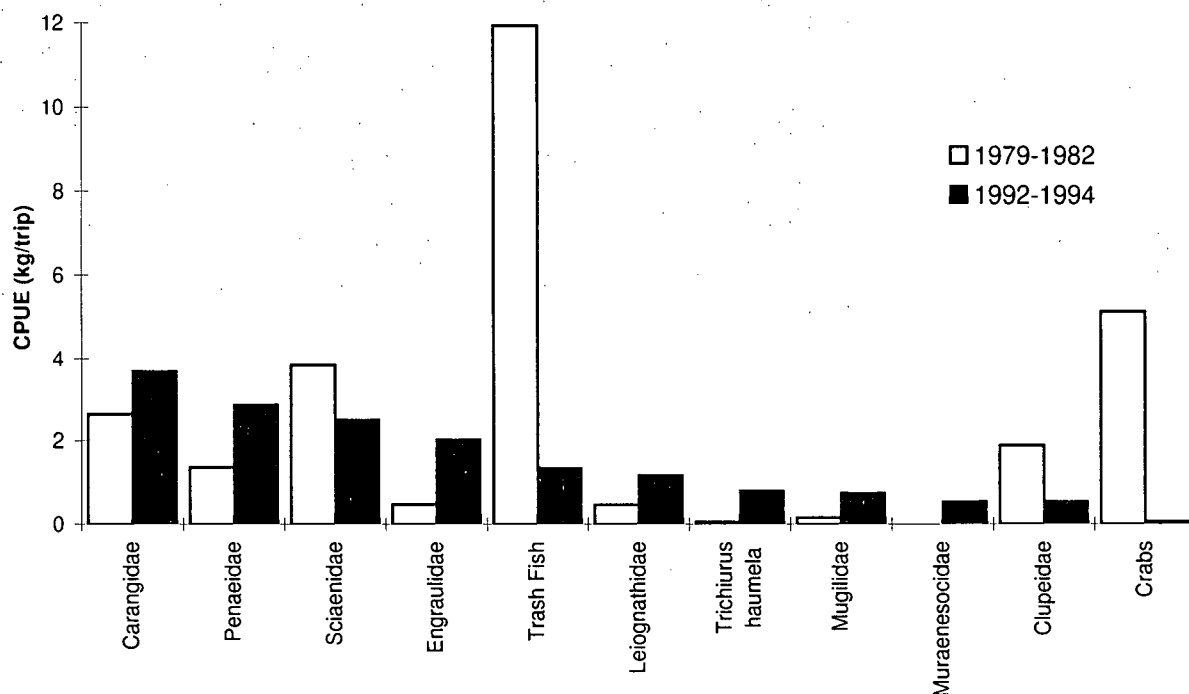


Figure 2.18 CPUE of the top species and groups in the Fish Corral catch from 1979-1982 and 1992-1994.

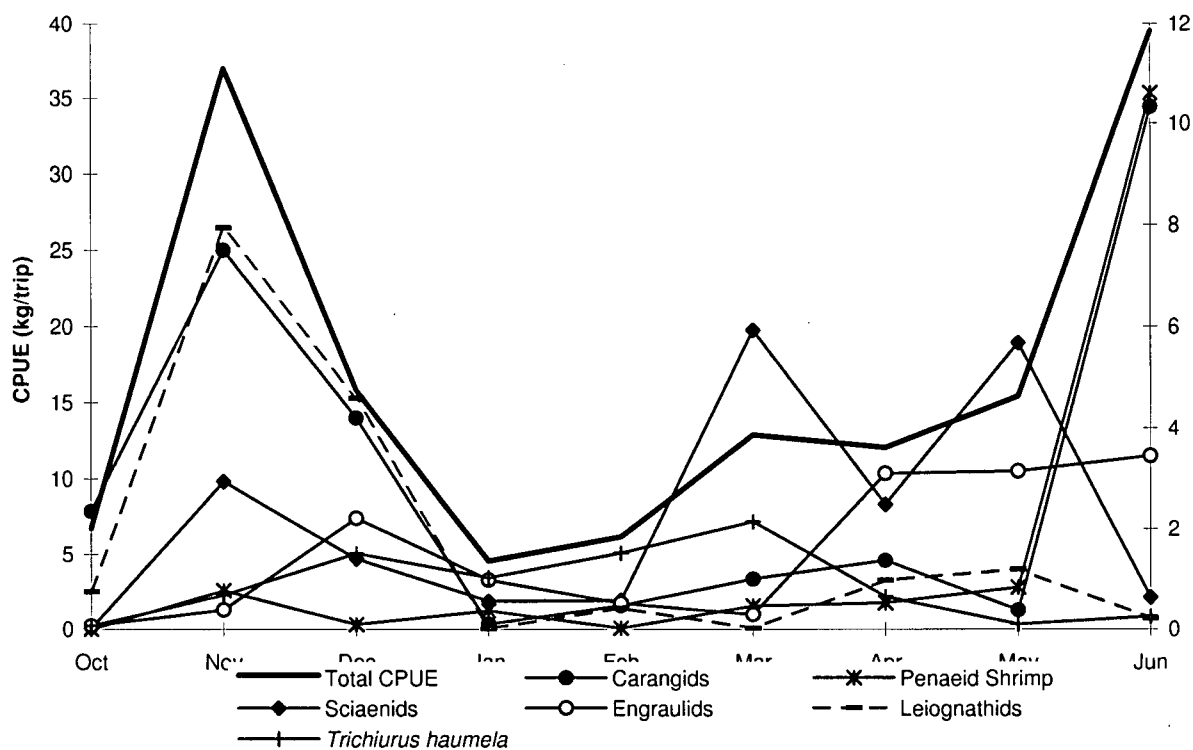


Figure 2.19 CPUE per month for selected species and groups in the Fish Corral catch of 1992-1994.

1994 data being representative of the annual lift net catch. The lift net is estimated, on the basis of its CPUE and effort, to land 50% of the fixed gear annual catch²⁷.

Line Gear

The set longline accounts for 3.2 % of the total catch and the handline accounts for 0.4% of the total catch. Although a small percentage of the total catch, these gears, particularly the longline, catch the larger and older fish in San Miguel Bay.

Forty percent of the longline catch is composed of the carangids and 30% by rays. The shark, *Carcharhinus melanopterus* accounts for another 10%, the eels, Muraenidae and Muraenesocidae, 12% and the centropomid *Lates calcarifer* 5% of the catch. The only comparison that can be made with the 1979-1982 catch is that then, 20% of the longline catch were carangids and 20% pomadasysids (these were not recorded in the 1992-1994 catch). The rest of the catch was not identified (Pauly *et al.* 1982).

The set longline is used all year around. There is CPUE data for the months of December to June (Figure 2.20). Again there is considerable month to month variability. In January and February, virtually only eels are caught and this is the only group which is caught in every month sampled. Carangids are only caught from March to June. From March to June the catch is more diverse.

²⁷ In a parallel survey, designed to monitor fishing operations and conducted by the socio-economic component of the ICLARM project, the lift net was sampled from January to June 1993 and its species composition consisted mostly of engraulids, as in the 1979-1982 survey (Padilla *et al.* 1995).

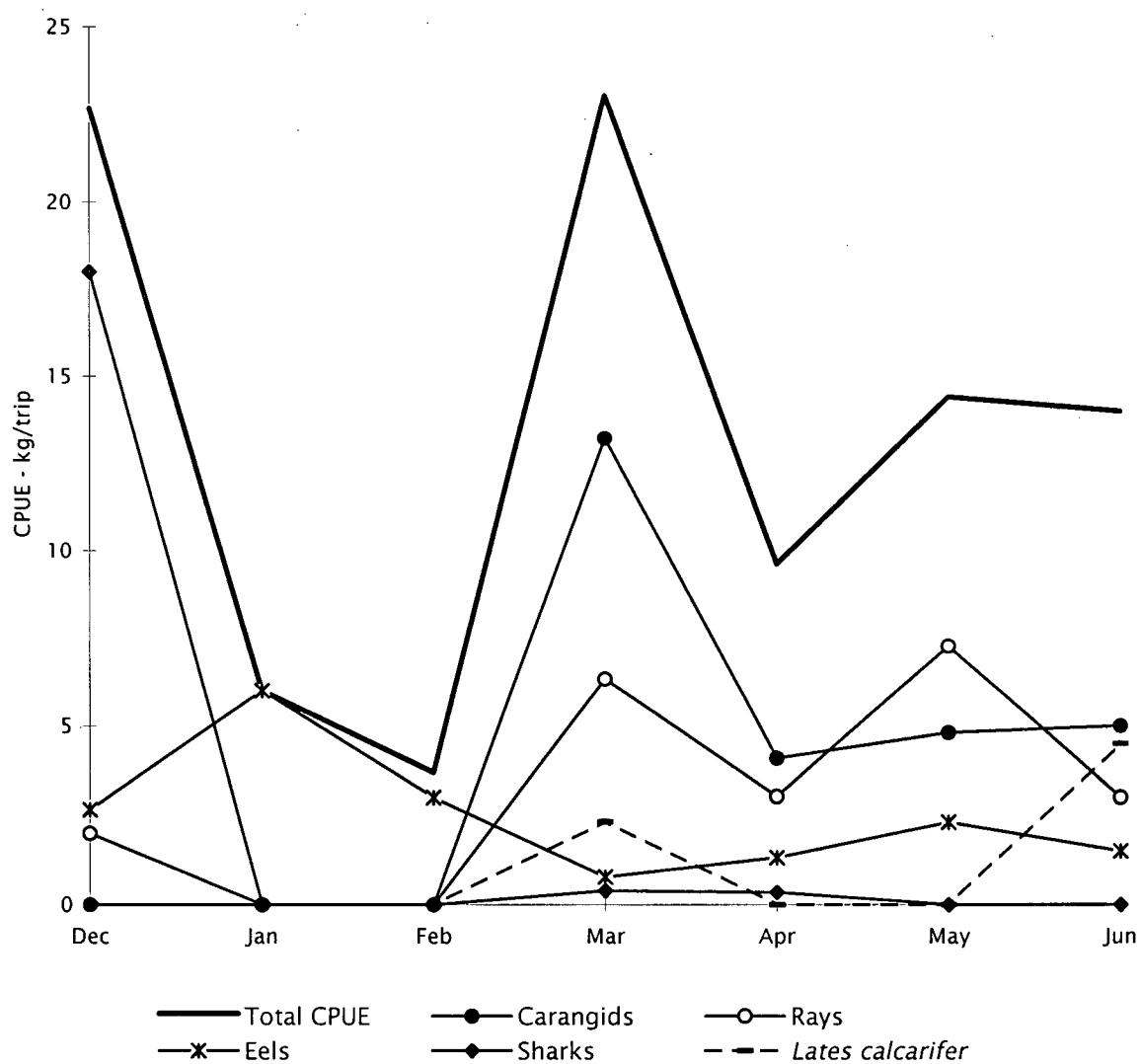


Figure 2.20 CPUE per month for selected species and groups in the Set Longline catch of 1992-1994.

The handline is also used year round. Surprisingly, for it is a simple and cheap gear to operate, the use of this gear has declined since 1979-1982. It declined both in total number of gear and the numbers of trips made per gear (Table 2.13). Its catch in 1992-1994 consists of the sillago, *Sillago sihama*, (67%), the tetraodontid, *Lagocephalus* (24%) and Gobiidae(5%). This composition is derived from only one sample. There is no comparative data from 1979-1982 because the catch then was not identified into its components.

Other Gears

Other gears include the crab lift net, fish trap, push net (or scissor net), as well as the stationary tidal weir, beach seine, spear gun, pullnet and ring net. These gears land 10% of the total catch. The number of push nets decreased since 1979-1982 but the catch increased. As noted above this catch estimate was based on only one sample, which consisted entirely of sergestids. There are however, three types of push net. One operates from January to July, one operates all year, but mainly from September to February and the third operated all year. Combined, the push nets land 7% of the total catch in San Miguel Bay.

The data for the crab lift net and fish trap are similarly scant. The crab lift net operates all year. Two landings were sampled and the catch consists of the portunid crab, *Scylla serrata*. In 1979-1982 the catch was also composed of 100% crabs although it is not specified whether the crabs were *Scylla* species or *Portunus* species. The fish trap is also used all year, but with more effort during April to September. Its catch was sampled in March and was composed of mostly of the rabbit fish, *Siganus javus* and the snapper, *Lutjanus russelli*. There are no data for the

catch composition from 1979-1982. The fish trap is the only gear that catches either the lutjanids or the siganids.

There is no information on the catch distribution or seasonal changes in catch for the other gears listed above. Together, they account for 2.8% of the total catch.

There is no record of discards in the fishery. It is very likely, as suggested by Pauly (1994) that there are no discards or by-catch because all catch is used, be it as table fish or fishmeal, fishfeed or fish sauce. As Pauly notes "most fish caught in Southeast Asia are landed, even when taken incidentally with shrimp (Pauly 1994:99).

Status of the Major Species in San Miguel Bay

For each of the top ten families/groups listed in Table 2.17 an assessment is made of their status. A comparison is made between the modal length of fish in the catch and the length at maturity. For those species with no available length at maturity data, length at maturity is calculated from the L_{∞} using an empirical relationship²⁸. Their seasonal abundance is also described, but this is often confounded by the lack of adequate coverage of all seasons. Where possible, the recruitment pattern²⁹ is compared to the seasonal abundance.

²⁸ The empirical relationship was calculated from the species for which there were data on L_{∞} and L_{mat} . It was determined that L_{∞} was between 1.5 and 2 times L_{mat} . In order to indicate an approximate relationship between mean length in the catch and L_{mat} , it was assumed that $L_{mat} = 0.5 * L_{\infty}$.

²⁹ The recruitment pattern is determined from the trawl survey data length frequency data. FiSAT (Gayanilo *et al.* 1996) software was used. FiSAT has a routine which estimates the likely number of recruitment peaks from a length frequency sample. It is an approximate routine and should be used as a guide only.

Sciaenidae

The results of the mortality analysis and the yield-per-recruit analysis from the trawl survey data all indicated that *O. ruber* is a highly exploited species. Its relative abundance in the fished biomass and the trawl survey CPUE has decreased since 1979-1982. The main gears which catch sciaenids are the ordinary gillnet (41%), the hunting gillnet (36%), the shrimp gillnet (14%) and, to a lesser extent in 1992-1994 than in 1979-1982, the baby trawl (4%). The combined sciaenid catch of these gears has decreased since 1979-1982 and the CPUE of sciaenids by each gear has also decreased. In addition, an analysis of the length composition of the sciaenids (*O. ruber*) in the catch of these gears reveals that the trawlers, mini-trawler, bottom-set gillnet and filter net catch all of their catch of *O. ruber* well below the length of maturity (Figure 2.21). The model length of *O. ruber* in the catch of the hunting gillnet, ordinary gillnet and shrimp gillnet, is not much greater than the length at maturity. Length at maturity was calculated from the empirical relationship between L_{∞} and L_{mat} in Mathews (1990). Also shown is the L_{mat} estimated by Almatar (1993). This is higher than that calculated from Mathews, and if correct, would imply that virtually all *O. ruber* are caught below the length at maturity.

There is no clear seasonal phase of abundance of the sciaenids in Figure 2.22. The CPUE from the trawl survey, the ordinary gillnet and the hunting gillnet track each other well. There may be two peaks in abundance, one in February/March and the other in September/October, which coincide with the end and beginning of the Northeast Monsoon.

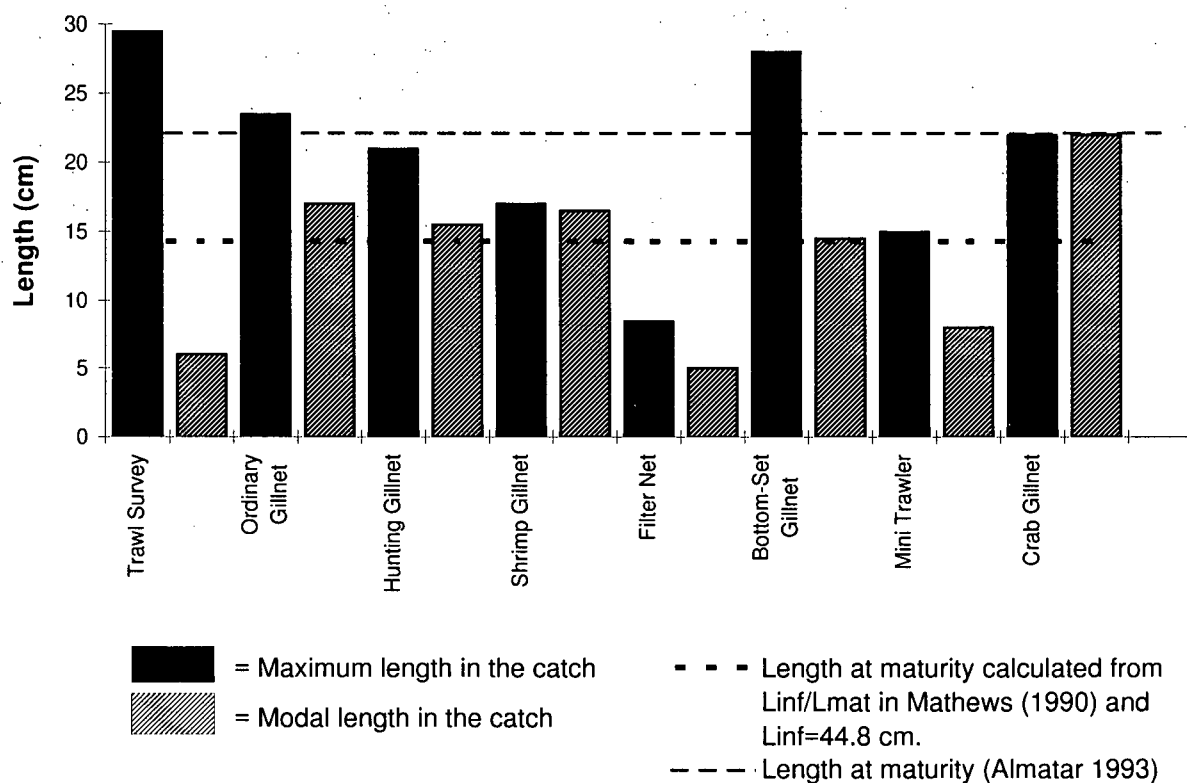


Figure 2.21 Modal lengths of *Otolithes ruber* in the catch.

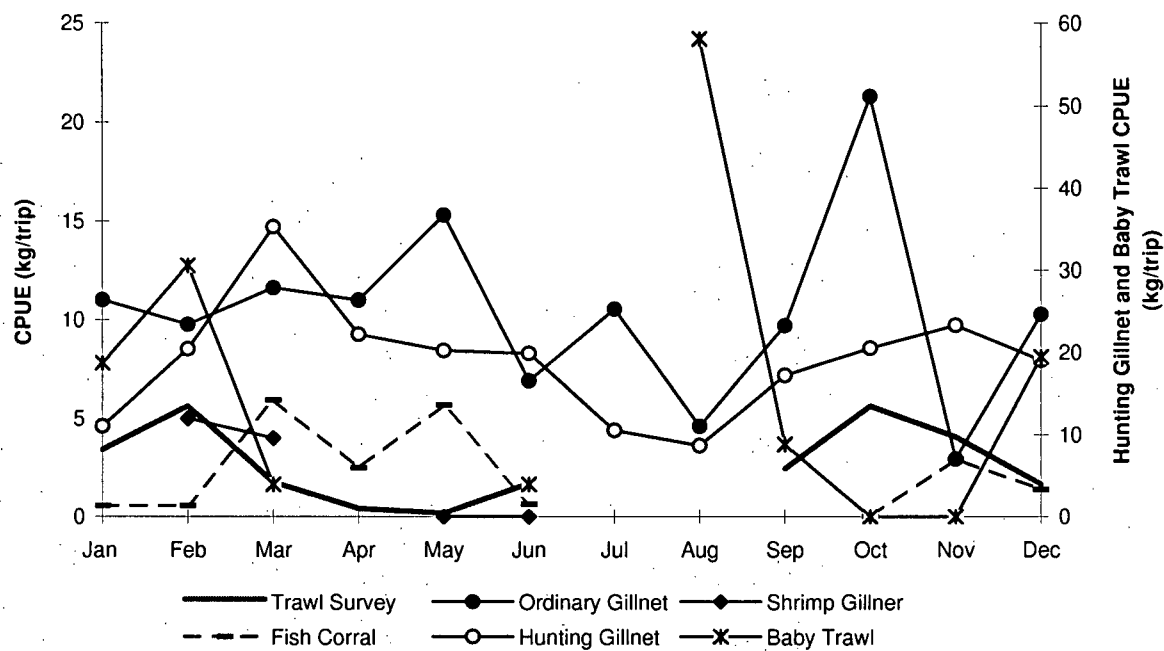


Figure 2.22 Monthly CPUE of the sciaenids by the main gears that catch them.

The recruitment pattern from the trawl survey length frequency data predicts only one recruitment peak.

These results are in accordance with the findings of Navaluna (1982). What is surprising is that the catch of sciaenids is as large as it is, more than 10 years after Navaluna concluded that *Otolithes ruber* was both over-exploited and that mesh sizes should be increased. Fishing mortality has not decreased nor mesh sizes increased since 1979-1982.

Sergestidae

Since the sergestids are not caught by trawl gear other than the mini-trawler, their assessment is based on information from the catch analysis. The results indicate that these shrimps are suffering greater fishing pressure than in 1979-1982. The main gears catching the sergestids are the mini-trawlers, the filter net and the scissor net. The total catch of the sergestids has decreased since 1979-1982 and so has the sergestid CPUE of the mini-trawler which catches almost 80% of the total sergestid catch. The CPUE of the filter net and scissor net increased, although the latter was poorly sampled. Since the mini-trawler catches the greatest part of the sergestid catch and its landings were sampled throughout the entire year, the CPUE of this gear is a more reliable indicator of change. The seasonality of this species was described above for the mini-trawler *pamalaw* (see Figure 2.11.), where the peak in abundance occurred during the period of the Northeast Monsoon.

Penaeidae

The assessment results for the penaeids all indicate that the penaeids stocks are doing well. Their abundance in the trawl survey has slightly decreased but the CPUE has remained at around the 1979-1982 level. Several gears catch considerable quantities of the penaeids: the mini-trawler (27%), shrimp gillnet (19%), other gillnets (16%), baby trawl (15%), the ordinary gillnet (10%). The filter net and fish corral also catch smaller quantities of sciaenids. In all cases, the CPUE of the penaeids has increased, or remained stable, since 1979-1982³⁰, despite increased effort by many of the gears. This contrasts with the results of Pauly's assessment (Pauly 1982a) of the penaeids which indicated that the yield-per-recruit of the penaeids would decrease if fishing mortality was increased.

The monthly CPUEs of the various gears listed above shed little light on the seasonality of abundance of the penaeids. The trends shown in Figure 2.23 indicate that there might be peak in abundance around March and another in September/October. The mini-trawler *pamasayan* CPUE describes a single peak however.

Leiognathidae

Although the relative abundance of the leiognathids in the trawl survey has not changed since 1979-1982 and the CPUE has increased, other indicators suggest that this group is being overexploited. Mortality estimates were made for the three main species in the group. All

³⁰ The CPUE of the other gillnets is not included because it is a composite group of gillnets.

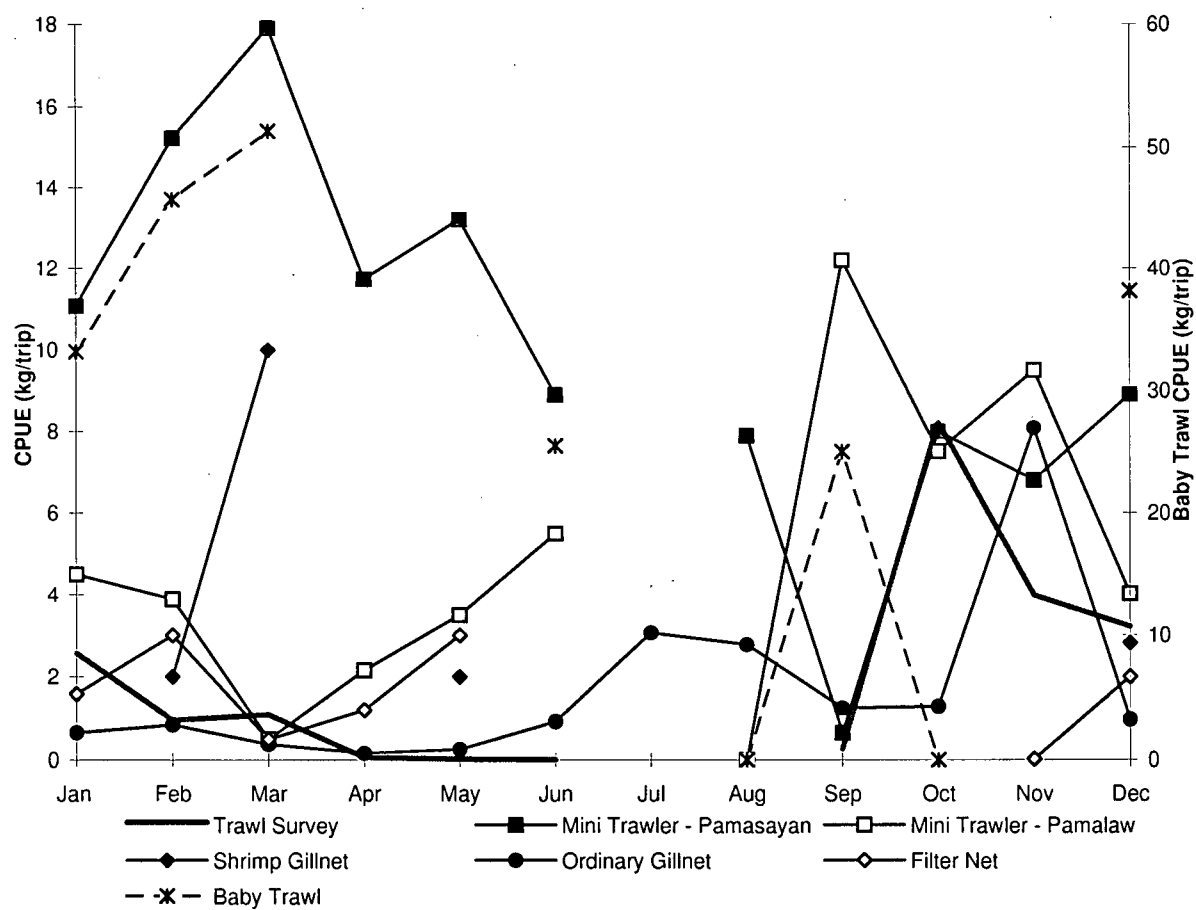


Figure 2.23 Monthly CPUE of the penaeids by the main gears that catch them.

were very high. The yield-per-recruit analysis indicated that the current rate of exploitation is much greater than the optimum rate for two of the three species. The leiognathids are caught by the lift net (62%) and baby trawl (30%) and medium trawl (4%). However, as explained above, the lift net catch estimate is highly uncertain. For this reason, the leiognathid catch of the fish corral (1%) and the filter net (1%) were also examined. The total leiognathid catch decreased and the CPUE of the baby trawl and filter net decreased. The lift net and fish corral CPUE increased.

The modal length of *L. splendens* and *S. ruconius* in the trawl catch is around 6 cm and 5 cm respectively in Figure 2.24. Also shown is the length of maturity for *L. splendens* and *S. ruconius*. Clearly, much of the leiognathid catch is landed before it reaches maturity. There is no length information for the catch of the lift net. However, since it is a fine-meshed gear, it is also, as with the filter net, likely to catch undersized fish. As in the case of the sciaenids, the high proportion of juveniles in the catch is troubling. However it is possible, since generally, juvenile natural mortality is high, that the fishery is catching juveniles that would die anyway, from natural mortality.

The trends in monthly CPUE shown in Figure 2.25 indicate that there is a large peak in abundance from August to November. The August part of the peak is caused by the medium trawlers. There is also a smaller peak in March. The separate recruitment patterns for the three species predict two peaks in recruitment for *L. splendens* and *L. bindus* and one peak for *S. ruconius*. The monthly CPUE of *L. splendens* and *S. ruconius* shows two peaks, one in

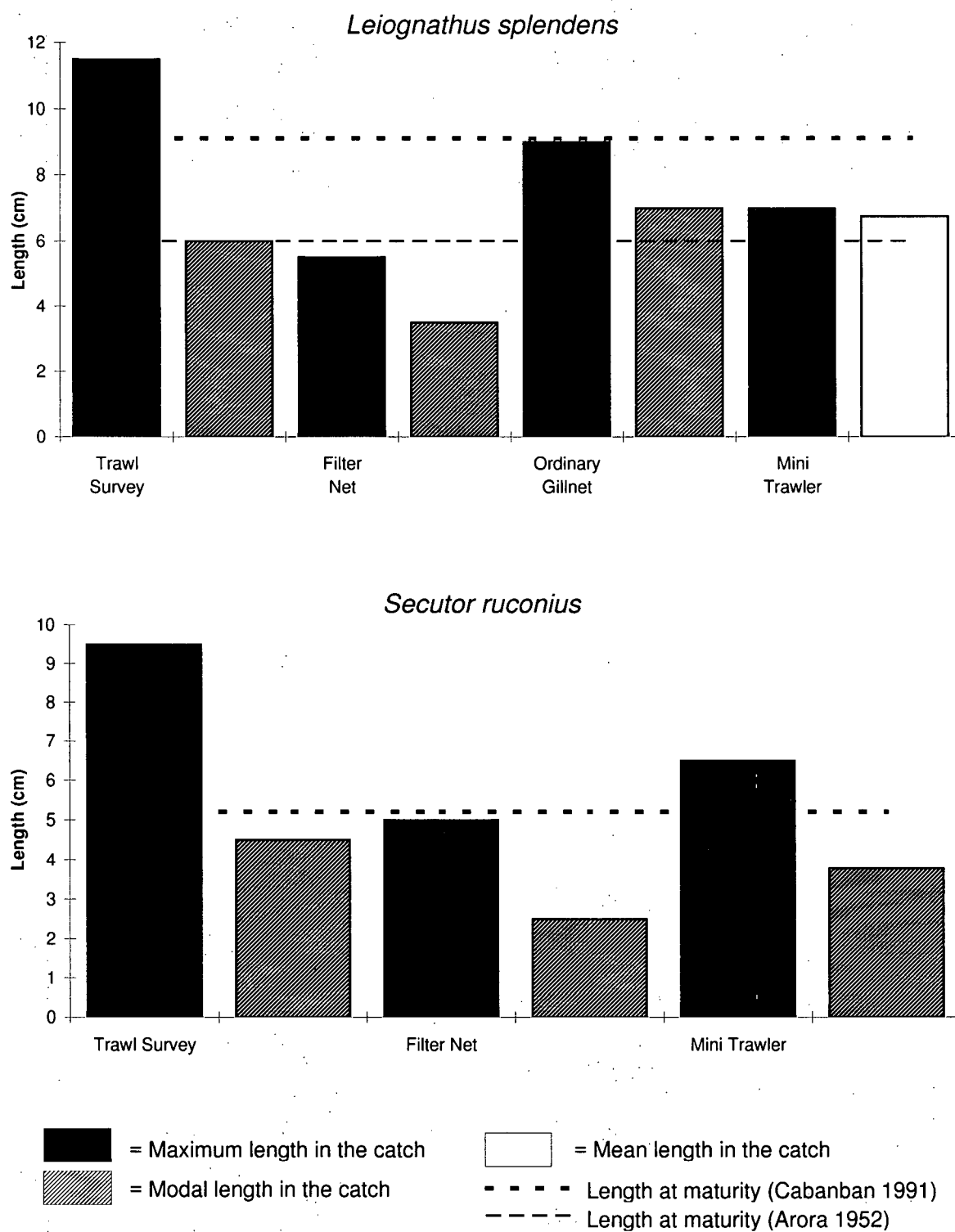


Figure 2.24 Modal lengths of leiognathids in the catch, (a) *Leiognathus splendens*, and (b) *Secutor ruconius*. The mean is used where the mode cannot be calculated.

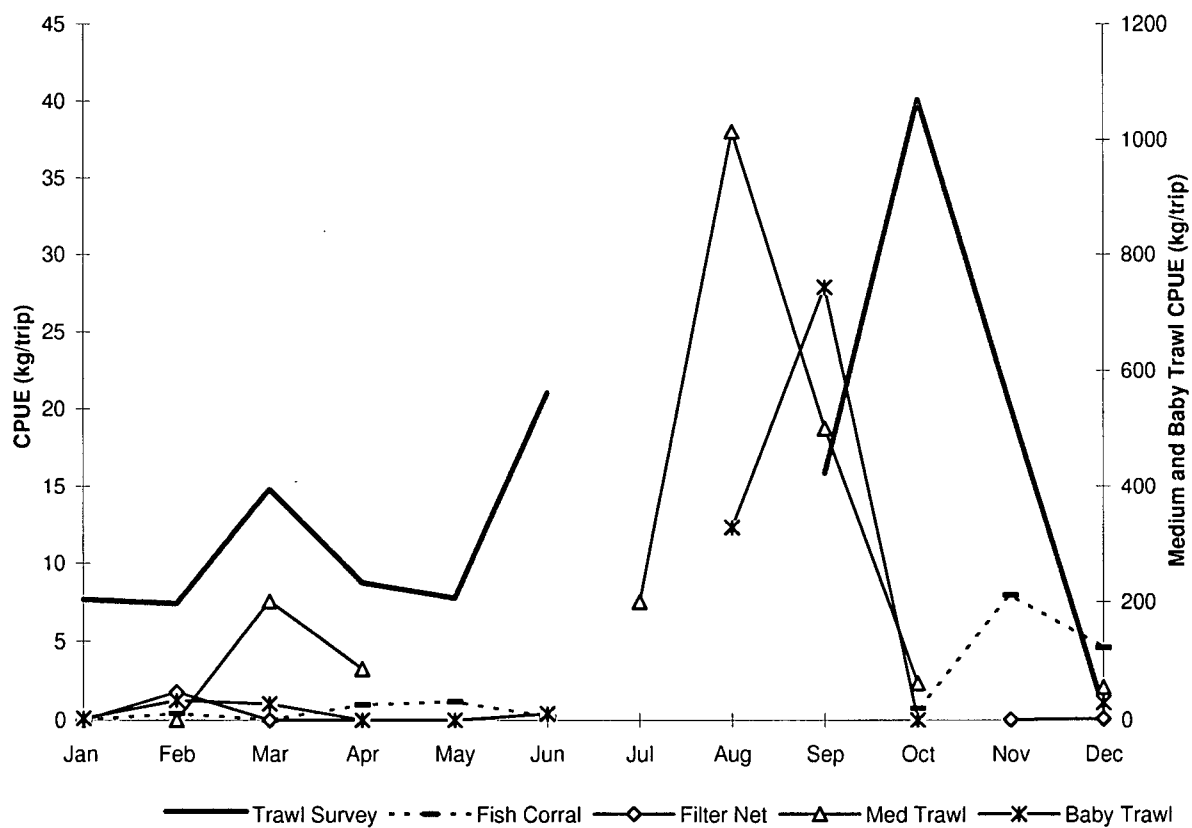


Figure 2.25 Monthly CPUE of the leiognathids by the main gears that catch them.

October/November and one in March, while there is only one discernible peak in CPUE for *L. bindus* in April.

Engraulidae

The relative abundance and CPUE of the engraulids in the trawl survey has decreased since 1979-1982. The total catch has also decreased and the CPUE of the main gears that catch engraulids has decreased. A range of gears catch engraulids. Almost 40% are caught by the baby trawl, 20% by the filter net, 16% by the shrimp gillnet, 6% by the lift net, 5% by the medium trawl and 4% by the ordinary gillnet.

In Figure 2.26 the modal length of *Stolephorus commersonii* in the catch of the trawl, filter net and ordinary gillnet is compared to the length of maturity. As in the cases described above, the modal lengths, and therefore much, if not all of the catch of some gears, is caught before maturity. Pauly (1982a) reported that the cod-ends of trawl nets were covered with a fine-meshed net of 8 mm stretched mesh. Length at first capture was estimated to be 2-3 cm, well below the length at maturity.

The monthly CPUE of some of the gears which catch engraulids are shown in Figure 2.27. The data indicate that there may be a peak in abundance during the summer, the period of the Southwest Monsoon, and one in December/January, the period of the Northwest Monsoon.

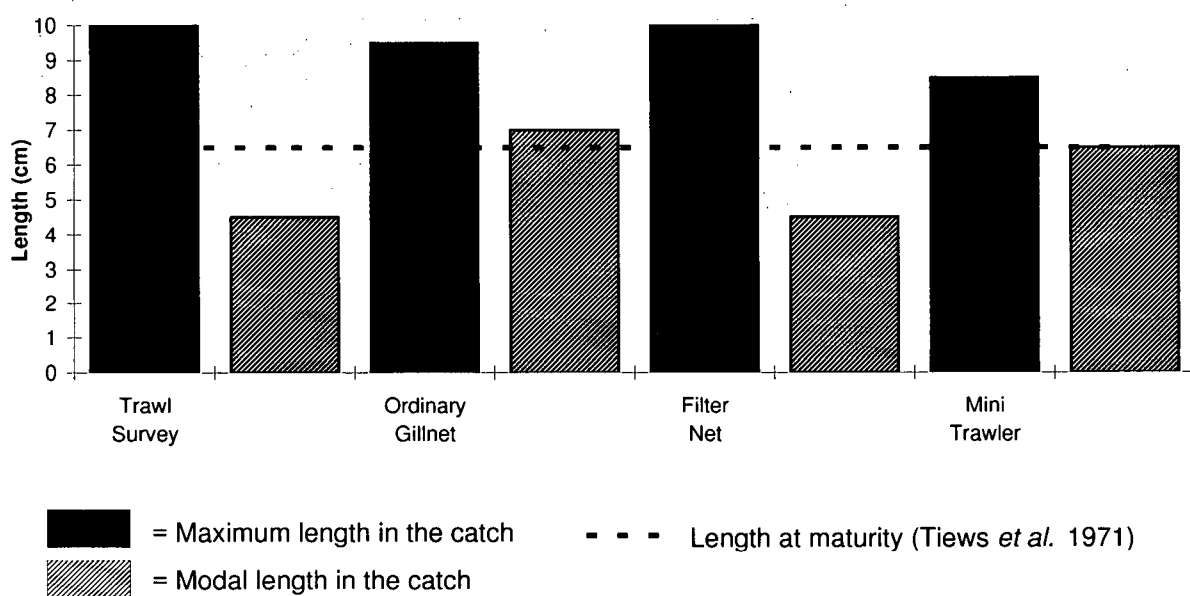


Figure 2.26 Modal lengths of the engraulid *Stolopherous commersonii* in the catch.

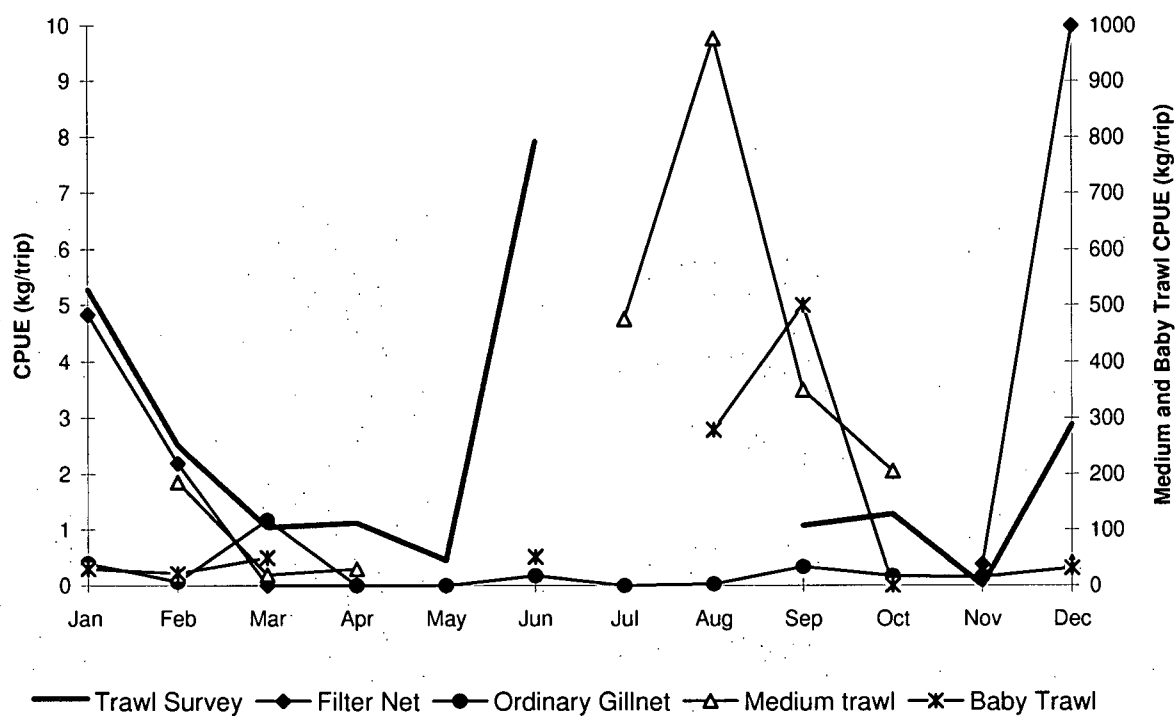


Figure 2.27 Monthly CPUE of the engraulids by the main gears that catch them.

Portunidae - *Portunus pelagicus*

It was noted above that the main gear that catches *Portunus pelagicus* is the crab gillnet which catches 80% of the total catch. The baby trawl, mini-trawler, and several of the gillnets take incidental catches. The catch and CPUE have increased since 1979-1982. In the 1979-1982 study, it was reported that crab fishers had complained that their catches had been decreasing (Pauly 1982a). However, the available data for 1992-1994 indicate that this species is not over-exploited and indeed may have increased in abundance since 1979-1982.

Trichiuridae

Trichiurus haumela is another species which is not currently suffering from over-exploitation. Its abundance and CPUE in the trawl survey has increased since 1979-1982. Relative to the other species for which mortality was estimated, its mortality is low. The yield-per-recruit curve places the current rate of fishing mortality at around the optimum level. The total catch of *Trichiurus haumela* has increased and the CPUE of gears which exploit it, the ordinary gillnet (50% of the catch of *Trichiurus haumela*), the baby trawl (27%), and the hunting gillnet (17%) has increased. In addition, as shown in Figure 2.28, these gears catch mature fish.

The monthly CPUEs from the trawl survey and fishing gears are shown in Figure 2.29. There is a definite period of high abundance which ranges from October/November through to

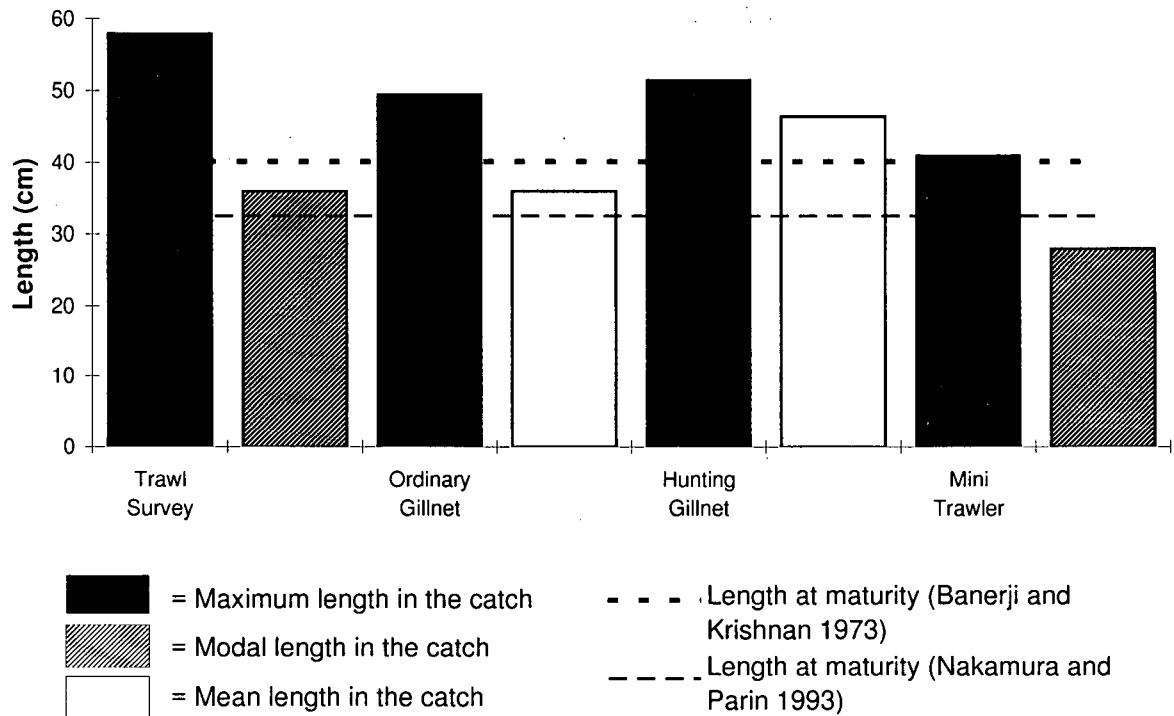


Figure 2.28 Modal lengths of *Trichiurus haumela* in the catch. The mean is used where the mode could not be calculated.

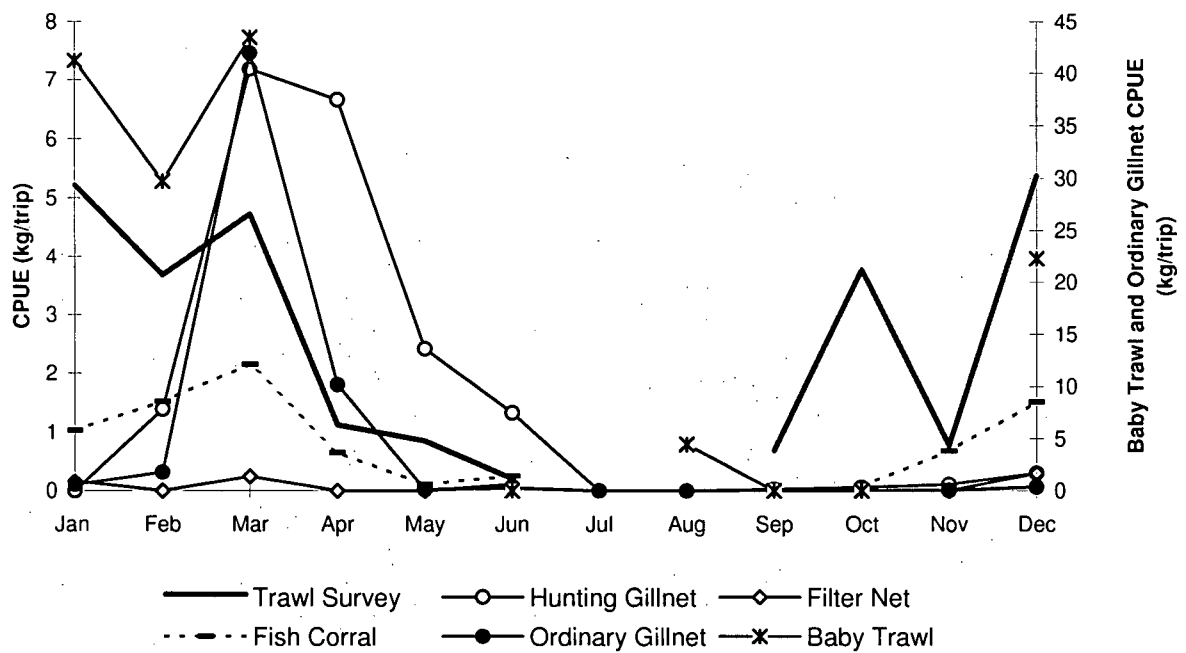


Figure 2.29 Monthly CPUE of *Trichiurus haumela* by the main gears that catch them.

March/April, the period of the Northeast Monsoon. The recruitment pattern from the trawl survey length frequency data has one strong peak, lasting 6-7 months.

Mugilidae

The CPUE of the mugilids increased in the two gears that each catch over 30% of the total catch of the mugilids. However, all data indicate that the mugilids are overfished. Their abundance and CPUE in the trawl survey decreased. The total catch decreased and the baby trawl and ordinary gillnet CPUE of mugilids decreased. It is possible that the bottom-set gillnet and surface gillnet fishers have discovered means to improve their catches. In addition, since the effort of these two gears has decreased since 1979-1982, there will be less competition amongst remaining fishers. In Figure 2.30, the modal length of mugilids in the catch are shown. The length at maturity is calculated from the L_{∞} using the empirical relationship described above. Although the L_{mat} is thus only a hypothetical value, it is not likely to be lower than shown in Figure 2.30, yet the modal length in the catch is lower. The monthly CPUE data are too sparsely distributed through the year to give any sense of seasonal abundance.

Carangidae

The carangids are a large group, with over 14 species occurring in San Miguel Bay. As a family the abundance of the Carangidae in the trawl survey has remained stable and the CPUE has increased slightly. The total catch and catch CPUE have also increased. The set longline

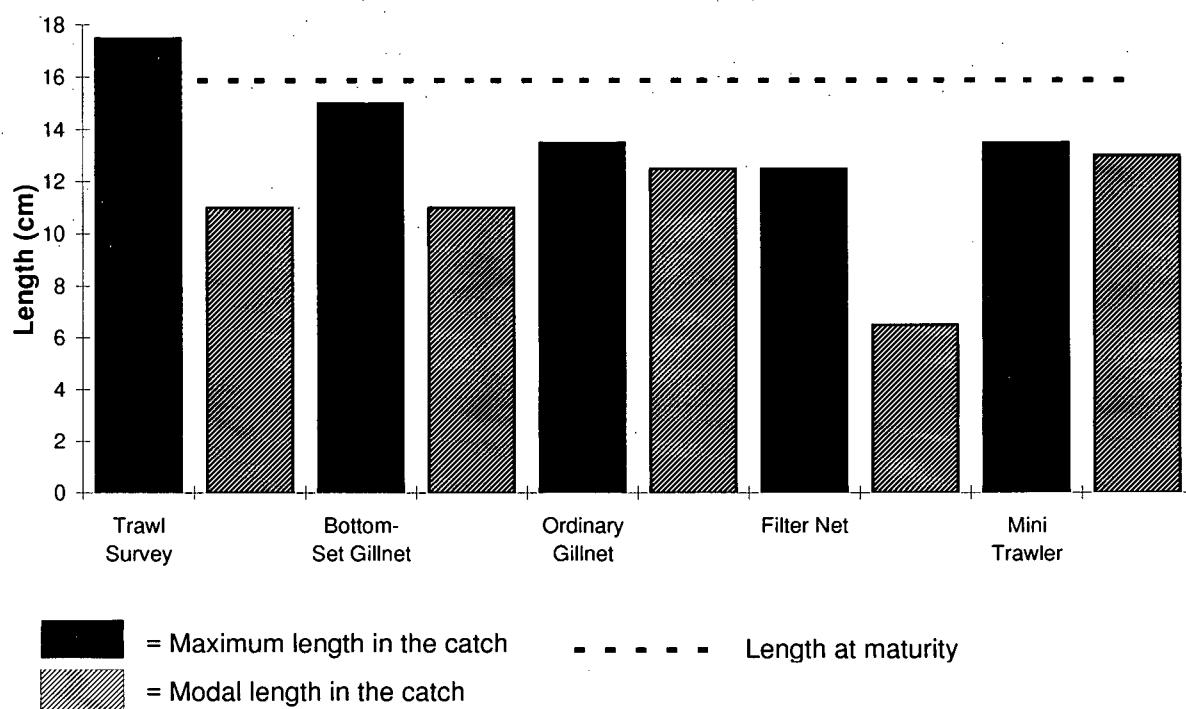


Figure 2.30 Modal lengths of Mugilidae in the catch. Length at Maturity is calculated from an empirical formula - see text for further details.

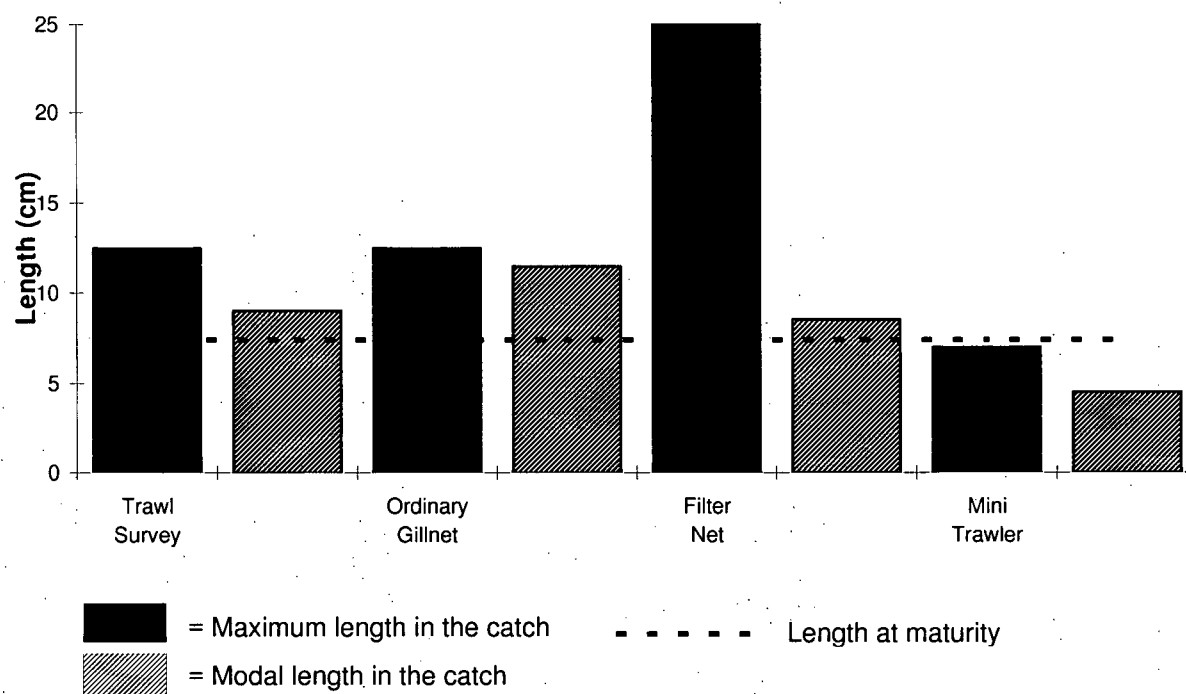


Figure 2.31 Modal lengths of the carangid, *Alepes dejedaba* in the catch. Length at Maturity is calculated from an empirical formula - see text for further details.

lands 42% of the carangids, the ring net³¹ 19%, the baby trawl 13% and the fish corral 10%.

The modal length in the catch is shown in Figure 2.31. L_{mat} was calculated as described above.

For *Alepes djedaba*, the only carangid for which there were data on the length composition of the catch, most of the catch is above the estimated length of maturity. As in the case of the mugilids, the monthly CPUE data of the carangids are too sparsely distributed through the year to give any sense of seasonal abundance.

Trash Fish

The baby trawlers land 47% of the trash fish (see footnote 14), the mini-trawlers 33%, the filter net 9% and the hunting gillnet 5% of the trash fish. The monthly CPUE data indicate that there might be a higher catch rate of trash fish from November to March/April. However, the data are too scant to make a firmer statement on the seasonal distribution of the trash fish.

Value of the Catch

The top families listed in Table 2.17 are also some of the most valuable in the catches of San Miguel Bay. In particular, the ex-vessel price of the larger crustacean species, the penaeids and the portunids are the highest in the total catch, with the exception of the Serranidae. The latter are priced at between 200 and 240 pesos/kg³², while the penaeids make an average price of between 42 and 86 pesos/kg and the portunids between 14 and 122 pesos/kg (Padilla *et al.* 1995). The sciaenids have an average ex-vessel price of between 12 and 46 pesos/kg, the

³¹ There are data for two landings for the ring net in the Catch data from San Miguel Bay from July.

³² At the current, 1997 foreign exchange rate, US\$1 = 26 pesos: 240 pesos is equivalent to \$9.23.

carangids 33 to 60 pesos/kg and the mugilids 27 pesos/kg. The leiognathids, engraulids, trichiurids and sergestids are all less valuable. The species listed above that were not recorded in the 1979-1982 catch, such as the Mullidae and the Synodontidae, are some of the lowest value species.

Discussion

Given the assessed status of San Miguel Bay in the early 1980s, it is hard to conceive that the fishery could have improved in the intervening years, since none of the recommendations of the 1979-1982 survey were enacted. In 1982, a 5 year ban on the operation of commercial fishing boats did commence, yet it apparently only affected a small number of commercial operators (Luna 1992), and had little or no effect. All the evidence suggests that, for most species, the status of the fishery has worsened. A series of analyses were presented above, and essentially all confirm that the Bay is highly overexploited. Since there is no time series of catch and effort data for San Miguel Bay, much of this analysis was based on length data from the trawl survey and comparative analysis with data from the 1979-1982 survey. In the latter, comparisons were made between the composition of the trawl survey, the catch, CPUE per gear and effort changes.

In their emerging fisheries classic, Hilborn and Walters (1992) devote a chapter to length-based methods in fisheries assessment. They do not hide the fact that they are not enthusiasts of the approach, and indeed point to many of its difficulties and uncertainties. Their principal

point is that obtaining a representative sample, covering all sizes and ages in the population, is confounded by gear selectivity and fish behaviour, including fish movement, gear avoidance and distributional changes related to ontogenic migration. Many of these concerns have been addressed by researchers, who have examined the sensitivity of length-based methods to some of these problems (see papers in Pauly and Morgan (1987) for example).

In the San Miguel Bay survey, the Bay was sampled by trawl survey. Since the vast majority of the Bay is sand and sandy-mud, few areas were inaccessible to trawlers. The trawl survey was distributed throughout the year (with the exceptions noted above) and throughout the Bay (see Figure 2.2). In the analysis of the length data, the FiSAT program was used to allow for selectivity by the trawl gear. Only data from the trawl survey were used to estimate growth parameters and mortalities. In addition, growth parameters were only estimated for the species for which there were visible modes that could be tracked over several months. It was assumed that there was no net movement of fish in or out of the Bay, in accordance with the steady-state assumptions of the length-based methods used³³. The implication of this assumption is discussed below. These measures were taken to ensure, as far as possible, a representative sample.

In order to estimate mortalities and yield-per-recruit, the growth parameters were first estimated using ELEFAN I. The ELEFAN I method was used because it is well tried and tested (Pauly 1987, Hampton and Majowski 1987, Majowski *et al.* 1987, Rosenberg and

³³ In fact in all likelihood, emigration and immigration occur in San Miguel Bay. Pauly (1982b) has described the nursery role of San Miguel Bay. However, it is questionable, as a consequence of the observed level of fishing pressure in San Miguel Bay, whether many of the adult fish escape the Bay before they are captured. The question of immigration and emigration is returned to in Chapters 3 and 5.

Beddington 1987), it uses a time series of data, it has routines which allow for the effects of selectivity to be accounted for and the results can be used directly in the mortality estimation using the FiSAT software. In a sensitivity analysis of ELEFAN I, Rosenberg and Beddington (1987) found ELEFAN I to be sensitive to variation in length at age (assumed constant in ELEFAN I). In addition, ELEFAN I consistently underestimated the growth parameter K , unless the true value of K was known to within 20-25% of its true value. Hampton and Majowski (1987) concluded that L_{∞} was over-estimated and K underestimated when fishing is size selective. However, both of these simulation trials were conducted on earlier versions of ELEFAN I, and the program has since been improved (Pauly 1987, Gayanilo *et al.* 1996). In any case, the rationale of using the generic values of L_{∞} and K (estimated from the literature) as guides in the analysis was to overcome such biases and to avoid the problem of the correlation between L_{∞} and K . Of the large number of species for which there were length frequency data, growth parameters were only estimated for six, because it was not possible to discriminate a growth curve from the data for the other species. Whilst not wishing to appear too confident about the growth parameters estimated above, there is sufficient certainty that they are in the right "ball park".

The length-based data and growth parameters were used to estimate mortality and yield-per-recruit. Mortalities were estimated using the Length Converted Catch Curve, Beverton and Holt's Mean Length Method, the Powell-Wetherall Plot, and the relationship between fishing mortality, catch and biomass. The first two methods were considerably more successful than the latter two methods. Four methods were used because of the uncertainties associated with

length-based methods to estimate mortality. All are based on the equilibrium assumption and accompanying assumptions.

Hampton and Majowski (1987) found that Z was generally overestimated by the Length Converted Catch Curve Method and that the greater the variation in growth, the greater the positive bias. In addition, variation in cohort strength, which is assumed constant, can cause bias. In the catch curve analysis above, the samples were pooled in order to smooth out recruitment pulses and thus better simulate equilibrium conditions (Pauly 1987).

Recruitment variation also causes bias in the Beverton and Holt Mean Length Method (Ralston 1989). A recruitment surge causes a negative bias and vice-versa. As above, pooling samples, taken over a period of time, reduces the bias. It is also sensitive to the input parameters, L_{∞} , L_c , the length at first capture, and the difference between mean length and L_{∞} , or L_c . The sensitivity to L_c was shown above in the mortality estimation for *Leiognathus splendens*. A L_c of 6.5 cm produced a Z estimate of 12.8 year^{-1} while a L_c of 6 cm gave a Z estimate of 10.2 year^{-1} . It is however, quite robust against variation in individual length at age since it uses the mean length (Laurec and Mesnil 1987). Walters and Hilborn suggest that "only the most naive biologist would use this method with any confidence" (1992:425). Its use here is justified since it was used in conjunction with other methods. The Z estimates from the Beverton and Holt method were often in agreement with the estimate from the Length Converted Catch Curve.

The Powell-Wetherall Plot was unsuccessful and only produced one useful Z estimate. The estimation of M from Pauly's equation produced realistic values, but the estimates of F

generally resulted in total mortality estimates which were either much greater or much lower than the Length Converted Catch Curve or the Beverton and Holt estimates. Given the uncertainty surrounding the biomass estimate (see above), this result is not too surprising.

That length-based methods are more approximate than age-based methods leaves little room for dispute. Many authors begin their texts with an admonition about these methods, while recognising that for many fisheries, particularly in the tropics, the only option to not using length-based methods would be to use no methods at all, and to wait for better data. The same argument is used here. The hope, in estimating mortality, was not to get pin-point figures, but to obtain a sense of the likely mortality that is imposed on San Miguel Bay fishes.

This was achieved for six important species in San Miguel Bay. Their levels of mortality can be used as an indicator of mortality levels in the fishery generally. The yield-per-recruit analysis confirmed that these species, with the exception of *Secutor ruconius* and *Trichiurus haumela* are highly overexploited.

The estimated mortalities are very high. There are at least four explanations for these high mortalities.

1. The mortalities are over-estimated for the reasons discussed above, that is the sensitivity of the methods. (For example, the Length Converted Catch Curve and Beverton and Holt methods gave Z estimates of between 10 and 12 for *Leiognathus splendens* while the $Z=F+M$ method estimated Z to be half of this value).
2. The Z s are high because of emigration from the Bay.

3. The Zs are high because the samples are largely juvenile and they have higher mortality than adults.
4. The Zs reflect the actual mortalities. Such high Zs are known in other tropical systems. For example, the mortalities estimated by Tandog-Edralin *et al.* (1988) for heavily and moderately fished areas of the Philippines have a similar range.

In essence the actual values of Z are less important than their range. The relevant message is the recognition that the extent of over-fishing of San Miguel Bay has not improved since 1979-1982. The results from the analysis of the trawl survey data and the catch survey data conclusively give the same result.

The fishery analysis above showed that while effort had decreased in the large-scale sector, both diversity and effort increased in the small-scale sector. Yet, Sunderlin (1995) reports that the number of small-scale fishers has remained effectively unchanged since 1979-1982. At that time, the total of 5,600 fishers had grown rapidly since 1939 (Bailey 1982). In 1992-1994, there are 4,800 full-time fishers and 500 part-time fishers³⁴, a slight decrease from the 1979-1982 figures. Furthermore, although the population of fishing villages has grown, the proportion of fishing households decreased from 54% to 45% of the total number of households. There was a greater dependence in fishing households on non-fishing income and income from females and children (Sunderlin 1994). These socio-economic factors indicate that the increase in small-scale effort and diversity is an effort by fishers to maintain incomes, not an effort to either increase incomes or to increase the number of fishers. Sunderlin also

³⁴ Sunderlin defines full-time fishers as those who get all or most of their income from fishing and part-time fishers as those who get all or most of their income from a non-fishing source. (Sunderlin 1995).

reports that 2/3 of small-scale fishers interviewed, complained that their catches had decreased over the previous two years, 36% said it was the same and 14% said that it was better. Given that the total small-scale catch has increased slightly and the total number of small-scale fishers decreased, on average there should be at least no change in catch.

The fishing population has stagnated, the total catch has decreased and the assessed status of the fishery has worsened since in 1979-1982. However, Padilla *et al.* (1995) report that economic rents are still made in the fishery. Padilla *et al.* examined the economic performance of eight gears, the large, baby and mini trawls, the lift net, filter net, fish corral, pushnet and gillnet³⁵. They concluded that pure profits were made by the mini-trawler, fish corral, filter net, gillnets and pushnet, while the baby trawlers and lift nets made negative pure profits (a loss). In total, they estimated that the total value of rents were equivalent to their value in 1979-1982.

In general the level of investment required to start in the fishery decreased between 1979-1982 and 1992-1994. One exception was the Push Net. The relative increase in the start-up costs for this gear could explain why the number of Push Nets have decreased since 1979-1982 (Table 2.13). The large increase in the number of filter nets could be related to their relatively low start-up costs compared to the other fixed gears, trawl gears and gillnets. In addition, the start-up costs of gillnets, the number of which has substantially increased since 1979-1982, decreased relative to the start-up costs of the trawl gears, lift net and fish corral.

The decline in the large-scale trawling sector may be for economic reasons. Padilla *et al.* showed that, over a period of 6 months, the large-scale sector made a slight positive income,

but when opportunity costs were taken into consideration, this sector suffered a loss. However, Padilla *et al.* used catch and effort figures which are different from those collected in the landings survey described above. They used catch and effort figures that were collected in a socio-economic survey and monitoring of fishing operations, a survey that was carried out in parallel to the landings and trawl surveys. From this survey, information was collected on catch, operating expenses, crew remuneration and the dynamics of fishing units over a period of 6 months from January 1993 to June 1993 (Padilla and Dalusung 1995).

Padilla *et al.* calculated the catch composition and CPUE for the eight gears monitored in the socio-economic survey. These figures do not correspond well with the results from the landings survey. For example, the CPUE of the gillnets calculated by Padilla *et al.* was 9.8 kg/trip, whereas the average CPUE for the gillnets above was 15.8 kg/trip. The discrepancy between the CPUEs for the baby trawl between the two surveys was also very large. Padilla *et al.* concluded that the baby trawl made a negative pure profit.

In Table 2.19, the net income is calculated for six of the eight gears using the CPUE and catch figures calculated from the landings survey above, and the economic figures in Padilla *et al.*

³⁵ Padilla *et al.* (1995) combined all gillnets into one category.

Table 2.19 Comparison of the Net Income derived from the catch figures in Padilla *et. al* (1995) and the catch figures calculated from the Landings Survey above.

| | <i>Value (Pesos*)</i> | <i>Value Pesos Padilla et. al (1995)</i> | <i>Costs (Pesos)</i> | <i>Net Income</i> | <i>Net Income Padilla et. al (1995)</i> |
|--------------------|---------------------------|--|--------------------------|-----------------------|---|
| Fish Corral | 3831 | 8272 | 3415 | 417 | 4857 |
| Filter Net | 2713 | 2300 | 1031 | 1683 | 1270 |
| Mini-Trawler | 11620 | 8361 | 6053 | 5567 | 2307 |
| Baby Trawl | 131435 | 8782 | 8277 | 123,158 | 505 |
| Large Trawl | 122944 | 104, 501 | 498 | 122,446 | 257 |
| Gillnet (combined) | 7716 | 5975 | 5011 | 2705 | 963 |

* 26.4 pesos = US\$1

(1995).³⁶ The net income figures from Padilla *et al.* (1995) are also given for comparison.

They are quite different. Only the results for the filter net are comparable. The differences seen for the other gears are caused by the different catch rates and hence catches.

On the basis of the data presented in this thesis, it cannot be concluded that the trawl gears are uneconomic. When the catch figures estimated in this work are used in the economic analysis, the net income is much greater than the net income estimated by Padilla *et al.*. This is also the case for the other gears listed in Table 2.19 with the exception of the fish corral. Therefore, all gears must make pure profit, except the fish corral, which may make a negative pure profit.

In summary, it could be said that the fishery of San Miguel Bay is in a state of *laissez-faire* anarchy. Management regulations exist, but in practise internal and informal sources regulate the level and type of effort. The main management measures have been mesh size and trawling restrictions, but neither of these have had much effect. The results of the analysis of the mean length in the catch of various gears attests to the former. That trawling regulations are not as effective as they should be is recognised by Luna (1992), Mike Pido (pers. comm.) and personal observation of fishers' complaints.

However, despite dire warnings at the beginning of the 1980s, the fishery still operates, it still sustains about the same number of fishers and it still generate profits in most sectors of the fishery examined. There is also greater equity in the fishery. The distribution of the catch and

³⁶ The net income for the lift net and the scissor net were not re-calculated because the data from the landings survey are based on one sample and therefore highly uncertain. In the case of these two gears, the catch data in Padilla *et al.* (1995) is more complete.

profits has changed with the small-scale sector now taking a larger, and more equitable share. Many of the species are over-fished, but other species are succeeding, for example, the portunid crabs and the trichiurids. The results indicate that, as in the Gulf of Thailand, this fishery is being fished down and is following the classic pattern, described by Pauly (1979a). Other than a severe reduction in effort by all sectors or complete closure of the fishery, it is difficult, on the basis of this analysis and the lack of adherence to the management regulations, to make specific recommendations. The increase in the fine-meshed gears such as the filter net, and the development of the hunting gillnet, an active gear, are indications that, despite the reduction in trawling effort, this fishery is going down a one-way street.

Chapter 3

An Ecosystem Model of San Miguel Bay

“The approach we propose is thus to use state and rate estimates for single species in a multispecies context, to describe aquatic ecosystems in rigorous quantitative terms, during the (arbitrary) period to which their state and rate estimates apply.”
Christensen and Pauly (1992a:2)

Introduction

San Miguel Bay has been described as a tropical multispecies fishery with a diverse small-scale sector and a large-scale sector. The latter has dominated the fishery, until recently. The Bay has been diagnosed as seriously overfished, but there is little real insight into how to ameliorate this situation, other than to reduce all fishing and prohibit trawling. The number of trawlers operating in the Bay has already declined by around 50% since 1979-1982, yet there is no noticeable improvement in the fishery resource as a consequence. The situation is complex because the multispecies resource is selectively and non-selectively exploited by a large number of diverse fishing gears.

Little ecological detail about the ecosystem on which the fishery is based is available, yet alone the impact of the different gear sectors. There are many species in the ecosystem, and some, probably many, are over-exploited. Pauly (1982a) speculated on the nature of likely interactions between some of the species in San Miguel Bay. He noted that the penaeids are predated upon and are in competition with a number of fish species. Since the penaeids

command one of the highest ex-vessel prices in the catch (Padilla *et al.* 1995), these relationships are important. Trawling has effectively reduced the biomass of many of the competitors and predators of the penaeids, thus promoting the huge surge in penaeids seen in San Miguel Bay, and in other fisheries in Southeast Asia (Pauly 1982a, Pope 1979). The leiognathids, for example, were and, although they have declined, are the most abundant species in San Miguel Bay. They are also competitors with the penaeids. Arguably, the penaeids have "profited" by the decline of the leiognathids. Other changes that have occurred in the species composition of San Miguel Bay since the late 1940s and the early 1980s were noted in Chapter 2. Directly and indirectly these changes are due to exploitation of the fishery.

When assessing and managing a multispecies fishery such as San Miguel Bay it would be unwise not to consider such ecological interactions. Until recently though, no ecological or multispecies modelling of San Miguel Bay had been conducted. The data from the 1992-1994 ICLARM survey is used here to construct a simple, linear mass-balance model of San Miguel Bay using ECOPATH³⁷ (Christensen and Pauly 1992a, 1992b, see also Palomares *et al.* 1995a³⁸). ECOPATH has been applied widely to aquatic systems (see contributions in Christensen and Pauly 1993a). It has been used to describe and examine the energy flows in ecosystems (e.g., Jarre-Teichmann *et al.* In Press, Christensen 1994, 1995) and as a diagnostic tool (e.g. Pauly and Christensen 1995, 1996).

³⁷ ECOPATH is available as a DOS version, ECOPATH II (Christensen and Pauly 1992a) and in a new Windows version, ECOPATH 3.0 (ICLARM 1995c). When this work was undertaken, only the DOS version of ECOPATH was available. When ECOPATH 3.0 became available results were checked and elaborations on method made - see below.

³⁸ Palomares *et al.* (1995a) made a first attempt at an ECOPATH model of San Miguel Bay. However, they relied almost exclusively on the data and structure of the ECOPATH model for Brunei Darussalam (Silvestre *et al.* 1995).

ECOPATH is a mass-balance description of trophic interactions. It is used here to model San Miguel Bay and to determine and describe the interactions between different components within the ecosystem. In this way, the major energy flows and pathways in the ecosystem, upon which the fishery is based, are identified. The state of development or maturity of the ecosystem is also examined using a series of indices outlined in Christensen and Pauly (1993c) and Christensen (1995). In addition, the key areas where information is poor are also identified. ECOPATH is a means to collate data about a system in a coherent form, enabling a better understanding of the entire system. Knowledge is increased.

That the changes in species composition noted in San Miguel Bay are due to the direct and indirect effects of fishing bears little questioning³⁹. The trawling sector of San Miguel Bay, and the small-scale sector are equally likely candidates for altering the ecosystem of the Bay. In order to directly model the effects of fishing, the fishery was included in the ECOPATH model as a "top predator". The fishery was modelled as a large-scale predator and a small-scale predator, and as a large-scale predator and a series of different small-scale predators.

³⁹ Environmental effects such as the destruction of mangrove habitats (Vega *et al.* 1995a,b), siltation (Mendoza and Cinco 1995), and pollution (Mendoza *et al.* 1995b) are also contributing factors.

Methods

ECOPATH

ECOPATH is a mass-balance model, developed by Christensen and Pauly (1992a,b) from earlier work by Polovina (1984). The basic premise of the model is that, over the time period for which the model is relevant, total production is equivalent to total loss, that is steady-state⁴⁰. That is, for each group "i" in the model:

Production by (i) - all predation on (i) - non-predation losses of (i) - export of (i) = 0,

or,

$$B_i P/B_i - \sum_j B_j * Q/B_j * DC_{ji} - P/B_i * B_i (1 - EE_i) - EX_i = 0 \quad (1)$$

where,

B_i = biomass of (i), P/B_i = production/biomass of (i), EE_i = ecotrophic efficiency of (i), (that is the proportion of the production that is exported or consumed by the predators in the system.

It should be close to 1 for most groups⁴¹), Q/B_i = consumption biomass ratio of (i), $DC_{j,i}$ =

⁴⁰ ECOPATH is not restricted to the steady-state. The user can enter a biomass accumulation term, although the model must maintain a mass-balance.

⁴¹ In a model that contained more detail on the contribution of vegetation to the ecosystem, a primary producer such as reeds may contribute greatly to primary contribution, but not be consumed in the ecosystem. The EE would thus be close to 0, and detritus accumulation would occur as a result.

fraction of prey (i) in the average diet of predator (j), and EX_i = export of (i), (Christensen and Pauly 1992a).

The ecosystem is modelled using a set of simultaneous linear equations derived from the above relationship. Each group in the model is represented by one balanced equation.

$$B_1P/B_1EE_1 - B_1Q/B_1DC_{1,1} - B_2Q/B_2DC_{2,1} \dots B_nQ/B_nDC_{n,1} - EX_1 = 0$$

$$B_2P/B_2EE_2 - B_1Q/B_1DC_{1,2} - B_2Q/B_2DC_{2,2} \dots B_nQ/B_nDC_{n,2} - EX_2 = 0$$

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$$B_nP/B_nEE_n - B_1Q/B_1DC_{1,n} - B_2Q/B_2DC_{2,n} \dots B_nQ/B_nDC_{n,n} - EX_n = 0$$

This system of linear equations is solved using matrix algebra. The attractiveness of the mass-balance ECOPATH approach is that it is not necessary to know all the parameters for all the groups represented in the ecosystem. For each group in the model there are six input parameters. The export and the diet composition of each group must be entered. Of the four other parameters, B, P/B, Q/B and EE, three must be entered for each group⁴². Since the linear equations represent a balanced system, they can be solved for the unknown parameters.

The steady-state assumption of the model is justified in the following way. ECOPATH has no time dimension. The time period represented by the model is determined by the user. In this way,

it is assumed that there is no net difference between production and loss, over the time period for which the model is relevant (Christensen and Pauly 1993b). In the case of San Miguel Bay, the time period is 1 year.

There are three key steps to modelling with ECOPATH.

1. *Aggregate the total number of species in the ecosystem into representative groups.* These groups should reflect similarities in habitat, size and diet and importance in the fishery. Choose a representative species for each group, or take the mean of a number of representative species.
2. *Calculate the parameters P/B , Q/B , biomass, EE and diet for each group.* In exploited ecosystems, the catch is modelled as an export from the system and the annual catch per group must be entered. The units for the energy related parameters are in $t\ km^{-2}$. The area of San Miguel Bay is taken as $1115\ km^2$ (Garces *et al.* 1995b).
3. *Balance the model.* It is highly unlikely that an ECOPATH model will be perfectly balanced when the parameters are first estimated. Balancing an ECOPATH model requires enough knowledge about the ecosystem to make reasonable adjustments to the original parameters in order for the model run and balance.

⁴² Under certain conditions, the model will estimate more than one unknown parameter for group(i) if all the parameters are known for the other groups. See Christensen and Pauly (1992a) for further details.

ECOPATH produces a range of diagnostic statistics. These include gross efficiency (GE), which is calculated as the ratio of its production to consumption, for each group, respiration⁴³, mortality coefficients, a matrix of predation mortality coefficients, trophic levels, transfer efficiencies and a variety of cycle and pathway information. Some of this output can be readily used to check the validity of a model. For example the EE should not be greater than 1; GE should be between 0.1 and 0.3; there can not be negative flows to detritus and the respiration/biomass ratio (resp/biom) should be less than 100 (Christensen and Pauly 1992, V. Christensen, pers.comm).

Aggregating Species into ECOPATH Groups

More than 100 species of fish and crustacea were recorded in San Miguel Bay during the 1992-1994 trawl and landing surveys (Appendix 1). In order to make the ECOPATH model tractable, this complexity had to be reduced, by aggregating the species into groups. Following the guidelines in Sugihara *et al.* (1984), these species were aggregated into eco-groups, according to similarities in habitat, body size, diet and co-occurrence in fishing gear. ECOPATH models in Christensen and Pauly (1993a) were also consulted for comparative purposes.

Eleven eco-groups resulted from the aggregation, three of which are crustacean groups (Table 3.1). The division of some of the groups was straightforward, because of their high relative abundance in the trawl survey and their importance in the catch. The sergestids and penaeids, for example,

⁴³ Respiration is calculated in the Ecopath model and is the difference between assimilated food and production. It corresponds to the biological definition of respiration. If actual data are available, the model can be tuned to this data (Christensen and Pauly 1993b).

Table 3.1 Grouping of species found in San Miguel Bay for Ecopath Model.

| <i>Ecopath Group</i> | <i>Family/species</i> | <i>Relative Abundance</i> | <i>Ecopath Group</i> | <i>Family/species</i> | <i>Relative Abundance</i> |
|----------------------|--------------------------------|---------------------------|--------------------------|----------------------------------|---------------------------|
| Sergestid Shrimp | | | Pelagics | | 8.24 |
| Penaeid Shrimp | | 5.30 | | Clupeidae | |
| | | | | Dussumieria | |
| Large Crustaceans | | 0.80 | | Carangidae | |
| | Portunid Crabs | | | Chirocentridae | |
| | Stomatopods | | | Scombridae Loligo | |
| Demersal Feeders | | 16.86 | Sciaenids | | 7.02 |
| | Mullidae | | | <i>Otolithes ruber</i> | |
| | Nemipteridae | | | <i>Dendrophysa russelli</i> | |
| | Haemulidae | | | <i>Pennahia macrophthalmus</i> | |
| | Priacanthidae | | | Pennahia sp | |
| | Theraponidae | | | | 11.11 |
| | Tetraodontidae | | Medium Predators | | |
| | Bothidae | | | Arridae | |
| | Apogonidae | | | Synodontidae | |
| | Lethrinidae | | | Trichiuridae | |
| | Sparidae | | | Muranaesocidae | |
| | Silliganidae | | | Sphyrinaeidae | |
| | Gobiidae | | | Formionidae | |
| | Mugilidae | | | Muraenidae | |
| | Siganidae | | | Plotosidae | |
| | Cynoglossidae | | | Serranidae | |
| | Polynemidae | | | Psettodidae | |
| | Platycephalidae | | | Lutjanidae | |
| | Tricanthidae | | | Fistularidae | |
| | Soleidae | | | Opichthidae | |
| | Lactaridae | | | | 2.32 |
| Leiognathids | | 37.85 | Large Zoobenthos Feeders | | |
| | <i>L. splendens</i> | | | Rays | |
| | <i>L. elongatus</i> | | | Ephippidae | |
| | <i>L. equulus</i> | | | | 0.31 |
| | <i>L. bindus</i> | | Large Predators | | |
| | <i>Gazza minuta</i> | | | <i>Carcharhinus melanopterus</i> | |
| | <i>Secutor ruconius</i> | | | <i>Lates calcifer</i> | |
| | <i>S. insidiator</i> | | | | |
| Engraulids | | 6.13 | | | |
| | <i>Stolephorus commersonii</i> | | | | |
| | <i>Stolephorus indicus</i> | | | | |
| | <i>Thryssa setirostris</i> | | | | |

were clear groups, as were the leiognathids, engraulids and the sciaenids.

The large crustaceans group is comprised mostly of the portunid crabs *Portunus pelagicus* and *Scylla serrata*, but also includes stomatopods. *P. pelagicus* is most abundant and is taken as the representative species for the large crustaceans.

The demersal feeders is a more diverse group, representing 20 families, and almost 17% of the trawlable biomass. The most abundant families include the Mullidae, Tetraodontidae, Nemipteridae and Apogonidae. Others include the Mugilidae, Gobiidae and Siganidae.

The pelagics are another mixed group, and account for over 8% of the trawlable biomass⁴⁴. This group includes the Clupeidae, Carangidae, Scombridae and Squids (*Loligo* sp.) The latter were included in the pelagics, following Pauly (1985) and Silvestre *et al.* (1993). Pauly (1985) compared the growth performance of squid to fast growing scombrids.

The medium predators account for over 11% of the trawlable biomass and include 13 families. The most abundant are the Trichiuridae which comprise over 60% of the total group biomass and the Synodontidae which comprise 11% of the biomass. Other families include the Sphyrnidae, Muraenidae, Muranaesocidae and Psettodidae. It also includes, at much lower levels, the traditional predators of San Miguel Bay, the Ariidae and Serranidae.

⁴⁴ Since pelagic fish are not the main target of demersal trawls, this figure probably underestimates the actual relative abundance of the pelagics.

The large zoobenthos feeders are comprised of various species of rays and the Ehippidae. They represent 2% of the trawlable biomass. The large predators account for 0.3 % of the biomass. They consist of the shark, *Carcharinus melanopterus* and the Centropomid, *Lates calcarifer*.

In addition to the above groups, there are five other eco-groups in the ECOPATH model. These are the phytoplankton, zooplankton, meiobenthos, macrobenthos and detritus.

Parameterising the ECOPATH Model

Estimation of the Production Biomass ratio, P/B

The production biomass ration (P/B) is estimated in two ways. The first uses the assumption that the ratio of annual production to mean biomass (P/B) is equal to the annual instantaneous rate of total mortality, Z , under equilibrium conditions and assuming the von Bertalanffy growth function (Allen 1971). For the eco-groups for which it was not possible to estimate Z , a second method to estimate P/B was used. Values of P/B were taken from comparative tropical shallow marine ecosystems described in the literature, using the available Z estimates as a guide.

Estimating P/B from $P/B=Z$

There are three estimates of Z for the leiognathids (see Tables 2.7-2.9). Christensen and Pauly (1992a) recommend using a weighted mean to calculate combined parameters. This produced a P/B of 9.42 year^{-1} . There is only one Z estimate for the sciaenids, for *Otolithes ruber*. Since

this species represents 65% of the trawlable biomass of the sciaenids, it was taken as representative, giving a P/B of 4.39 year^{-1} . The medium predators are represented by the Z estimate of 2.5 year^{-1} for *Trichiurus haumela*. The results of the Z estimation for the pelagics were inconclusive (Table 2.10) and ranged from 1.04 to 5.26 to 9.64 year^{-1} . Their P/B value is estimated below.

Estimating P/B from other Ecosystems.

As a first step, the P/B estimates for the leiognathids, the sciaenids, the medium predators and the pelagics were compared to estimates of P/B for other ecosystems described in Christensen and Pauly (1993a) and Pauly and Christensen (1993). Comparisons were made between the San Miguel Bay estimates and estimates from the individual ecosystems, the mean of the various shelf and lagoon ecosystems and the mean of all ecosystems. The set of estimates that compared best with the San Miguel Bay figures were those from the Gulf of Thailand.

Although with an area of $300,000 \text{ km}^2$ (Pauly 1979a) the Gulf of Thailand is much larger than San Miguel Bay, the comparison with San Miguel Bay is reasonable. In addition to their similarity in geographic location, both water bodies have been intensively fished for several decades and have a similar fauna (Pauly 1979a, Pauly and Mines 1982). The comparison of estimates, and the final P/B values are given in Table 3.2.

Some of the groups in the Gulf of Thailand model differ from the San Miguel Bay eco-groups. The Gulf of Thailand intermediate predators include both the San Miguel Bay medium predators and the sciaenids. The latter are both compared with the intermediate predator

Table 3.2 P/B ratios from the Gulf of Thailand compared to the estimates from San Miguel Bay.

| <i>Gulf of Thailand Groups</i> | <i>Gulf of Thailand P/B (year⁻¹)</i> | <i>San Miguel Bay Z estimates (year⁻¹)</i> | <i>Final P/B (year⁻¹)</i> | <i>San Miguel Bay Groups</i> |
|--------------------------------|---|---|--|----------------------------------|
| Phytoplankton | 200 | — | 200 | Phytoplankton |
| Zooplankton | 40 | — | 67 | Zooplankton |
| Benthos | 6.85 | — | 10 | Meiobenthos |
| Molluscs | 5 | — | 6.8 | Macrobenthos |
| Microcrustaceans | 62 | — | 62 | Sergestids |
| Large Crustaceans | 6 | — | 6.48 | Penaeids |
| | — | — | 2.8 | Large Crustaceans |
| Demersal ZB feeders | 6 | — | 6 | Demersal Feeders |
| Leiognathidae | 6 | 7.5-11.3 | 9.42 | Leiognathids |
| | — | — | 6 | Engraulids |
| Small pelagics | 6 | — | — | |
| Loligo | 3.1 | 1.04-9.64 | 5.45 | Pelagics |
| Medium Pelagics | 4 | — | — | |
| Intermediate Predators | 4 | 2.5 | 2.5 | Medium Predators |
| | — | 4.4 | 4.4 | Sciaenids |
| Large ZB Feeders | 1.3 | — | 1.3 | Large ZB Feeders |
| Large Predators | 2 | — | 2 | Large Predators |

ZB = Zoobenthos

Data for the Gulf of Thailand from Pauly and Christensen (1993)

estimate. The pelagics from San Miguel Bay include small pelagics, the Loligo and the medium pelagics. These are treated separately in the Gulf of Thailand data, so the San Miguel Bay estimate was compared with each of the Gulf of Thailand P/B estimates and the weighted mean of the estimates. The San Miguel Bay pelagics P/B values were comparable to all the other model P/B estimates, because they cover such a large range.

On the basis of the Gulf of Thailand P/B figures, the P/B of the demersal feeders is 6 year^{-1} , the large Zoobenthos feeders, 1.3 year^{-1} and the large predators, 2 year^{-1} . However, due to some differences in the way that the species from San Miguel Bay and the Gulf of Thailand were grouped, the derivation of the other P/B values requires further explanation.

The Gulf of Thailand microcrustacea include sergestids and in the absence of other information, the microcrustacea P/B of 62 year^{-1} was taken to represent the sergestids of San Miguel Bay.

The large crustaceans group of San Miguel Bay is composed mostly of portunid crabs whereas the large crustacea of the Gulf of Thailand includes lobsters and shrimps in addition to crabs. The P/B estimate from Campeche Bank, Gulf of Mexico (Arreguín-Sánchez *et al.* 1993a), a tropical area also similar to San Miguel Bay, of 2.8 year^{-1} for portunid crabs was used to represent the large crustacea of San Miguel Bay.

There is no P/B estimate for the penaeids from the Gulf of Thailand because they are included in the large crustacea. However, there are two P/B estimates for penaeids in Christensen and

Pauly (1993a) from Campeche Bank, Gulf of Mexico (Arreguín-Sánchez *et al.* 1993a) and the southwestern Gulf of Mexico (Arreguín-Sánchez *et al.* 1993b). The mean of these two estimates, 6.48 year^{-1} , was taken as the P/B of the penaeids in San Miguel Bay.

The engraulids were not modelled as a separate group in the Gulf of Thailand data and there were no P/B estimates in the other available data. In this case, the Gulf of Thailand small pelagics P/B of 6 year^{-1} was used. This was considered valid since the engraulids are small pelagics and they are overfished in San Miguel Bay with exploitation ratios greater than 0.5 (Chapter 2, Pauly 1982a).

Since there was no reliable empirical estimate of Z for the pelagics, their P/B was also estimated from the Gulf of Thailand data. The Gulf of Thailand P/B values for the small pelagics and *Loligo* both fall within the range of Z estimates. The weighted average of these, 5.45 year^{-1} , was taken as the representative P/B value for the San Miguel Bay pelagics.

The P/B value of 200 year^{-1} for the phytoplankton and the zooplankton P/B of 67 year^{-1} were taken from Silvestre *et al.* (1993). The Gulf of Thailand P/B estimate for the benthos could not be used directly since the benthos in the San Miguel Bay model is divided into meio and macro benthos. The P/B estimate of 10 year^{-1} for the meiobenthos was taken from the only estimate available, from Celestun Lagoon in the Southern Gulf of Mexico (Chávez *et al.* 1993). The macrobenthos of San Miguel Bay is composed of 80.5% annelid worms, 3.8% molluscs (Garces *et al.* 1995b) and 15.7% of other matter, termed here, heterobenthos. The macrobenthos P/B was calculated from the weighted mean of the P/Bs for each of these

components from the Gulf of Thailand ecosystem The resultant weighted mean P/B was 6.8 year⁻¹, very similar to the benthos estimate of 6.85 for the Gulf of Thailand.

Estimation of Consumption Biomass Ratio, Q/B

The consumption biomass ratio is the quantity of food consumed by the biomass of fish in the population, within a given time period. It is estimated here in two ways. Where possible it is derived from the empirical formula described below. For the other eco-groups, comparative Q/B estimates were taken from the literature, as described for the P/B estimates.

Estimating Q/B from an empirical formula

Palomares and Pauly (1989) derived an empirical formula to estimate Q/B, expressed as a daily rate. The annual form of the equation, given in Christensen and Pauly (1992) is:

$$Q/B = 3.06 * W_{\infty}^{-0.2018} * T_c^{0.6121} * A_r^{0.5156} * 3.53^{H_d} \quad (2)$$

where,

W_{∞} = asymptotic weight (g)

T_c = mean habitat temperature (°C)

A_r = Aspect ratio = (height of caudal fin)²/surface area of caudal fin

H_d = food type (0=carnivorous, 1= herbivores+detritivores).

The novelty of this approach is the connection between the readily measurable physical proportions of the tail, and fish energetics. The Palomares and Pauly (1989) equation was derived from a regression analysis of 33 Q/B estimates. Pauly (1989) confirmed these results using a data set of 75 Q/B estimates. The regression model explained 75% of the variation in the data set and Ar, the aspect ratio, explained 50% of the total variance. Equation (2) provides a simple empirical formula to estimate Q/B, and requires only four, easy-to-estimate parameters.

The Q/B values used by the above authors in their regression analyses were estimated from an age-structured model, integrated over the life span of the group (Pauly 1986). It requires six parameters including estimates of growth rate and mortality, in addition to W_{∞} and other constants. However, both Palomares and Pauly (1989) and Pauly (1989), used natural mortality, M, to model the abundance of various age groups (D. Pauly, *pers. comm.*). Equation (2) is thus, strictly speaking, only applicable to unfished systems, where $Z=M$. The fishery analysis presented in Chapter 2 clearly demonstrated that San Miguel Bay is very far from an unfished system, and hence Q/B estimates derived from equation (2) will tend to underestimate Q/B⁴⁵.

This creates a problem because there is no alternative to using equation (2) to empirically estimate Q/B. There are insufficient data to calculate Q/B from the integral equation for more than four species, and thus avoid the $M=Z$ assumption. Instead, a three-step method was

⁴⁵ Since, if $Z>M$, there will be relatively more young fish in the population consuming more by unit weight than a population dominated by old fish.

developed to adjust Equation (2) to include the effects of fishing mortality and thus counter the $M=Z$ assumption.

1. Q/B was first estimated for the four species using the integration model of Pauly (1986)⁴⁶.

The growth and mortality results from the length-frequency analysis (Chapter 2) and length-weight parameters calculated from the length-weight data from the trawl survey data⁴⁷ were used.

2. Q/B was estimated from equation (2). The mean habitat temperature in San Miguel Bay during 1991-1992 was 29°C (Mendoza *et al.* 1995a). The aspect ratios were kindly provided by Mr. Francisco Torres Jr., ICLARM. Values of W_{∞} were calculated from the trawl survey data. For some species, values or mean values from the literature were used. In almost all cases, these were values from Philippine fisheries.
3. The two forms of Q/B estimate were compared for the four species. The mean ratio between them was used as a raising factor for the Q/B estimates of the other fished eco-groups.

Results of Q/B estimates

Estimates of Q/B from the integral equation were made for *Leiognathus splendens*, *L. bindus*, *Otolithes ruber* and *Trichiurus haumela*, representing the leiognathids, sciaenids, and medium

⁴⁶ Q/B was estimated using the ICLARM software program MAXIMS (Jarre *et al.* 1990).

⁴⁷ See also Cinco and Diaz 1995

predators respectively⁴⁸. Q/B estimates using Equation (2) were made for all fished groups apart from the large zoobenthos feeders and the large predators. The results are shown in Table 3.3.

In Table 3.4 the results of the Q/B estimates from the two methods are compared for the four species. The mean ratio between them is 1.63.

Q/B Estimates from the Literature

In order to obtain the Q/B estimates for the rest of the eco-groups in the San Miguel Bay ECOPATH model, comparable Q/B values in the literature were consulted. As for the P/B estimates, the Q/B estimates derived above⁴⁹ were compared to Q/B estimates from various tropical marine ecosystems in Christensen and Pauly (1993a) and Pauly and Christensen (1993). There was no one ecosystem to which all the San Miguel Bay Q/B estimates were comparable. For this reason, each eco-group was considered separately and where possible, estimates from southeast Asian coastal areas were used. The results are detailed in Table 3.5.

The Brunei Darussalam (Silvestre 1993) Q/B estimate of 280 year⁻¹ was used for the zooplankton. The meiobenthos Q/B estimate (50 year⁻¹) was taken from Celestun Lagoon, Mexico (Chávez *et al.* 1993), the same source as the P/B estimate. The macrobenthos Q/B of

⁴⁸ The Maxims program requires six parameters, W_{∞} , K , the VGBF growth parameter, "b" the exponent of the length/weight relationship, Z , total mortality, W_r , the smallest weight in the population and W_{max} , the maximum weight in the population and β , a constant. W_r was assumed to be 0.1 and W_{max} , the value recommended by the program (D. Pauly pers. comm.). β was calculated by first making the model fit the data to the Palomares and Pauly (1989) estimates of Q/B using M instead of Z . This was an iterative process.

⁴⁹ The Q/B estimates from Equation (2) were used for the comparison.

Table 3.3 Input parameters and Q/B estimates from (1) Pauly's Integral Equation and (2) Palomares and Pauly's Empirical Equation.

| ECOPATH Group | Representative Species | W_{∞} (g) | K (year ⁻¹) | Exponent 'b' of length/ weight relationship | Ar | Q/B (1) (year ⁻¹) | Q/B (2) (year ⁻¹) |
|------------------|--------------------------------|-----------------------|------------------------------|---|------|-------------------------------------|-------------------------------------|
| Demersal Feeders | <i>Upeneus sulphurus</i> | 145.3 ^{b,c} | — | 3.00 ^d | 2.72 | — | 14.7 |
| Leiognathids | <i>Leiognathus splendens</i> | 87.5 ^a | 1.0 ^a | 3.07 ^a | 2.06 | 25.2 | 14.2 |
| | <i>Leiognathus bindus</i> | 27.6 ^a | 1.2 ^a | 2.96 ^c | 2.45 | 27.6 | 19.5 |
| Engraulids | <i>Stolephorous commersoni</i> | 15.1 ^{b,f} | — | 3.19 ^a | 1.21 | — | 15.1 |
| | <i>Stolephorous indicus</i> | 40.5 ^{b,g} | — | 3.32 ^h | 1.56 | — | 14.3 |
| Pelagics | <i>Alepes djedaba</i> | 42.1 ^{b,i} | — | 2.76 ^j | 3.18 | — | 20.5 |
| Sciaenids | <i>Otolithes ruber</i> | 898.5 ^a | 0.40 ^a | 3.06 ^a | 0.85 | 11.3 | 5.6 |
| Medium Predators | <i>Arius thalassinus</i> | 4217.8 ^{b,k} | — | 3.02 ^l | 1.25 | — | 5.0 |
| | <i>Saurida tumbil</i> | 679.6 ^{b,m} | — | 3.02 ^a | 0.8 | — | 5.75 |
| | <i>Trichiurus haumela</i> | 760.6 ^a | 0.43 ^a | 2.66 ^a | — | 7.6 | — |

Ar = Aspect Ratios. Values provided by Mr. Francisco Torres Jr., ICLARM.

^a Parameter calculated from length frequency and length weight data from the 1992-1994 trawl survey of San Miguel Bay (see Chapter 2).

^b Calculated from L_{∞} and length-weight relationship, using L_{∞} and/or length-weight parameters from the literature.

^c L_{∞} an average of values from the Philippines, Armada and Silvestre (1980), Corpuz *et. al* (1985), Silvestre (1986), Sambilay (1991) and Frederizon (1993).

^d Frederizon (1993) Philippines

^e Murty (1983). India

^f Ingles and Pauly (1984) Philippines

^g L_{∞} an average of three estimates from the Philippines, Sambilay (1991), Corpuz *et. al* (1985) and Padilla (1991).

^h Cinco (1982)

ⁱ Corpuz *et. al* (1985)

^j Cinco (1982)

^k L_{∞} an average of values from Menon (1986) India, Wahyuono and Budiardjo (1985) Indonesia, Bawazeer (1987) Kuwait and Dwiponggo *et. al* (1986) Indonesia.

^l Bawazeer (1987)

^m L_{∞} an average of two estimates from Manila Bay, the Philippines, Tiews *et. al* (1972b), Ingles and Pauly (1984).

Table 3.4 Results of the Pauly (1986) Q/B estimation method compared to the Palomares and Pauly (1989) regression equation.

| <i>Species</i> | <i>Q/B from Pauly (1986) integral equation (year⁻¹)</i> | <i>Q/B from Palomares and Pauly (1989) (year⁻¹)</i> | <i>Ratio between the two Q/B estimates</i> |
|------------------------------|--|--|--|
| <i>Leiognathus splendens</i> | 25.2 | 14.2 | 1.77 |
| <i>Leiognathus bindus</i> | 27.6 | 19.5 | 1.41 |
| <i>Otolithes ruber</i> | 11.3 | 5.6 | 2.02 |
| <i>Trichiurus haumela</i> | 7.6 | 5.7* | 1.33 |
| Mean Ratio | — | — | 1.63 |

* This is the weighted mean of the Q/B estimates for *Arius thalassinus* and *Saurida tumbil*.

Table 3.5 The final Q/B estimates corrected for fishing mortality.

| <i>Ecopath Group</i> | <i>Q/B (year⁻¹)</i> | <i>Final Q/B (year⁻¹)</i> |
|------------------------------|------------------------------------|--|
| Zooplankton | 192 | 192 |
| Meiobenthos | 50 | 50 |
| Macrobenthos | 25.9 | 25.9 |
| Sergestid Shrimp | 310 | 506 |
| Penaeid Shrimp | 19.2 | 31.4 |
| Large Crustaceans | 8.5 | 13.9 |
| Demersal Feeders | 14.7 | 24.5 |
| Leiognathids | — | 26* |
| <i>Leiognathus splendens</i> | 14.2 | 25.2 |
| <i>Leiognathus bindus</i> | 19.5 | 27.6 |
| Engraulids | 14.7 | 24 |
| Pelagics | 17.9 | 28.9 |
| Sciaenids | 5.6 | 11.3 |
| Medium Predators | 5.7 | 7.6 |
| Large Zoobenthos Feeders | 8.2 | 11.7 |
| Large Predators | 8.4 | 11.9 |

Values in italics are those that are changed in the final analysis.

* This is the weighted mean of the two leiognathid species.

25.9 year⁻¹ is the weighted mean of the Q/B estimates for annelid worms, molluscs and “other” benthos from the Gulf of Thailand and Brunei Darussalam.

There was no Q/B estimate for the sergestid shrimp from the Gulf of Thailand so the estimate of 310 year⁻¹ for the small crustacea groups from the Brunei Darussalam ecosystem (Silvestre 1993) was used. The Q/B estimate of 8.5 year⁻¹ for the large crustaceans group was taken from the Campeche Bank, Gulf of Mexico (Arreguín-Sánchez *et al.* 1993a), the same source as the P/B estimate. The penaeid shrimp Q/B was also taken from the Campeche Bank ecosystem.

The Gulf of Thailand Q/B estimates were used for the large zoobenthos feeders and the large predators.

Q/B was estimated above for the pelagics representative, *Alepes djedaba* using the empirical formula. However, *A. djedaba* represents only 25% of the pelagic biomass. In order to obtain a more representative Q/B estimate for the pelagics group, literature values were also consulted. The following Q/B values were assumed for three other members of the pelagics⁵⁰, Loligo, Q/B=25 year⁻¹, (Pauly and Christensen 1993); small pelagics (Clupeids and Dussumierids), Q/B=17.9 year⁻¹; (Pauly and Christensen 1993) and other Carangids, mean Q/B=9.2 year⁻¹ (Arreguín-Sánchez 1993b, Mendoza 1993). The weighted mean Q/B of these values and the Q/B of *A. djedaba* is 17.9 year⁻¹.

⁵⁰ Although 41% of the pelagics biomass is *Scomberomorus commerson*, only juveniles are found in the Bay (L_{\max} reported in this study was 37 cm (TL), Pauly (1982b)). Since Q/B is the mean consumption biomass ratio over the life of a fish, *S. commerson* cannot be used to represent the group.

The Final Q/B Estimates, Corrected for Fishing Mortality.

The initial and final Q/B estimates are listed in Table 3.5. Both the empirically derived and the literature derived values were raised by 1.63 to allow for the effects of fishing.

Estimation of Biomass

The biomass of each eco-group is estimated from the biomass estimate in Chapter 2. The calculation of biomass is made on the basis of the relative abundance of each eco-group in the trawlable biomass (Table 3.1). However, biomass could only be calculated for those groups which appear in the trawl survey. For the other groups, except the detritus, no estimate was made.

Estimation of Detritus Biomass

The detritus biomass was estimated using an empirical relationship derived by Pauly *et al.* (1993). It relates detritus biomass to primary productivity and euphotic depth.

$$\log_{10} D = -2.41 + 0.954 \log_{10} PP + 0.863 \log_{10} E \quad (3)$$

where,

D = detritus standing stock (gCm^{-2} (grams of Carbon per square metre)), PP = primary productivity ($\text{gCm}^{-2} \text{ year}^{-1}$), E = euphotic depth (m).

The fit of the regression equation to the data is not very good, but as suggested by Pauly *et al.*, it “might be considered sufficient in cases where no other information is available” (1993:13).

Ricafrente-Remoto and Mendoza (1995) made three estimates of primary production in San Miguel Bay using the ‘light-dark bottle technique’, nutrient data and chlorophyll ‘a’ data. Their results were quite variable. The results using the nutrient data gave an average annual value of $391 \text{ gCm}^{-2} \text{ year}^{-1}$, which is comparable to values in the literature.⁵¹ This value was assumed to represent the average annual primary production in San Miguel Bay.

The euphotic depth is calculated from the Beer-Bouger Law where,

$$\ln I(1) - \ln I(2) = k (D(2)-D(1))$$

and,

$I(1) = 100\%$ irradiance (at the surface), $I(2) = 1\%$ irradiance (at the euphotic depth), $D(1) =$ depth at surface (0m), $D(2) =$ euphotic depth, $k =$ light attenuation co-efficient.

‘k’ is calculated from the relationship $k = 1.45/D_s$ (Walker 1980), where D_s is the secchi depth. An average secchi depth of 2.4 m was estimated from Figure 3 in Mendoza *et al.* (1995a). The

⁵¹ The “light-dark bottle technique” produced an average annual PP of $2833.14 \text{ gCm}^{-2} \text{ year}^{-1}$, and the chlorophyll ‘a’ method an average annual PP of $162.57 \text{ gCm}^{-2} \text{ year}^{-1}$.

range was 0 – 10 m. Solving the Beer-Bouger Law for $D(2)$ produces an average annual euphotic depth of 1.92 m.

Substituting the primary production value of $391 \text{ gCm}^{-2} \text{ year}^{-1}$, and the euphotic depth of 1.92 m into equation (3) produced a detritus biomass of 1.98 gCm^{-2} . This translates into 19.8 tkm^{-2} using a conversion factor of $10 \text{ g wet weight} = 1 \text{ gC}$ (as suggested by Christensen and Pauly 1992a:20).

Export (Catch)

The catch of each eco-group was calculated from the catch figures in Chapter 2 and expressed as tkm^{-2} , taking the area of San Miguel Bay as 1115 km^2 (Table 3.6).

Ecotrophic Efficiency

The ecotrophic efficiency is a measure of productivity that is not “other mortality”. In other words, ecotrophic efficiency is the proportion of the production that is exported or consumed by predators in the ecosystem. It has no units and is difficult to measure (Christensen and Pauly 1992a). Ecotrophic efficiencies of 0.95 are commonly used as approximations (Christensen and Pauly 1992a, 1993, Polovina 1984). This procedure was adopted here.

Input values for the parameters P/B, Q/B, biomass, export and ecotrophic efficiency are given in Table 3.6.

Table 3.6 Input parameters for the ECOPATH model of San Miguel Bay.

| <i>Ecopath Group</i> | <i>P/B</i> (year ⁻¹) | <i>Q/B</i> (year ⁻¹) | <i>Biomass</i> (t km ⁻²) | <i>Export/Catch</i> (t km ⁻²) | <i>EE</i> |
|--------------------------|-------------------------------------|-------------------------------------|---|--|-----------|
| Phytoplankton | 200 | — | — | — | 0.95 |
| Zooplankton | 67 | 192 | — | — | 0.95 |
| Meiobenthos | 10 | 50 | — | — | 0.95 |
| Macrobenthos | 6.8 | 25.9 | 0.107 | 0.036 | 0.95 |
| Sergestid Shrimp | 62 | 506 | — | 2.403 | 0.95 |
| Penaeid Shrimp | 6.48 | 31.4 | 0.144 | 1.677 | 0.95 |
| Large Crustaceans | 2.8 | 13.9 | 0.022 | 0.854 | 0.95 |
| Demersal Feeders | 6 | 24.5 | 0.458 | 1.273 | 0.95 |
| Leiognathids | 9.42 | 26 | 1.03 | 1.434 | 0.95 |
| Engraulids | 6 | 24 | 0.167 | 1.071 | 0.95 |
| Pelagics | 5.45 | 28.9 | 0.224 | 1.149 | 0.95 |
| Sciaenids | 4.39 | 11.3 | 0.191 | 3.388 | 0.95 |
| Medium Predators | 2.5 | 7.6 | 0.302 | 1.090 | 0.95 |
| Large Zoobenthos Feeders | 1.3 | 11.7 | 0.063 | 0.309 | 0.95 |
| Large Predators | 2 | 11.9 | 0.008 | 0.131 | 0.95 |
| Detritus | — | — | 19.8 | — | — |
| | — | — | — | — | — |

Diet Composition

Empirical data from diet studies conducted in San Miguel Bay exist for five of the 15 eco-groups represented in the ECOPATH model. For the other eco-groups, information from the literature was used to estimate diet composition. The main source of information were diet compositions from other ECOPATH models in Christensen and Pauly (1993a), although other sources were also used. Of the ECOPATH models in the literature, two are derived from similar habitats to San Miguel Bay, that is the models for the Gulf of Thailand (Pauly and Christensen 1993) and Brunei Darussalam (Silvestre *et al.* 1993). The latter is based on an earlier model from Malaysia (Liew and Chan 1987). All three models are from a similar latitude and longitude to San Miguel Bay and are also shallow. Where possible, the diet compositions from these two models were used to represent the diet of the eco-groups in San Miguel Bay. The resultant diet matrix is shown in Table 3.7, and the details of the derivation for each group are given below. Diet composition is measured by weight⁵².

Zooplankton

The Gulf of Thailand and Brunei Darrussalam models had similar zooplankton diets comprising phytoplankton (70% and 65% respectively), 30% and 25% detritus and, in Brunei

⁵² MacDonald and Green (1983) discuss three methods by which stomach contents can be measured, % occurrence of food items, numerical abundance of food items and the weight or volume of food items. Their analysis demonstrated that the three measures are highly correlated on the first principal component of a PCA, indicating that there is some redundancy. This results indicates that it is not necessary to take all three measurements in order to accurately measure stomach content. However, the three measures do measure different things. Percent occurrence measures the variability in predator diets. Numerical abundance gives information on the density dependent prey eating behaviour of predators and can therefore give insight into feeding behaviour. The weight or volume of food items measures the nutritional value of a prey species.

Darussalam, 10% zooplankton. Since there was no basis for choosing any one set of values, the values were averaged over the two systems.

Meiobenthos

The only diet composition data found for meiobenthos are from a lagoon in Veracruz, Mexico (Cruz-Aguero 1993). In the absence of other data, these values were used for the meiobenthos in San Miguel Bay.

Macrobenthos

The macrobenthos in San Miguel Bay is dominated by annelid worms (80.5%). The other components are 3.8% molluscs and 15.7% "others" (Garces *et al.* 1995). Data on diet composition for annelid worms and molluscs were taken from other ECOPATH models (Arreguín-Sánchez *et al.* (1993a), Arreguín-Sánchez *et al.* (1993b) Chávez *et al.* (1993) and Vega-Cendejas *et al.* (1993)) since the two models noted above did not have separate data on these groups. Data for the "others" were assumed to be represented by the heterobenthos groups in the Gulf of Thailand and Brunei Darussalam. For each group, the average diet composition across models was calculated. A weighted average, based on the relative abundance of each group in the macrobenthos, was taken as the final representative diet composition for the macrobenthos.

Sergestid Shrimp

Sergestids are small pelagic crustaceans, essentially zooplankton (Omori 1975) . However, there is no information on their dietary preferences. In all likelihood they will feed on small herbivorous zooplankton, phytoplankton and detritus. Thus the sergestid shrimp diet is a "guesstimate" of 40% zooplankton, 50% phytoplankton and 10% detritus.

Penaeid Shrimp

Tiews (1976) provides information on the diet of four penaeid species in San Miguel Bay (*P. semisculatus*, *P. merguensis*, *P. canaliculatus* and *Metapenaeus monocerus*). Unfortunately these data were recorded as % occurrence in the stomachs and cannot be used directly since the diet composition is measured in % weight. However, they can be used as a guide when using quantitative data from other systems. Data from six ECOPATH models were used (Abarca-Arenas and Valero-Pacheco (1993), Arreguín-Sánchez *et al.* (1993a), Arreguín-Sánchez *et al.* (1993b) Chávez *et al.* (1993), Cruz-Aguero (1993) and Vega-Cendejas *et al.* (1993)) and the mean diet composition compared to Tiews *et al.*'s figures. The resultant diet composition is given in Table 3.7 The same food groups appeared in both data sets, although in Tiew *et al.*'s data, there was less emphasis on the detritus and more on meiobenthos. This could be due to the different methods of measurement.

Large Crustaceans

The large crustacea are composed almost entirely of the swimming crab *Portunus pelagicus*. Diet data from two Mexican systems for crabs of the *Callinectes* sp. were used (Arreguín-Sánchez *et al.* (1993a), Arreguín-Sánchez *et al.* (1993b)), plus the large crustacea data from Brunei Darussalam and the Gulf of Thailand. The resultant mean data were compared to some data from Edgar (1990) and Wassenberg and Hill (1987) who describe the diet of *P. pelagicus* as consisting mainly of benthic invertebrates such as bivalves, polychaetes and crustaceans. The diet in Table 3.7 reflects this preference.

Demersal Feeders

This group is composed of 20 fish families, 80% of which are the Mullidae, Nemipteridae, Tetraodontidae, Gobiidae and Apogonidae. In order to account for this diversity and the fact that there was no direct diet information from San Miguel Bay for any of these species, diet information for similar groupings of demersal feeders from the Gulf of Thailand and Brunei Darussalam ECOPATH models were used in combination with diet data for some of the individual families from other ECOPATH models (Abarca-Arenas and Valero-Pacheco (1993), Arreguín-Sánchez *et al.* (1993a), Arreguín-Sánchez *et al.* (1993b) Chávez *et al.* (1993), Cruz-Aguero (1993), Mendoza (1993) and Vega-Cendejas *et al.* (1993)). The resultant diet composition shown in Table 3.7 is based on the weighted mean diet of the different families of demersal feeders.

Table 3.7 Diet Composition for the ECOPATH model. Figures in brackets were changed during the balancing process, figures in bold are the new values.

| <i>Ecopath Group</i> | <i>ZP</i> | <i>Meiob</i> | <i>Macrob</i> | <i>Serg</i> | <i>Pen</i> | <i>LC</i> | <i>DF</i> | <i>Leiog</i> |
|----------------------|-----------|--------------|---------------|-------------|------------|-----------------------|-----------|--------------|
| Zooplankton | 0.05 | | | 0.40 | 0.05 | (0.10) 0.04 | 0.10 | 0.275 |
| Meiobenthos | | 0.1 | 0.05 | | 0.10 | 0.13 | 0.05 | 0.050 |
| Macrobenthos | | | 0.05 | | 0.40 | (0.30) 0.38 | 0.45 | 0.425 |
| Sergestid | | | | | | (0.15) | 0.10 | 0.150 |
| Shrimp | | | | | | 0.04 | | |
| Penaeid | | | | | | (0.05) | 0.05 | |
| Shrimp | | | | | | 0.04 | | |
| Large | | | | | | (-) | | |
| Crustaceans | | | | | | 0.01 | | |
| Demersal | | | | | | | | |
| Feeders | | | | | | | | |
| Leiognathids | | | | | | | | |
| Engraulids | | | | | | | | |
| Pelagics | | | | | | | | |
| Sciaenids | | | | | | | | |
| Medium | | | | | | | | |
| Predators | | | | | | | | |
| Large | | | | | | | | |
| Zoobenthos | | | | | | | | |
| Feeders | | | | | | | | |
| Large | | | | | | | | |
| Predators | | | | | | | | |
| Phytoplankton | 0.70 | | 0.05 | 0.50 | 0.05 | (-) 0.00 | | 0.050 |
| Detritus | 0.25 | 0.9 | 0.85 | 0.10 | 0.40 | (0.40) 0.36 | 0.25 | 0.050 |

ZP = Zooplankton, Meiob = Meiobenthos, Macrob = Macrobenthos, Serg = Sergestids, Pen = Penaeids, LC = Large Crustaceans, DF = Demersal Feeders, Leiog = Leiognathids,

Some values are rounded to two figures, thus not all columns add up to 1.

Table 3.7 (cont.) Diet Composition for the ECOPATH model. Figures in brackets were changed during the balancing process, figures in bold are the new values.

| <i>Ecopath Group</i> | <i>Eng</i> | <i>Pel</i> | <i>Sci</i> | <i>MP</i> | <i>LZB</i> | <i>LP</i> |
|----------------------|------------|--------------------|------------|-----------------------|------------|-----------------------|
| Zooplankton | 0.40 | 0.50 | 0.10 | (0.10) 0.09 | | (0.10) — |
| Meiobenthos | | | 0.05 | | | |
| Macrobenthos | 0.15 | 0.10 | 0.30 | (0.19) 0.14 | 0.9 | (0.10) 0.04 |
| Sergestid | 0.25 | 0.05 | 0.25 | 0.15 | | |
| Shrimp | | | | | | |
| Penaeid | 0.15 | 0.1 | 0.15 | (0.10) 0.09 | 0.05 | |
| Shrimp | | | | | | |
| Large | | 0.05 | 0.10 | | | |
| Crustaceans | | | | | | |
| Demersal | | | | 0.10 | | (0.15) |
| Feeders | | | | | | 0.13 |
| Leiognathids | | | | (0.11) 0.18 | | (0.10) 0.14 |
| Engraulids | | (0.20) — | 0.03 | 0.10 | | (0.15) 0.16 |
| Pelagics | | | | 0.10 | | (0.15) 0.20 |
| Sciaenids | | | 0.02 | (0.05) 0.03 | | (0.15) 0.20 |
| Medium | | | | (—) | | (0.10) |
| Predators | | | | 0.03 | | 0.13 |
| Large | | | | | | |
| Zoobenthos | | | | | | |
| Feeders | | | | | | |
| Large | | | | | | |
| Predators | | | | | | |
| Phytoplankton | | (—) 0.20 | | | | |
| Detritus | 0.05 | | | | 0.05 | |

Eng = Engraulids, Pel = Pelagics, Sci = Sciaenids, MP = Medium Predators, LZB = Large Zoobenthos Feeders, LP = Large Predators.

Leiognathids

Data from Palomares *et al.* (1995b) indicate that crustaceans formed 46% of the diet of *Leiognathus bindus*, but no other food items were identified. There is data on the diet composition of leiognathids from Tiews *et al.* (1972a) which is measured in % occurrence. These data for *L. splendens*, *L. bindus*, *S. ruconius* and *S. insidiator* were compared to diet data for leiognathids from the Gulf of Thailand and the Brunei Darussalam ECOPATH models. The diet data recorded by Tiews *et al.* are qualitatively comparable to the diet data in the two ECOPATH models. The relative proportions of the ECOPATH models were used and the resultant diet is given in Table 3.7.

Engraulids

There are some data on the diet compositions of *Stolephorus commersonnii* and *S. indicus* from San Miguel Bay (Palomares *et al.* 1995b) which indicates that crustaceans and plankton are an important part of their diet. However, these data are not detailed enough to quantify the diet of the engraulids. Diet data were taken from Abarca-Arenas and Valero-Pacheco (1993) and Arreguín-Sánchez *et al.* (1993b). About 40% of the diet were crustaceans which fits with the San Miguel Bay data.

Pelagics

The pelagics represent six families. Since there were no empirical diet data, data for the clupeids, carangids, scombrids and loligo were taken from several ECOPATH models (Arreguín-Sánchez *et al.* (1993a), Arreguín-Sánchez *et al.* (1993b), Cruz-Aguero (1993) and Mendoza (1993)). A weighted average was taken of these groups plus an “others” groups representing the rest of the pelagics. Data for the “others” group were taken from the pelagics in the Gulf of Thailand and the Brunei Darussalam ECOPATH models.

Sciaenids

The most important food item in the diet of *Otolithes ruber* is crustaceans, followed by “other”, then fish (Palomares *et al.* 1995b). This compares well with the fish and crustacean components of the diet composition of sciaenids from Venezuela (Mendoza 1993). It was assumed that the Venezuelan sciaenid diet is representative of the San Miguel Bay sciaenid diet.

Medium Predators

The medium predators are another diverse group, although over 60% of the biomass is made up of *Trichiurus haumela* (Table 3.1). However, there were no empirical diet data for any of the families represented by the medium predators. Instead, diet data for the Synodontidae and the Ariidae were taken from ECOPATH models (Arreguín-Sánchez *et al.* (1993b), Cruz-

Aguero (1993), Mendoza (1993)). The medium predators diet was then taken as the weighted mean of these data plus the medium predator data from the Gulf of Thailand and the Brunei Darussalam ECOPATH models.

Large Zoobenthos Feeders and Large Predators

In the absence of other data, the diets of the large zoobenthos feeders and large predators were taken as the average of the diet compositions described in the Gulf of Thailand and the Brunei Darussalam ECOPATH models.

The ECOPATH Parameters

It is recognised that some of the parameters that have been determined for the ECOPATH model are very approximate. Still the approach taken here is justified for three reasons. First, Hoenig *et al.* (1987:331), writing on the use of indirect rapid assessment and the use of comparative studies, conclude that comparative studies can “lead to better theoretical models and understanding of the systems by suggesting structural relationships which require explanation”. Secondly, Sugihara *et al.* (1984:139) suggest that “it is possible that coarse allometric estimates of growth/mortality parameters may be more appropriate for certain (large-scale) models than are data obtained directly from individuals”. They are suggesting that coarse, but representative measures of parameters for aggregated groups may be more suitable than precisely defined parameters for single species. This approach was adopted here for the more heterogeneous eco-groups. Thirdly, if approximate parameter values were not estimated

or taken from the literature, it would not be possible to even attempt an ecosystem model of San Miguel Bay. Thus, instead of probing and modelling, learning and gaining knowledge, with assumptions in full view, no knowledge would be gained nor new hypotheses formulated.

The final parameters which were used in the first run of the ECOPATH model are given in Table 3.6 above.

Running and Balancing the ECOPATH Model

It was noted that ECOPATH models rarely, if ever, balance when first run with derived parameters. This was indeed the case here. The initial problem with the San Miguel Bay ECOPATH model concerned the biomass estimates. The model was first run using the biomass estimates from the trawl survey and the P/B and Q/B estimates. ECOPATH recalculates the EE when the other three parameters are entered. Some resultant EEs were above 1, and thus there were also negative flows to the detritus, GEs above 0.3, and some respiration/biomass ratios were too high. Essentially, the biomass was too low to support the catches being removed from the ecosystem.

This problem was first addressed by entering the EE estimates for all groups apart from the detritus, large predators, medium predators and large zoobenthos feeders and allowing the model to estimate biomass. This approach was more successful and corrected many of the problems noted above. One consequence is that the estimated total fished biomass is now three

to four times higher than the biomass estimated from the trawl survey in Chapter 2. The approach was refined to the point where only the biomass of the large predators was entered. Through an iteration method, the large predators biomass was reduced as much as possible in order to minimise the total fished biomass. Total biomass is reduced when the biomass of the top predator is reduced because ECOPATH is a top-down modeling method and all flows are scaled to the biomass levels at the top of the food web. Christensen and Pauly (mimeo, Marine Ecosystem Management: An ode to Odum) used this method to simulate climax ecosystems by increasing the biomass of top predators, thereby causing increases in biomass at lower levels of the food web⁵³.

The fitting criteria noted above were used to constrain the model. This entailed several minor changes in the input parameters, including the diet matrix. The P/B ratios of the leiognathids and medium predators were decreased in order to reduce their GEs below 0.3. Both new values remain within the confidence limits of the Length Converted Catch Curve estimates of Z in Chapter 2⁵⁴. The P/B of the pelagics was increased in order to increase the GE above 0.2. This was considered reasonable since the Z estimate in Chapter 2 was highly uncertain, and the value of 5.45 year⁻¹ in Table 3.6 is a composite value.

The Q/Bs of the sergestids was decreased by almost 50% in order to reduce their respiration/biomass ratio below 100. This value approximates the Q/B value of the

⁵³ The authors found that when climax ecosystems were simulated, the ecosystem took on many of the properties predicted by Odum's theory of ecosystem development (1969), particularly the retention and recycling of detritus.

⁵⁴ For the leiognathids, the new P/B value of 7.5 year⁻¹ is within the confidence limits of the Length Converted Catch Curve estimate of Z for *Secutor ruconius* and *Leiognathus bindus*. However, it only falls within the confidence limits of one of the Z estimates for *L. splendens* (Table 2.7).

zooplankton of 280 year^{-1} . The latter was reduced to the same value as the sergestids, since both are zooplanktivores⁵⁵. The Q/B of the sciaenids was also changed. In this case, Q/B was changed because the value of GE was too high. GE was set to 0.27, the P/B and EE values were entered and the model allowed to estimate Q/B⁵⁶. This produced a Q/B estimate of 16.3 year^{-1} .

Small changes to the diet of the pelagics, medium predators and large predators were made because their predation mortality on some of the lower eco-groups was too high. Changes were also made to the large crustaceans diet⁵⁷. The new diet composition is shown (in bold) in Table 3.7.

The balancing of the model was performed manually, iteratively changing parameters in order to better fit the model. The latest version of ECOPATH, ECOPATH 3.0 contains a routine which allows the user to input a range of values and ascribe probability ranges to them. ECOPATH 3.0 then fits the parameters using various criteria of best fit (ICLARM 1995). The above parameters were tested using this routine, but essentially, these parameters are the “best fit”⁵⁸.

⁵⁵ The proportion of food intake which is not assimilated was also increased from the default value of 0.2 to 0.4, for these two eco-groups, following Christensen (1994) and Christensen and Pauly (1992a), in order to reduce the resp/biomass ratio.

⁵⁶ The model was made to estimate Q/B, instead of P/B, because the latter was considered a more robust parameter since it was directly estimated from length-frequency data in Chapter 2.

⁵⁷ The changes to the large crustaceans diet were made in response to the testing of a prototype ECOSIM model (see Chapter 4), and on the basis of diet information in Wassenberg and Hill (1987).

⁵⁸ Slightly different fits were produced, but included parameters which resulted in GEs above 0.3 and respiration/biomass ratios above 100.

Results

Some Basic Results

Some of the ECOPATH results are shown in Table 3.8. Figure 3.1 is a box model which shows the relation of the eco-groups to one another, their trophic position⁵⁹ and the nature and strength of the flows between them. The ecosystem so described is one which is more dependent on its detritus-benthic components than the pelagic components of the ecosystem. Of the total flow in the ecosystem, 56% originates from the detritus and 44% from primary producers.

The ecosystem spans more than four trophic levels. The large predators occupy a trophic level of 4.1 and the fishery operates at an average trophic level of 4.0. Thus the large predators occupy a higher trophic level than the fishery. Most of the eco-groups however occur at a trophic level of between 2.5 and 3.5. Of these, the engraulids and pelagics are mainly dependent on the pelagic strands of the food web. The others, the sciaenids, leiognathids, demersal feeders, large zoobenthos feeders, large crustaceans and penaeids rely on the detritus food chains. Given the large number of groups occurring at a similar trophic level, there is likely to be strong competitive interactions between groups, in addition to predator/prey relationships.

⁵⁹ ECOPATH calculates trophic levels in two ways (Christensen and Pauly 1992a). In Figure 3.1, fractional trophic levels are used. Thus, the trophic level of each eco-group is calculated as the weighted average of its prey's trophic level. Producers and detritus are assigned a trophic level of 1. ECOPATH also calculates discrete trophic levels for the entire system using an aggregation routine - see note 63 below.

Table 3.8 Selected results from the San Miguel Bay ECOPATH model.

| | <i>B</i> (<i>tkm⁻²</i>) | <i>P/B</i> (<i>year⁻¹</i>) | <i>Q/B</i> (<i>year⁻¹</i>) | <i>EE</i> | <i>GE</i> | <i>Harvest</i> (<i>tkm⁻²</i> <i>year⁻¹</i>) | <i>Trophic</i> <i>Level</i> | <i>Flow to</i> <i>Detritus</i> (<i>tkm⁻²</i> <i>year⁻¹</i>) | <i>Omnivory</i> <i>Index</i> | <i>Resp/</i> <i>Biomass</i> |
|--------------|---|--|--|-----------|-----------|---|--------------------------------|--|---------------------------------|--------------------------------|
| 1. ZP | 0.757 | 67 | 260 | 0.95 | 0.258 | 0 | 2.05 | 81.237 | 0.053 | 89 |
| 2. Meiob | 3.973 | 10 | 50 | 0.95 | 0.2 | 0 | 2.11 | 41.718 | 0.111 | 30 |
| 3. Macrob | 7.786 | 6.8 | 25.9 | 0.95 | 0.263 | 0.036 | 2.11 | 42.98 | 0.111 | 13.92 |
| 4. Serg | 0.215 | 62 | 260 | 0.95 | 0.238 | 2.403 | 2.42 | 23.008 | 0.266 | 94 |
| 5. Pen | 1.286 | 6.48 | 31.4 | 0.95 | 0.206 | 1.677 | 2.60 | 8.495 | 0.303 | 18.64 |
| 6. LC | 1.154 | 2.8 | 13.9 | 0.95 | 0.201 | 0.854 | 2.75 | 3.371 | 0.332 | 8.32 |
| 7. DF | 0.336 | 6 | 24.5 | 0.95 | 0.245 | 1.273 | 2.88 | 1.747 | 0.279 | 13.6 |
| 8. Leiogs | 0.348 | 7.5 | 26 | 0.95 | 0.288 | 1.434 | 3.03 | 1.943 | 0.132 | 13.3 |
| 9. Eng | 0.39 | 6 | 24 | 0.95 | 0.25 | 1.071 | 3.18 | 1.987 | 0.119 | 13.2 |
| 10. Pel | 0.296 | 6.65 | 28.9 | 0.95 | 0.23 | 1.149 | 2.96 | 1.809 | 0.274 | 16.47 |
| 11. Sci | 0.972 | 4.39 | 16.3 | 0.95 | 0.27 | 3.388 | 3.38 | 3.374 | 0.097 | 8.617 |
| 12. MP | 0.651 | 2.2 | 7.6 | 0.95 | 0.289 | 1.09 | 3.70 | 1.061 | 0.188 | 3.88 |
| 13. LZB | 0.25 | 1.3 | 11.7 | 0.95 | 0.111 | 0.309 | 3.08 | 0.602 | 0.073 | 8.06 |
| 14. LP | 0.095 | 2 | 11.9 | 0.689 | 0.168 | 0.131 | 4.14 | 0.285 | 0.112 | 7.52 |
| 15. Ph/P | 12.825 | 70 | 0 | 0.2 | - | 0 | 1 | 717.795 | 0 | - |
| 16. Detritus | 19.8 | - | - | 0.462 | - | 0 | 1 | - | 0.283 | - |

Figures in italics are estimated by ECOPATH.

ZP = Zooplankton, Meiob = Meiobenthos, Macrob = Macrobenthos, Serg = Sergestids, Pen = Penaeids, LC = Large Crustaceans, DF = Demersal Feeders, Leiog = Leiognathids, Eng = Engraulids, Pel = Pelagics, Sci = Sciaenids, MP = Medium Predators, LZB = Large Zoobenthos Feeders, LP = Large Predators, Ph/P = Phytoplankton.

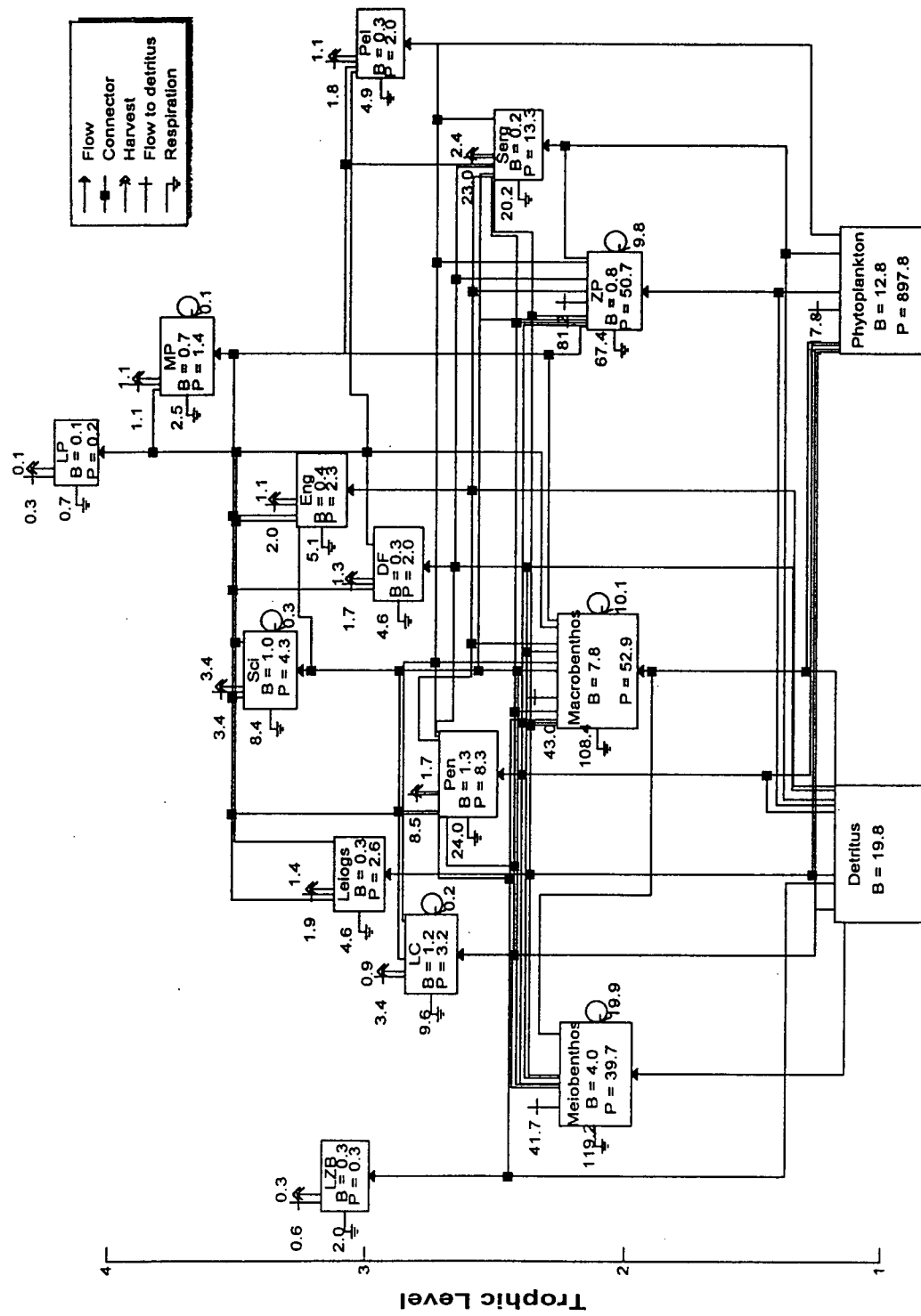


Figure 3.1 Trophic mass-balance model of San Miguel Bay, Philippines showing biomass and production, inflows and outflows and the trophic level for each eco-group. The units are in t km^{-2} , and the area of the Bay is 1115 km^2 .

The omnivory index in Table 3.8 is a measure of the variability in the trophic level at which the eco-groups feed (Christensen and Pauly 1992a). A value of 0 indicates complete specialisation in prey selection. Higher values indicate less specific feeding habits. The sciaenids and the large zoobenthos feeders have low omnivory indexes. The penaeids, large crustaceans and demersal feeders have the highest values. This makes sense for the former are recognised as generalists and the demersal feeders eco-group is composed of a large number of species. However, the omnivory index does not capture well, if at all, the variability of prey within trophic levels. For example, the prey of the large predators includes (in almost equal proportions) demersal feeders, leiognathids, engraulids, pelagics, sciaenids and medium predators plus some macrobenthos. However, the engraulids have a slightly higher omnivory index, yet they feed on fewer eco-groups, but over a greater number of trophic levels.

Indices of Maturity: Is San Miguel Bay a Mature Ecosystem?

There are several measures that indicate the level of maturity of an ecosystem. These measures are adapted from the maturity index of Odum (1969) and systems theory developed by Ulanowicz (1986). Christensen (1995, 1994) and Christensen and Pauly (1993c) describe these measures and relate them to 41 ECOPATH models. Christensen (1995) showed that for most of these measures, the articulation of the model (the number of boxes) does not matter, when the models include at least 8-12 boxes. Thus comparisons between models with different numbers of boxes can be made. Their analyses can be usefully applied to the San Miguel Bay

ECOPATH results. An attempt is made here then, to compare, where possible, these measures for the 41 ECOPATH models with the San Miguel Bay model.

As ecosystems mature, they should become more dependent on detrital flows than on flows from primary producers Odum (1969). 56% of the total flow in San Miguel Bay originates in the detritus.

The ratio of the total primary production to total respiration (PP/R) is 2.35. Odum (1971) proposed that in immature systems, primary production would be much greater than respiration while in mature systems, the ratio would approach 1. Christensen and Pauly found that for the 41 ECOPATH models, the modal PP/R value was between 0.8 and 1.6. They compared this to the work of Lewis (1981) who found the model value to be between 1.6 and 3.2. The San Miguel Bay value is comparable to latter PP/R values.

The ratio of the total system productivity to total system biomass (P/B) is high in developing systems and low in mature systems (Margalef 1968). The San Miguel Bay value of 28.7 falls within the lower range of those in Christensen and Pauly (1993c).

Christensen and Pauly concluded that cycling in ecosystems is related to maturity, as proposed by Odum (1969). Cycling is quantified in ECOPATH using the Finn Cycling Index (FCI, Finn 1976), expressing the percentage of the total throughput which is actually recycled. The total throughput is the sum of all flows, that is, consumption, exports, respiratory flows and flows to the detritus. The FCI in San Miguel Bay is 7.5%. When FCI was plotted against system

overhead⁶⁰, Christensen and Pauly found that ecosystems with very low cycling, such as upwelling systems have low system overhead. System overhead is positively related to the stability of the ecosystem. It was suggested by Christensen and Pauly that ecosystems with very high FCI may be less stable because they need to maintain a pattern of intricate flows. However, Vasconcelles *et al.* (*In press*), compared the response of 18 systems to perturbation, and concluded that FCI is positively related to system maturity. The San Miguel Bay FCI and system overhead (71.3%) place it at an intermediate level (Figure 6, Christensen and Pauly 1993c) which Christensen and Pauly suggest may be optimal.

Christensen and Pauly also plotted the FCI against the PP/R ratio and against mean path length. The mean path length is the average number of eco-groups that a unit of flow passes through on its way from inflow to outflow, and is calculated as $\text{Throughput}/(\Sigma \text{Export} + \Sigma \text{Respiration})$, (Christensen 1995). Path length will be affected by diversity of flows and cycling. Since these increase with increasing maturity, it is assumed that long path lengths are associated with mature ecosystem. San Miguel Bay again falls within the range of the 41 ECOPATH models described in Christensen and Pauly (1993c Figures 8 and 9). The mean path length in San Miguel Bay is 2.89. This would be classified within either the tropical shelves category or tropical estuaries of Christensen and Pauly. The tropical systems so far examined, tended to have longer path lengths than the non-tropical ecosystems.

⁶⁰ System overhead is a calculated statistic in ECOPATH. It is the difference between two measures, the ascendancy and capacity of an ecosystem. Ascendancy is a measure of the average mutual information in a system, scaled by system throughput. The upper limit of ascendancy is the capacity. The overhead, the difference between these two, is a measure of how much the ascendancy can increase. It is thus a measure of what the system has in reserve, a measure of its ability to cope with change or stress. It is a measure of stability (Christensen and Pauly 1992a).

The residence time of energy in the ecosystem is also a measure of maturity (Hannon 1979, Herendeen 1989). It is measured as the ratio between the total system biomass and the sum of all respiratory flows and export. In San Miguel Bay, the residence time is 0.035 years. This places it at an intermediate level in the overall ranking, by residency time of the 41 ECOPATH models in Christensen and Pauly (1993c, Figure 15).

Christensen (1994, 1995) ranked the 41 models using a maturity index based on seven of Odum's attributes. He then investigated the behaviour of different goal functions with respect to this maturity ranking, again using the 41 models. One of the goal functions used by Christensen was ascendancy. Ascendancy is assumed to be a measure of the growth and development of an ecosystem (Ulanowicz 1986). Growth is measured by the increase in the energy throughput of an ecosystem while development is measured by the increase in information content of the flows in an ecosystem. Ascendancy is calculated then as the product of the energy throughput and the average mutual information (Christensen and Pauly 1992b, after Ulanowicz and Norden 1990). Christensen found that there was no good relation between ascendancy and maturity and concluded that only the growth part of the function was reflected, due to a great variation in the system throughput between ecosystems. He suggests that the computational aspect of ascendancy should be reconsidered.

However, Christensen (1995) found that another goal function, relative ascendancy was highly,

but negatively correlated with maturity⁶¹. Relative ascendancy is the ratio of ascendancy to capacity (the upper limit of ascendancy) and is a dimensionless function. The relative ascendancy in San Miguel Bay is 28.7. This places San Miguel Bay amongst the 12 most mature models, ranked by Christensen on the basis of their relative ascendancy values⁶². The other top 12 models included the Gulf of Thailand, Brunei Darrussalam, Campeche and Yucatan, that is, other tropical shelf systems.

Compared to the 41 ECOPATH models ranked by Christensen (1994, 1995) and Christensen and Pauly (1993c), the San Miguel Bay ecosystem is at an intermediate level to high level of maturity. The results also indicate that the San Miguel Bay model is comparable to the other model results, giving it some validation. However, there are also some inconsistencies which are noted below.

⁶¹ This result is contrary to Ulanowicz's premise that ascendancy increases with ecosystem maturity. Christensen (1995) offers a rationale "Bearing in mind that the complementary measure to the relative ascendancy is the system overhead, and that the overheads are a measure of system stability (Rutledge *et al.* 1976).....this can be interpreted to mean that the maturity ranking is strongly (positively) correlated with ecosystem stability". (see also footnote 60)

⁶² The relative ascendancy values for the 41 models were kindly provided by V. Christensen

Some Inconsistencies in the ECOPATH Results

The transfer efficiencies (TE) between trophic levels are high, ranging from 16.9 to 22.8 (Table 3.9)⁶³. The classic value for TE is 10%, that is, one tenth of the energy that flows into a trophic level is transferred to the next trophic level (Lindeman 1942). ECOPATH calculates the TE as the ratio between the total exports from and predation on a given trophic level and the energy throughput or consumption by that trophic level (Christensen and Pauly 1992a). This definition includes the fishery as predators of the ecosystem. Herein lies the cause of the high TEs. The fishery exerts a very high degree of fishing mortality at trophic levels 3, 4 and 5 (Table 3.9). Fishing accounts for 40% of the TE at trophic level 3, 72.5% of the TE at trophic level 4 and 87% of the TE at trophic level 5. The TE at trophic level 6 is entirely accounted for by mortality due to fishing. High fishing mortality is also cited by Moreau *et al.* (1993) to explain the high transfer efficiencies found in Lake Victoria.

High fishing mortality does not explain the TE of 17.7 for trophic level 2. However, this value is comparable to the TEs at trophic level 2 for other tropical coastal systems such as Brunei Darrussalam and several coastal ecosystems from Mexico. It is higher than the Gulf of Thailand TE however. Christensen and Pauly (1993c) note that high GEs lead to high TEs, since GE is the TE of the eco-groups. They also note that the high TEs found for several

⁶³ TEs are calculated from the trophic aggregation routine in ECOPATH. This routine aggregates the ecosystem into discrete trophic levels and allocates the flows within the ecosystem between these trophic levels. The TEs thus refer to the discrete trophic levels, not to the fractional trophic levels in Figure 3.1. With discrete trophic levels, the eco-groups are not assigned to any one trophic level as was the case with the fractional trophic levels. Here, if 60% of the prey are detritus and phytoplankton and 40% are zooplankton, as in the case of the sergestids (Table 3.7), then these are the relative fractions of the flow through the group which are attributed to trophic levels 2 and 3 respectively. The total flow at each trophic level is thus the sum of the relative flows over the eco-groups.

Table 3.9 Transfer efficiencies, TE, and related indices between trophic levels in San Miguel Bay.

| <i>Trophic Level</i> | <i>II</i> | <i>III</i> | <i>IV</i> | <i>V</i> | <i>VI</i> |
|---|-----------|------------|-----------|----------|-----------|
| Transfer Efficiency (%) | 17.66 | 21.77 | 22.75 | 21.10 | 16.67 |
| <i>Outflows</i> | | | | | |
| Consumption by predators ($\text{tkm}^{-2}\text{year}^{-1}$) | 104.44 | 13.63 | 0.85 | 0.02 | 0 |
| Consumption by fishery ($\text{tkm}^{-2}\text{year}^{-1}$) | 3.30 | 9.10 | 2.25 | 0.16 | 0.004 |
| Sum of all out flows ($\text{tkm}^{-2}\text{year}^{-1}$) | 107.74 | 22.73 | 3.10 | 0.18 | 0.004 |
| <i>Inflows</i> | | | | | |
| Throughput/Consumption | 609.99 | 104.44 | 13.63 | 0.85 | 0.024 |
| % of TE due to fishing mortality | 3.07 | 40.03 | 72.50 | 86.67 | 100 |

tropical coastal shelf models may be a result of lack of independence in model construction. Since these models were used as a basis for some of the input parameters of the San Miguel Bay model, this reasoning may also help to explain the high TEs found for San Miguel Bay.

The TEs indicated that San Miguel Bay is an efficient ecosystem, with respect to the transfer of energy up the food web. The high level of fishing mortality contributes to this overall efficiency.

The production/biomass (P/B) ratios are also high. Some of the P/Bs were derived from the mortality estimates in Chapter 2. It was suggested that the high Zs may be due to inaccuracy in the estimation method, that they may be due to export or that they reflect high mortality due to the fact that a large proportion of fish were juveniles. These explanations remain valid. However, some of the P/B estimates were taken from the Gulf of Thailand which also has high P/Bs (Table 3.2). It has been suggested by Margalef that high P/Bs are indicative of ecosystems which are highly stressed (Rapport *et al.* 1985). If stress is equated with high levels of fishing mortality then both the Gulf of Thailand and San Miguel Bay would be categorised as stressed. The key question here is whether there is a significant export factor in San Miguel Bay. This question is addressed below.

The biomass estimates from the trawl survey data (Chapter 2) were not large enough to support the fishery. In order to make the ECOPATH model balance, it was necessary to allow the model to estimate biomass. The resultant biomass estimates differed from the biomass estimates from the trawl survey. The total fished biomass in the trawl survey was 2.72 tkm^{-2} .

The biomass estimated by the ECOPATH model was 5.78 tkm^{-2} , about twice the trawl survey estimate. This was estimated from a large predator biomass of 0.095 tkm^{-2} , 10 times the trawl survey estimate of 0.008 tkm^{-2} . The ECOPATH total fished biomass estimate is within the confidence limits of the trawl survey estimate (Table 2.4), although it was noted that the biomass estimate from the trawl survey was rather imprecise, that is, its confidence limits were wide. In addition, the trawl survey did not include the months of July and August. Data on catch rates indicate that biomasses are very high during these summer months. Thus, if the trawl survey had included data for the entire 12 months, the biomass estimate would have been greater than 2.72 tkm^{-2} . The total ECOPATH fished biomass estimate is also comparable to the total fished biomass estimates from other ECOPATH models. For example, the total fished biomass in the Gulf of Thailand was 9.35 tkm^{-2} (Pauly and Christensen 1993), Brunei Darussalam 10.53 tkm^{-2} (Silvestre *et al.* 1993) and in Malaysia was 4.02 tkm^{-2} (Liew and Chan 1987). The San Miguel Bay estimate is at the lower end of this range. In terms of the total fished biomass therefore, the ECOPATH estimate is acceptable.

However, the relative abundance of the eco-groups in the trawl survey and the ECOPATH model are different (Figure 3.2). There are some serious discrepancies. The differences are due to two factors. First, not every eco-group is equally available to the trawl survey. It is not possible for a survey designed with only one type of gear to target all species equally. Therefore some species will be under or over represented in the trawl catch. Secondly, ECOPATH uses catch estimates to estimate biomass. That is, ECOPATH calculates the biomass required to support a given catch: the greater the catch, the greater the biomass

Legend for Figures 3.2, 3.4-3.6

ZP = Zooplankton,
Meiob = Meiobenthos,
Macrob = Macrobenthos,
Serg = Sergestids,
Pen = Penaeids,
LC = Large Crustaceans,
DF = Demersal Feeders,
Leiog = Leiognathids,
Eng = Engraulids,
Pel = Pelagics,
Sci = Sciaenids,
MP = Medium Predators,
LZB = Large Zoobenthos Feeders,
LP = Large Predators,
Ph/P = Phytoplankton,

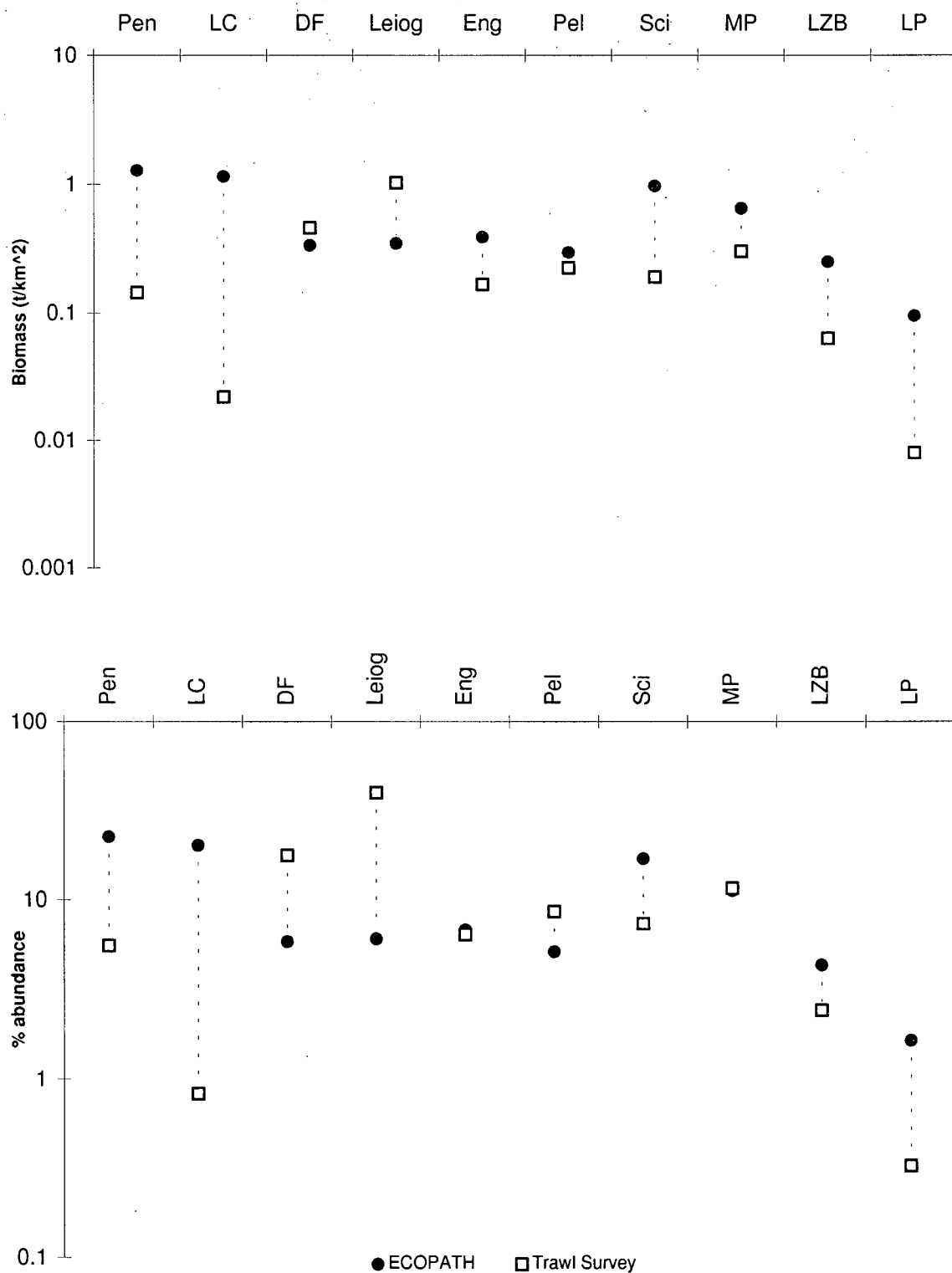


Figure 3.2 Comparison of (a) biomass estimates and (b) relative abundance from the ECOPATH model and the San Miguel Bay 1992-1994 Trawl Survey.

estimate⁶⁴. The catch estimates in San Miguel Bay are derived from the landings of over 20 different types of fishing gear, all targetting different groups. Relative catches are therefore quite different from the relative abundance in the trawl survey and, consequently, the relative abundance of the ECOPATH estimates of abundance are quite different too. The large crustaceans, for example, are hardly represented in the trawl survey, while they have a large catch, mostly by the crab gillnet. Similarly, the large predators are poorly represented in the trawl survey, but are caught in relatively large numbers by the set longline gear. Given these kinds of differences, it seems unlikely that ECOPATH could reproduce the relative abundance of a trawl survey from catch derived biomass estimates.

Sensitivity Analysis

A sensitivity analysis of the input parameters was performed using a routine within ECOPATH designed for this purpose. Each of the basic input parameters is varied by -50% to +50% in steps of 10%. The impact of this on the other missing input parameters is estimated as:

$$(\text{Estimated parameter} - \text{Original parameter}) / \text{Original parameter}.$$

The general result for all the eco-groups is illustrated in Figure 3.3. Here the sensitivity of the model to the sciaenid parameters is shown. There are several things to note.

⁶⁴ Reducing the catch of the sciaenids to 0.5 tkm^{-2} for example, reduces the biomass estimate from 0.972 tkm^{-2} to 0.221 tkm^{-2}

1. The P/B and EE input parameters have the same impacts on the other parameters,
2. Reducing P/B and EE by 50% has a greater effect than increasing them by 50%,
3. Changing the input parameters of most eco-groups, affects only that eco-groups,
4. The model results are not very sensitive to the Q/B input parameters.

Thus most of the sensitivity in the model is the sensitivity of the biomass estimates to the P/B and EE parameters. There are some exceptions. The phytoplankton biomass is sensitive to the zooplankton P/B and EE parameters. The meiobenthos biomass is extremely sensitive to its P/B and EE ratio and Q/B ratio. The sergestids P/B and EE impact on the biomass of the zooplankton and the EE of the phytoplankton. The sciaenids, depicted in Figure 3.3, have the widest ranging effects. The biomass of the large crustaceans, meiobenthos, macrobenthos, sergestids and penaeids undergoes at least a 50% increase when the P/B or EE of the sciaenids is reduced by 50%.

In general then, the model is robust, although it would be improved by a better empirical base for the P/B and EE parameters. This is especially pertinent given the discussion of high P/B ratios⁶⁵.

⁶⁵ This further legitimates the procedure in the balancing of the model, where the model was allowed to estimate the Q/B of the sciaenids rather than the P/B. Allowing the model to estimate Q/B, produced a lower P/B and a much higher sciaenid biomass.

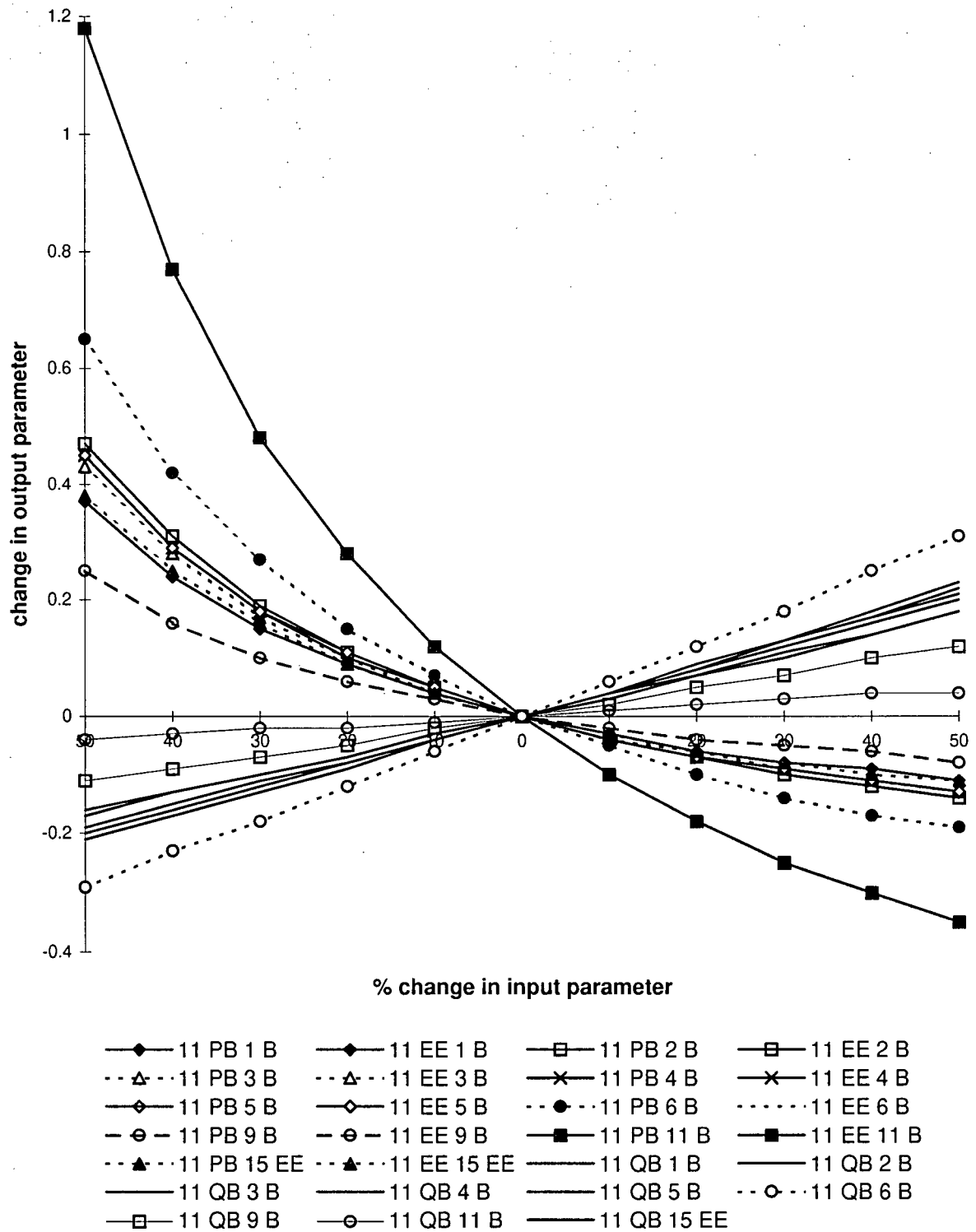


Figure 3.3 Sensitivity Analysis for the Sciaenids. Each of the input parameters is varied from -50% to +50% and its impacts on the unknown parameters plotted. For further details see the text.

PB = Production/Biomass, QB = Consumption/Biomass, EE = Ecotrophic Efficiency

Trophic Impact Routine

ECOPATH has a routine that enables the impact of increasing the biomass of one eco-group on the biomass of the other ecogroups to be studied. Thus it begins to be possible to ask “what if?” questions. The trophic impact routine originates in economic input/output theory.

Leontief (1951) developed a method to examine the direct and indirect interactions in the economy of the USA, using what has become known as the Leontief Matrix. The economic theory was adapted to ecology by Hannon (1973) and Hannon and Joiris (1989). Ulanowicz and Puccia (1990) used a similar approach and their methodology is incorporated into ECOPATH (Christensen and Pauly 1993b).

The results of the trophic impact routine are given in Figure 3.4. All eco-groups respond negatively to an increase in their own biomass, due to increased competition within the eco-group for resources. The eco-groups with the greatest impact on other groups are the detritus, phytoplankton, zooplankton and macrobenthos. The detritus has a positive impact on all groups apart from the phytoplankton. The phytoplankton has a positive impact on the zooplankton, sergestids, leiognathids, engraulids, pelagics, sciaenids, medium predators and large predators. The zooplankton has also has a positive impact on the sergestids, leiognathids, engraulids, pelagics, sciaenids, medium predators and large predators, and a negative impact on phytoplankton. The positive impact of the phytoplankton on these groups is probably due to its positive impact on the zooplankton. The macrobenthos has a positive impact on all groups apart from the zooplankton, meiobenthos, sergestids and detritus.

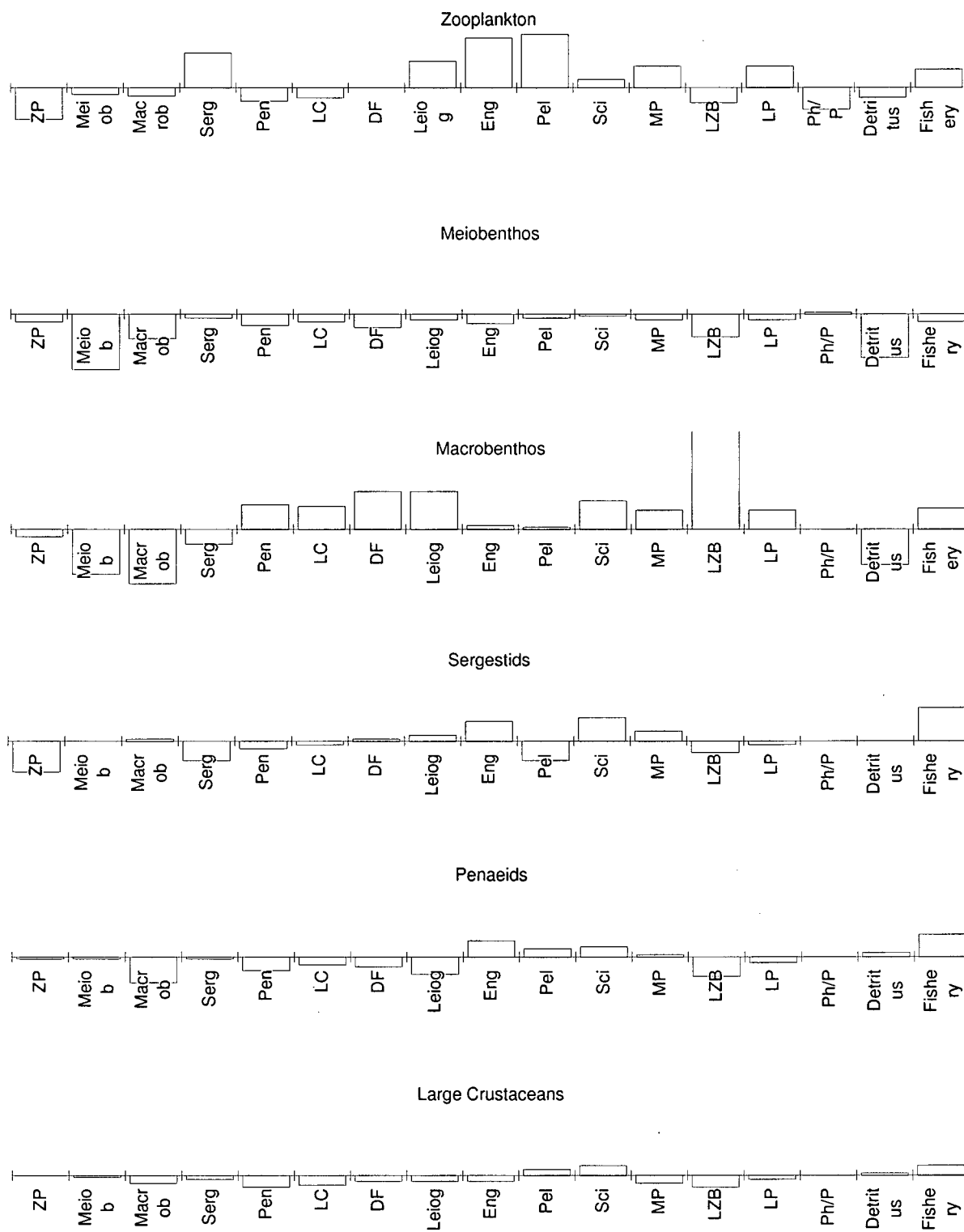


Figure 3.4 Results of the Trophic Impact Routine. The eco-groups on the x-axis are responding to an increase in biomass of the named group. The impacts are relative but are comparable between groups. See text for further details.

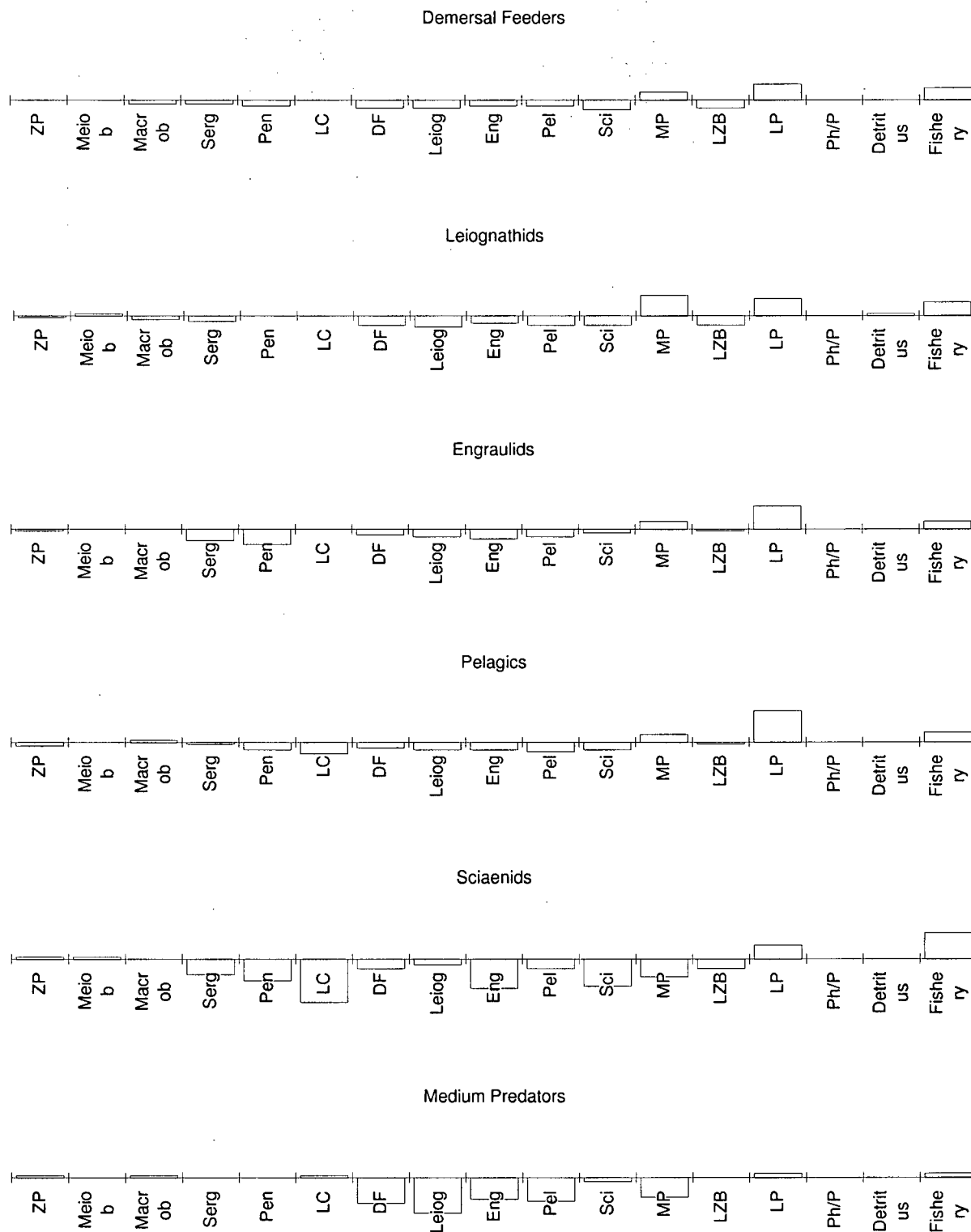


Figure 3.4 (cont) Results of the Trophic Impact Routine. The eco-groups on the x-axis are responding to an increase in biomass of the named group. The impacts are relative but are comparable between groups. See text for further details.

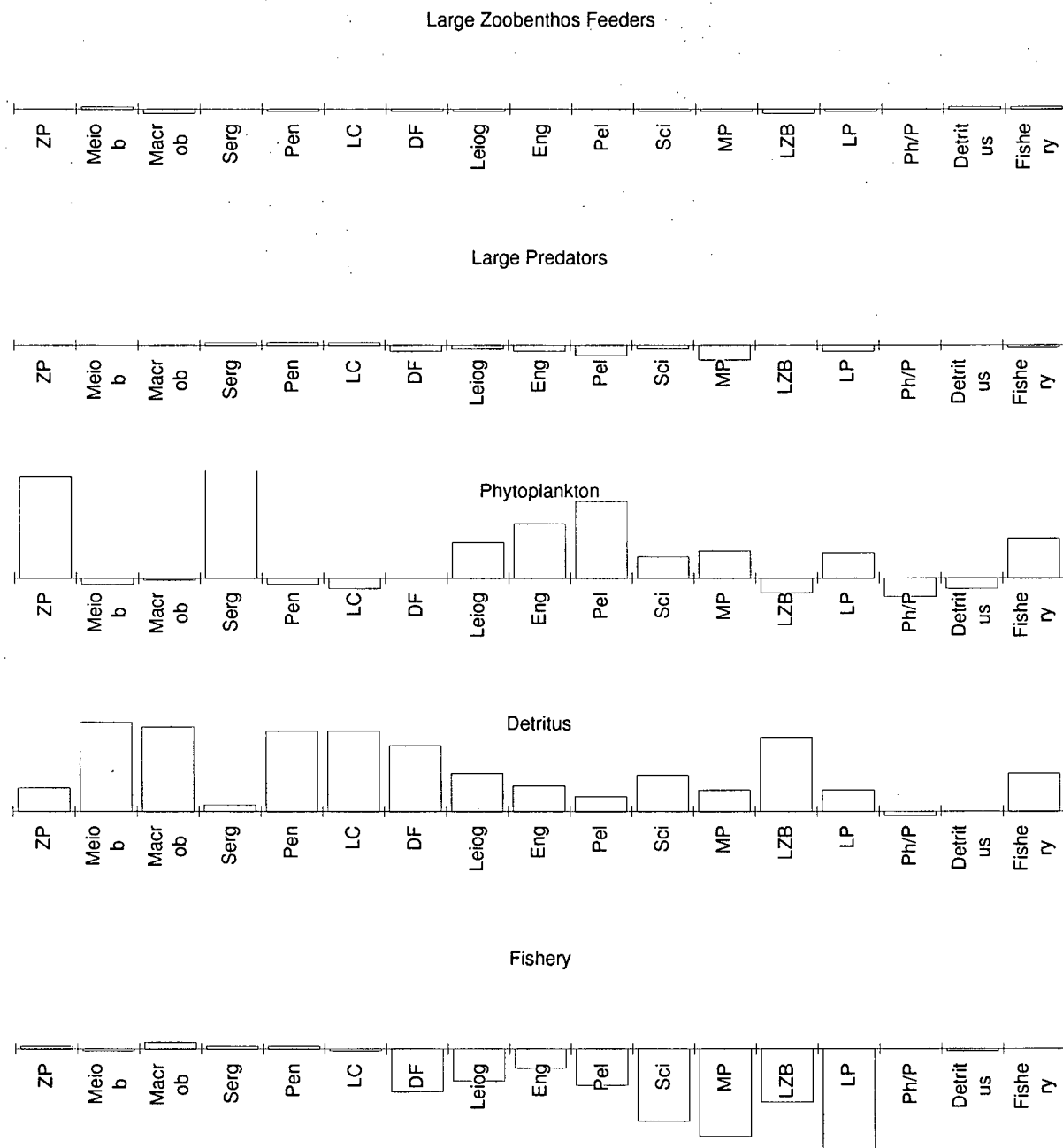


Figure 3.4 (cont.) Results of the Trophic Impact Routine. The eco-groups on the x-axis are responding to an increase in biomass of the named group. The impacts are relative but are comparable between groups. See text for further details.

None of the other eco-groups have comparable impacts. This would suggest that the San Miguel Bay ecosystem is largely controlled by the lower trophic levels. Considering that the fishery has long since reduced the biomass of the top predators (Chapter 2), this result is reasonable. It was noted in Figure 3.1, that many of the eco-groups are clustered around trophic level 3. An increase in the biomass of any of these groups, the leiognathids for example, leads to an increase in the medium predators and the large predators. The biomass of the other eco-groups at this trophic level, the demersal feeders, leiognathids, engraulids, pelagics decreases. The sciaenids and the penaeids also decrease. These groups probably decrease through the combined effects of predation by the large predators and the medium predators and competition between trophic level 3 groups. An increase in the biomass of the medium predators leads to a decrease in most of the other eco-groups and a small increase in the large predators. When the biomass of the large predators is increased, it produces small decreases in all the fish groups and small increases in the three crustaceans groups. This behaviour, and the behaviour noted above, resembles a trophic cascade (Carpenter *et al.* 1985).

The three crustacean groups, the sergestids, penaeids and large crustaceans each decrease in response to an increase in the biomass of the other two crustaceans. This indicates that there is competitive interactions between these three groups. Only the engraulids, pelagics, sciaenids and medium predators respond positively to an increase in any of the crustacean groups.

The trophic impact routine describes a fishery that is more affected by the lower trophic levels than higher trophic levels. It has a large number of eco-groups at trophic level 3 which are in competition with one another for resources. These include two to three crustacean groups

which also compete with one another. The top predators in the system do impact on the intermediate groups, releasing lower eco-groups from predation pressure. Some trophic cascade effects are apparent. On top of this ecosystem, there is the fishery. The fishery always increases in response to an increase of any of the eco-groups, including those which are not fished. The fishery benefits from the unfished groups because of the positive impact these groups have on the higher trophic levels. An increase in the fishery has a negative impact on all of the fished groups, except the sergestids and penaeids which benefit from the reduction in predation and competition. The zooplankton and macrobenthos also increase for the same reasons.

Introducing the Fishery into the ECOPATH Model as Large-Scale and Small-Scale “Predators”

The trophic impact routine indicates that the fishery has a big impact on the ecosystem. The fishery is composed of many gear types (see Chapter 2) which are specifically designed to target certain fish or crustacean groups. Different fishing gears would therefore be expected to have different effects on the ecosystem. This is examined in more detail. Instead of including the fishery in the ECOPATH model simply by including catches as export, the fishery is broken down into gear groups and included as predators within the model structure.

The validity of this approach has been questioned (McGoodwin 1991). McGoodwin discusses the equating of fishers as predators in ecological systems and concludes “Ecosystemic models

which conceptualize humans as any other predator in a marine environment.....assert or at least imply erroneous parallels between human fishers and other marine predators by inhering questionable assumptions about the nature of the relationships between the human fishers and their prey” (1991:68). McGoodwin’s premise is that humans and their marine prey have not co-evolved as predators and prey within the marine ecosystem have, humans are not recycled into the ecosystem, the link between human predator and marine prey does not include marine prey predating on humans at a different life cycle stage, and fishers predate on the marine environment for more and different reasons than marine predators predate.

This approach is nonetheless justified here because it is used in order to understand the effects of fishing on the ecosystem of San Miguel Bay. It is not used to predict or manage the behaviour of fishers. ECOPATH is a static model and the dynamic feedback cycles envisioned by McGoodwin do not apply here.

Method

In order to include the fishery in the ECOPATH model as predators, certain assumptions are made concerning the input parameters of the “fishery predators” (Villy Christensen, pers. comm.). First, their biomass is arbitrarily designated to be 1⁶⁶, consumption is the catch and production is the quantity of fish landed. Unassimilated food is equated with discards, and thus flow to detritus, and respiration, the energy required by the system can be equated with fish

⁶⁶ This is valid since biomass simply operates as a multiplier in the linear system of equations.

consumed aboard the vessel. The diet composition of the “fishery predators” is equivalent to their catch composition.

The harvest in Table 3.8 is thus now “consumed” by the “fishery predators”. There is no export from the original eco-groups. Instead, all exports are exported via the “fishery predators”. The total export from the ecosystem remains the same. In effect, a fishery is fishing the “fishery predators”. This is a ploy to fool the model; it does not affect the computation of flows, although it does add an extra trophic level to the system.

The fishery was modelled as “fishery predators” in two ways. First, the fishery was included as a large-scale predator and a small-scale predator (*LS-SS predator* model). The large-scale predators include the large, medium and baby trawlers, and the small-scale predators include all the other fishing gears (see Chapter 2). In order to examine the impact of the small-scale sector in greater detail, it was then subdivided into the gear groups shown in Table 3.10 (*LS-SSgears* predator model). This model consisted of one large-scale predator and eight small-scale predators. The input parameters for the “fishery predators” are also shown in Table 3.10.

Results

The results of the *LS-SS* predator and *LS-SSgears* ECOPATH models for the biological eco-groups are as described above⁶⁷. The results for the “fishery predators” are shown in Table 3.10. The EE and GE are 1 since consumption and production are equivalent and there is no respiration.

The trophic levels at which the small-scale fishing gears fish are variable. The gears which fish at the highest trophic levels are the ordinary gillnet and the hunting gillnet. The mini-trawler and the fine meshed gears fish at the lowest trophic levels. The former target sciaenids and the latter sergestids and penaeids. The large-scale sector fishes at an intermediate trophic level. It has a non-specific target catch. The small-scale sector collectively fishes at a lower trophic level than the large-scale sector.

The “fishery predators” have very low “omnivory indices”, indicating that they do not fish across many trophic levels (cf. Table 3.8). This is particularly true for the crab gear which has an omnivory index of 0.002 and catches almost 100% crabs. The hunting gillnet, mini-trawler and ordinary gillnet also have low indices. Of the small-scale gears, the highest omnivory index is incurred by the “other gear”. This makes sense since it is a composite group of gears, which vary from handlines to beach seines to spear guns. The “other gillnets” and the fine meshed gear also have higher omnivory indices. The small-scale sector has a much higher

⁶⁷ In some cases there are very slight differences. For example the EE is estimated in the fishery predator model because all other parameters for the biological eco-groups are entered. The estimated EE for some groups is 0.949 or 0.951 instead of 0.95. These differences are due to rounding differences in the input parameters and the calculations.

Table 3.10 Input parameters and results for the two “fishery predator” ECOPATH models, *LS-SS predator* and *LS-SSgears predator*.

| <i>FISHING GEAR</i> | <i>INPUT PARAMETERS</i> | | | | | | <i>RESULTS</i> | | |
|-------------------------|--|---|---|--|---|-----------|----------------|--------------------------|---------------------------|
| | <i>Biomass (tkm^{-2})</i> | <i>P/B ($year^{-1}$)</i> | <i>Q/B ($year^{-1}$)</i> | <i>Export (tkm^{-2} $year^{-1}$)</i> | <i>Harvest (tkm^{-2} $year^{-1}$)</i> | <i>EE</i> | <i>GE</i> | <i>Trophic Level</i> | <i>Omnivory Index</i> |
| Other Gear | 1 | 0.68 | 0.68 | 0.68 | 0.68 | 1 | 1 | 4.20 | 0.164 |
| Fixed Gear | 1 | 1.23 | 1.23 | 1.23 | 1.23 | 1 | 1 | 4.05 | 0.024 |
| Crab Gear | 1 | 0.66 | 0.66 | 0.66 | 0.66 | 1 | 1 | 3.75 | 0.002 |
| Fine Mesh Gear | 1 | 1.83 | 1.83 | 1.83 | 1.83 | 1 | 1 | 3.58 | 0.112 |
| Other Gillnets | 1 | 2.75 | 2.75 | 2.75 | 2.75 | 1 | 1 | 4.00 | 0.116 |
| Ordinary Gillnet | 1 | 2.37 | 2.37 | 2.37 | 2.37 | 1 | 1 | 4.31 | 0.088 |
| Hunting Gillnet | 1 | 1.51 | 1.51 | 1.51 | 1.51 | 1 | 1 | 4.33 | 0.040 |
| Mini-Trawler | 1 | 1.56 | 1.56 | 1.56 | 1.56 | 1 | 1 | 3.54 | 0.070 |
| Large Scale | 1 | 2.23 | 2.23 | 2.23 | 2.23 | 1 | 1 | 4.09 | 0.095 |
| Small Scale | 1 | 12.59 | 12.59 | 12.59 | 12.59 | 1 | 1 | 3.98 | 0.172 |

Other Gear = Set Longline, Handline, Fish Trap, Ring Net, Pull Net, Fish Weir, Beach Seine, Spear Gun.

Fixed Gear = Lift Net, Fish Corral.

Crab Gear = Crab Gillnet, Crab Liftnet.

Fine Mesh Gear = Filter Net, Scissor Net.

Other Gillnets = Shrimp Gillnet, Bottom-Set Gillnet, Surface Gillnet, Shark Gillnet, “Other Gillnets”.

Large Scale = Large Trawlers, Medium Trawlers, Baby Trawlers.

Small Scale = All small-scale gear, including Mini-Trawlers.

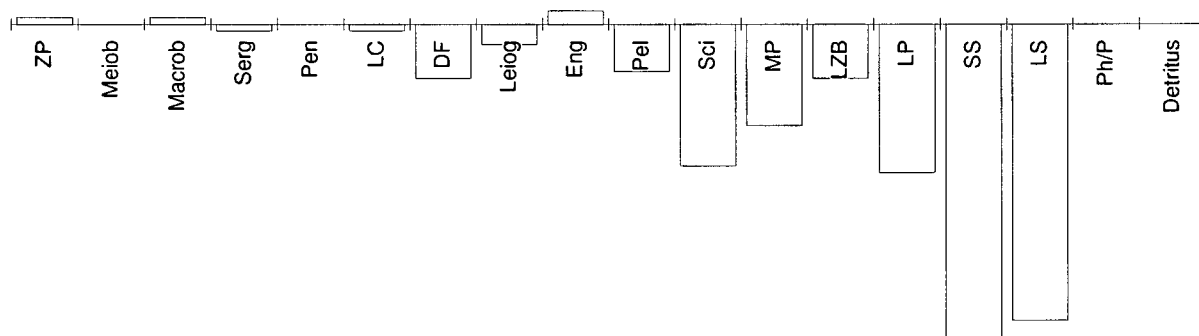
omnivory index than the large-scale sector, thus signifying that it fishes at a wider, if lower, range of trophic levels.

These results could be anticipated from the fishery analysis presented in Chapter 2. More interesting is the impact on the ecosystem of increasing fishing effort. This can be simulated using the trophic impact routine. Bearing in mind that this is a qualitative exercise, it is nonetheless possible to obtain a sense of the response of the eco-groups to an increase in fishing. The results of the trophic impact analysis of the *LS-SS predator* model are shown in Figure 3.5, and the results for the *LS-SSgear predator* model are given in Figure 3.6.

The two graphs in Figure 3.5 are drawn to the same scale. As would be expected, there are not many positive responses to an increase in fishing by either sector. It is immediately obvious however, that the small-scale sector has a much greater impact than the large-scale sector. The small-scale sector has a bigger impact on all the eco-groups except the leiognathids and the engraulids. The sciaenids suffer a large decrease when the small-scale sector is increased while they increase slightly when the large-scale sector is increased. The small-scale sector also indirectly impacts the non-fished groups, the zooplankton and macrobenthos, in the model. The latter increase in response to an increase in the small-scale sector. This manifestation is likely to be a response to a decrease in the biomass of their predators. The small-scale sector also has a greater negative impact on the large-scale and small-scale sectors themselves⁶⁸.

⁶⁸ This result should be interpreted with caution. The fishery is not subject to the same kind of feed-back cycles that the eco-groups incur (McGoodwin 1991).

Small Scale Sector



Large Scale Sector

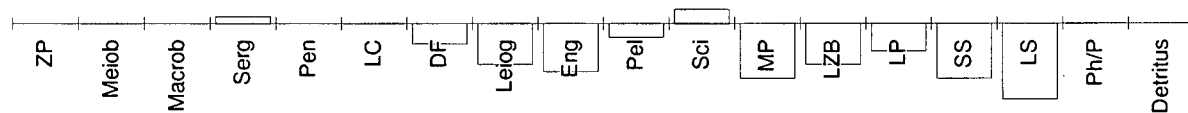


Figure 3.5 Results of the Trophic Impact Routine when (a) the small-scale sector and (b) the large-scale sector are increased. The eco-groups on the x-axis are responding to the increase. The impacts are relative but are comparable. See text for further details.

The graphs in Figure 3.6 are also drawn to the same scale, although it is smaller scale than in Figure 3.5. This is because the impacts of the small-scale gear groups individually are larger than the impacts of the small-scale sector collectively. This may be because when the small-scale sector as a whole is increased, the impacts of the different gears buffer each other. When an individual gear is increased, its effect alone is felt. However, it may also not be valid to make this comparison across models with a different number of eco-groups.

The individual small-scale gear (or more correctly, gear groupings) are more selective than the large-scale or small-scale sectors. This can be seen in their impact on the model and in the omnivory index. There is a differential response by the eco-groups to increased fishing by the different small-scale gears. For example, the ordinary gillnet and, to a lesser extent the hunting gillnet, target sciaenids and medium predators. These both decrease in response to increased fishing effort by the two gears. However, the eco-groups below them, with the exception of the pelagics, all increase, even though some are “consumed” by the ordinary gillnet and hunting gillnet. This increase is due to a decrease in predation from the sciaenids and the medium predators. The large zoobenthos feeders also increase.

The impact of the mini-trawler on the sergestids and the penaeids is negative. Since these are the target species of this gear, this result conforms with expectations. However, several of the other eco-groups increase. It is not immediately clear why these eco-groups should increase. The mini-trawler “consumes” all of these groups, albeit in relatively small quantities, with the exception of the large predators. It is possible that the demersal feeders and the leiognathids increase because they are competitors with the sergestids and penaeids which have decreased.

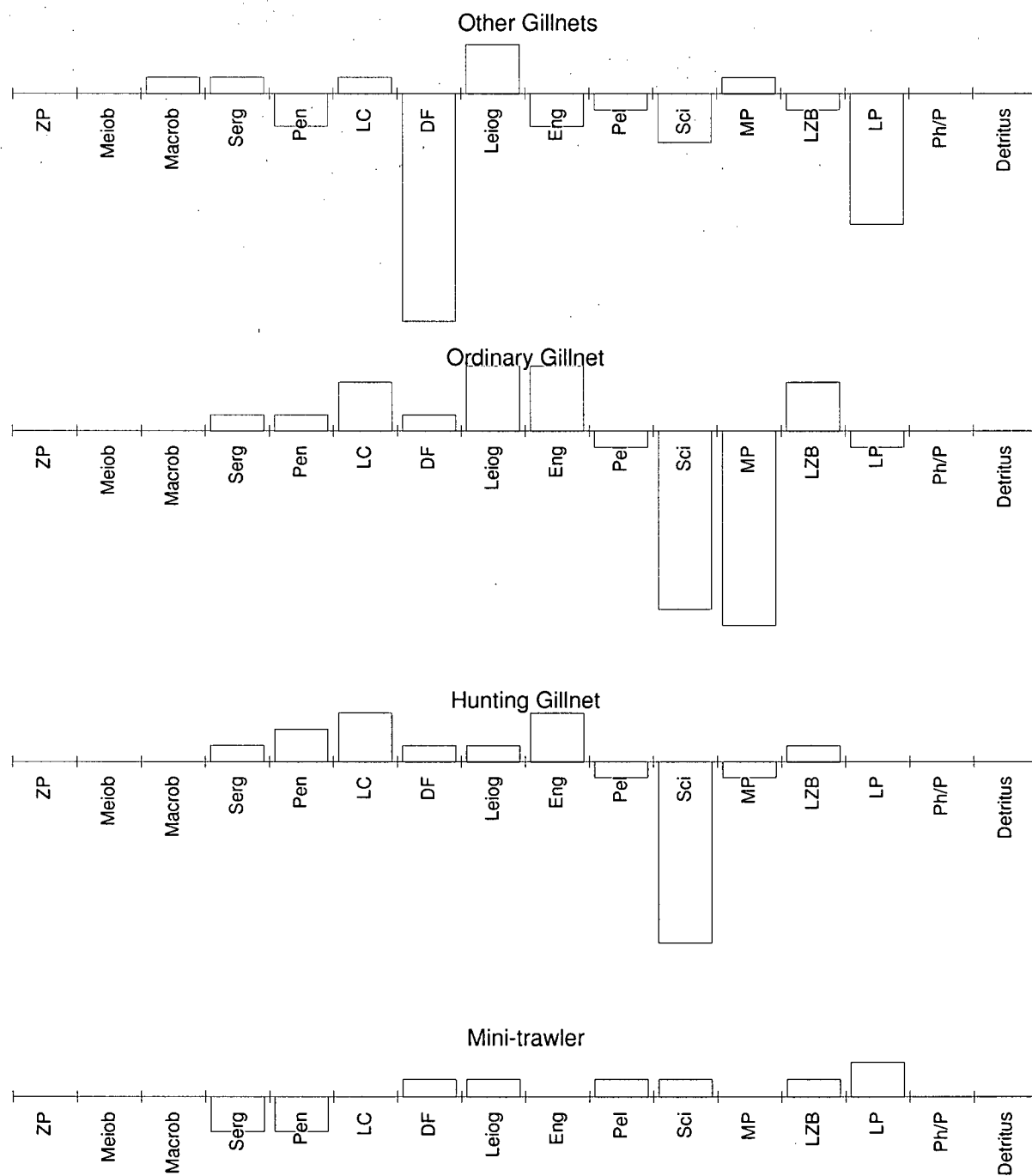


Figure 3.6 Results of the Trophic Impact Routine when different small-scale gears are increased. The eco-groups on the x-axis are responding to the increase. The impacts are relative but are comparable. See text for further details.

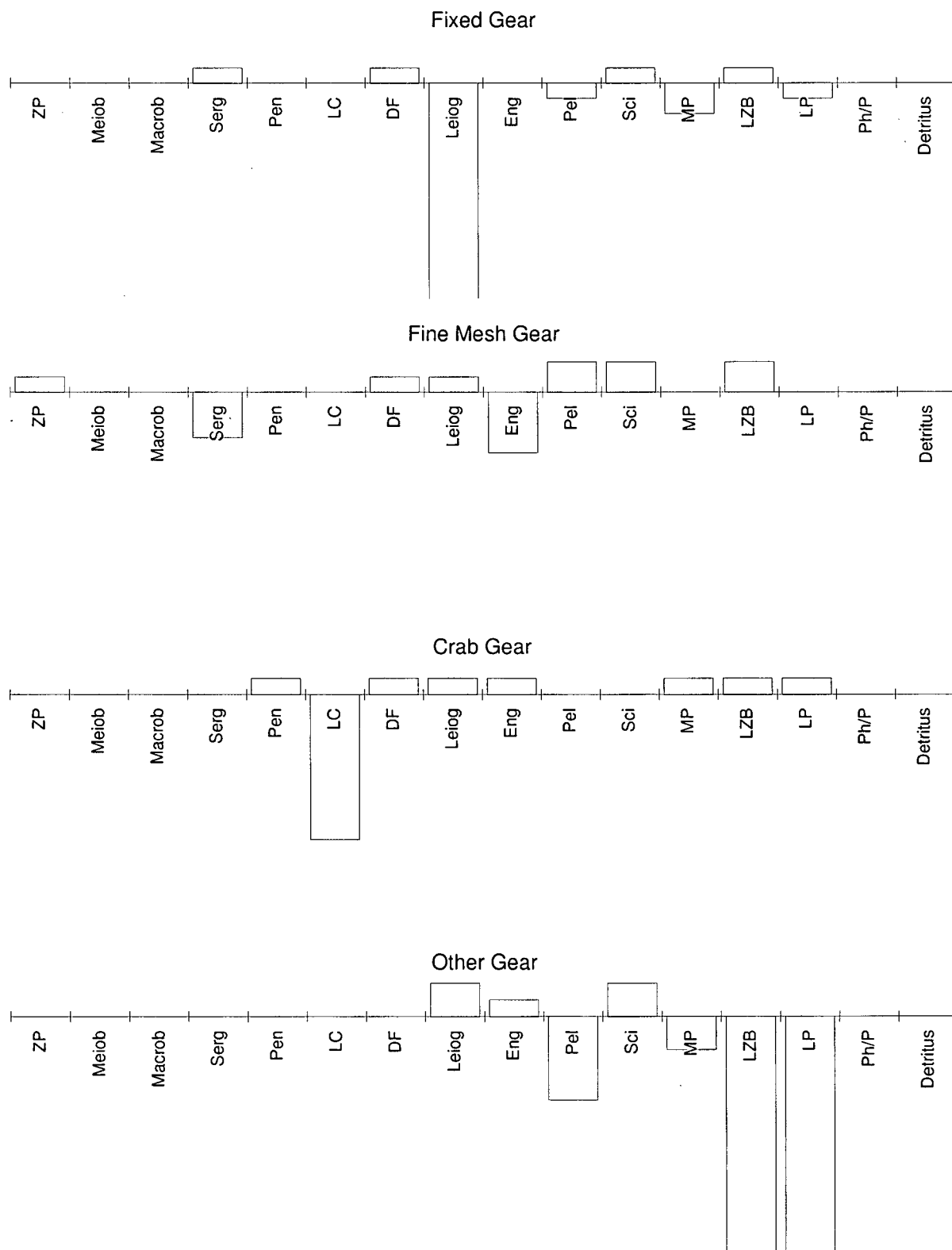


Figure 3.6 (cont.) Results of the Trophic Impact Routine when different small-scale gears are increased. The eco-groups on the x-axis are responding to the increase. The impacts are relative but are comparable. See text for further details.

Since the former increase, they provide more food for their predators. There is mixed support for this hypotheses from the earlier trophic impact analysis shown in Figure 3.4. The impact of an increase in the sergestids was a small increase in the leiognathids, demersal feeders and sciaenids and a decrease in the pelagics and large zoobenthos feeders. The impact of an increase in the penaeids was a larger decrease in the leiognathids, demersal feeders and large zoobenthos feeders and a decrease in the pelagics and sciaenids. Reversing these impacts to mimic a decrease in the sergestids and penaeids produced a very unclear picture. It may not be possible to understand the impact of a reduction in more than one eco-group by examining the impacts of individual eco-groups.

The crab gear target large crustaceans and their impact is clear. An increase in fishing by crab gear produces a large decrease in the large crustaceans. Other eco-groups increase. When the large crustacean biomass was increased in Figure 3.4, these eco-groups all decreased. Thus opposite impacts are seen when the biomass of the large crustaceans are increased, or decreased due to fishing. The increases in the biomass of the other eco-groups may be due to a reduction in competition with the large crustaceans for resources.

The fine meshed gear “consumes” a wide range of the fished eco-groups. The “diet” is dominated by the sergestids and, to a lesser extent, the engraulids. Other eco-groups are “consumed” in small quantities, and the large crustaceans, sciaenids and large zoobenthos feeders are absent from the “diet”. The impact of the fine meshed gear on the model is mixed. The sergestids and the engraulids decrease, the engraulids more than the sergestids. The zooplankton increase, probably due to a reduction in predation by the sergestids and

engraulids. Some of the eco-groups which are fished, such as the demersal feeders and leiognathids, increase in response to an increase in fishing.

The impact of the fixed gear is dominated by a large decrease in the leiognathids⁶⁹. The small increase in the demersal feeders and sciaenids may be a response to a reduction in competition. However, the fixed gear “consume” all the fished eco-groups except the large predators and the large crustaceans. So the mixed effects of fishing, predation and competition are being seen.

The “other” gillnets “consume” all the fished groups except the sergestids. They have a mixed impact. The greatest impact is on the demersal feeders which decrease. This response can be attributed to the impact of the bottom-set gillnet, the “diet” of which is dominated by demersal feeders. The increases in the medium predators, leiognathids, large crustaceans, sergestids and macrobenthos may be due to a decrease in predation by or competition with the large predators and demersal feeders. However, this small-scale gear group consists of many types of gillnets and it is not possible to determine whether the impacts on the other eco-groups are due to the direct or indirect impact of the Other gillnets.

This is also the case for the “other” gear. The large decrease in the large zoobenthos feeders and the large predators is due to the effect of increasing the fishing effort of the set longline. The leiognathids, the engraulids and the sciaenids increase. This may be due to a decrease in

⁶⁹ It was noted in Chapter 2 that the large catch of leiognathids by the lift net may be an artifact of poor sampling for this gear.

predation by the large predators and the medium predators. The decrease in the pelagics and medium predators is due to the direct impact of fishing.

Discussion

This analysis demonstrates the complexity of the interactions that occur between the effects of fishing mortality, predation mortality and competition. In some cases it is relatively straightforward to discover these interrelated effects, for example, the impact of the crab gear. However, for most of the other gears, the situation was more difficult. This might be reduced if the gear groups were broken down further. For some small-scale gears it was possible to link the likely effects of competition and predation and fishing. For others, for example, the mini-trawler the possible interactions of competition, predation and fishing were too interwoven to determine. An increase in one eco-group (A) produced an increase in eco-group (B), but an increase in eco-group (C) produced a decrease in eco-group (B). If both (A) and (B) increase, and (C) also increases, does this mean that the impact of group (A) is stronger, or are there other less obvious interactions occurring?

The trophic impact routine dramatically illustrates that the small-scale sector has a bigger and more widely ranging impact on San Miguel Bay than the large-scale fishery. The large-scale sector is non-selective and, although there are preferred catches, for example, penaeids, it catches a large range of species in its nets.

When taken as a whole the small-scale sector also catches a large range of species. Some of the individual gears catch as many different species as the large-scale sector, for example, the mini-trawler and the ordinary gillnet (Chapter 2). However, there is a fundamental difference. Almost all of the small-scale gears specifically and effectively target certain species or groups. Small-scale gears are more versatile than large-scale gears. In effect, the small-scale gears can and do fish any species in San Miguel Bay, in any niche, at any trophic level, anywhere. While the large-scale sector mass harvests non-selectively, the small-scale sector selectively mass harvests.

What this means is that there is no refuge in the Bay. When species "A" decreases, the small-scale sector has the versatility to switch to species "B" and so on. It was noted in Chapter 2 that the small-scale sector has diversified since the early 1980s. This diversification has increased the potential of the small-scale gear to target fish. The hunting gillnet, for example, uses scaring devices to drive sciaenids into the net. Apparently this gear was not present in the 1980s and thus the sciaenids had to swim into the net in order to be caught.

The common, and perhaps entrenched, view is that the large-scale sector is bad and the small-scale is good and benign and creates a livelihood for fishers. While the latter part of the statement is not contested, the benignity of the small-scale sector is. The small-scale sector in San Miguel Bay has the ability to fish at all trophic levels and does (Table 3.10). In addition, the small-scale sector imposes a greater fishing mortality than the large-scale sector on every eco-group, except the engraulids (Figure 3.7). The small-scale threat could be regarded as clandestine. Recent management actions in San Miguel Bay have aimed at banning the large-

scale sector (San Miguel Bay Integrated Coastal Fisheries Management Plan (unpublished manuscript). In effect large-scale effort may have been reduced (Mike Pido pers. comm.) However, this is not sufficient. The trophic impact results presented here support the conclusion reached in Chapter 2. All sectors of the fishery must be assessed and managed.

Interpretation and Discussion

The ECOPATH models of the San Miguel Bay fishery have increased knowledge about San Miguel Bay. The results of the first model indicate that the ecosystem is relatively mature⁷⁰, and thus relatively stable and able to respond to stress (Christensen 1995), that there are four trophic levels (Figure 3.1), that many of the species are grouped around trophic level 3, that there is likely to be strong competitive interactions between eco-groups at this level and that 56% of flows originate in the detritus. The trophic impact routine further indicated that considerable competition and predation occurs, although the eco-groups showing the greatest impact were those at the lower trophic levels. The results of the *LS-SS predator* and *LS-SSgears predator* models are an insight into the combined effects of fishing and biological interactions. The results also demonstrate how the fishery has “fished down” the food web.

⁷⁰ Relative to other models for which comparisons were possible.

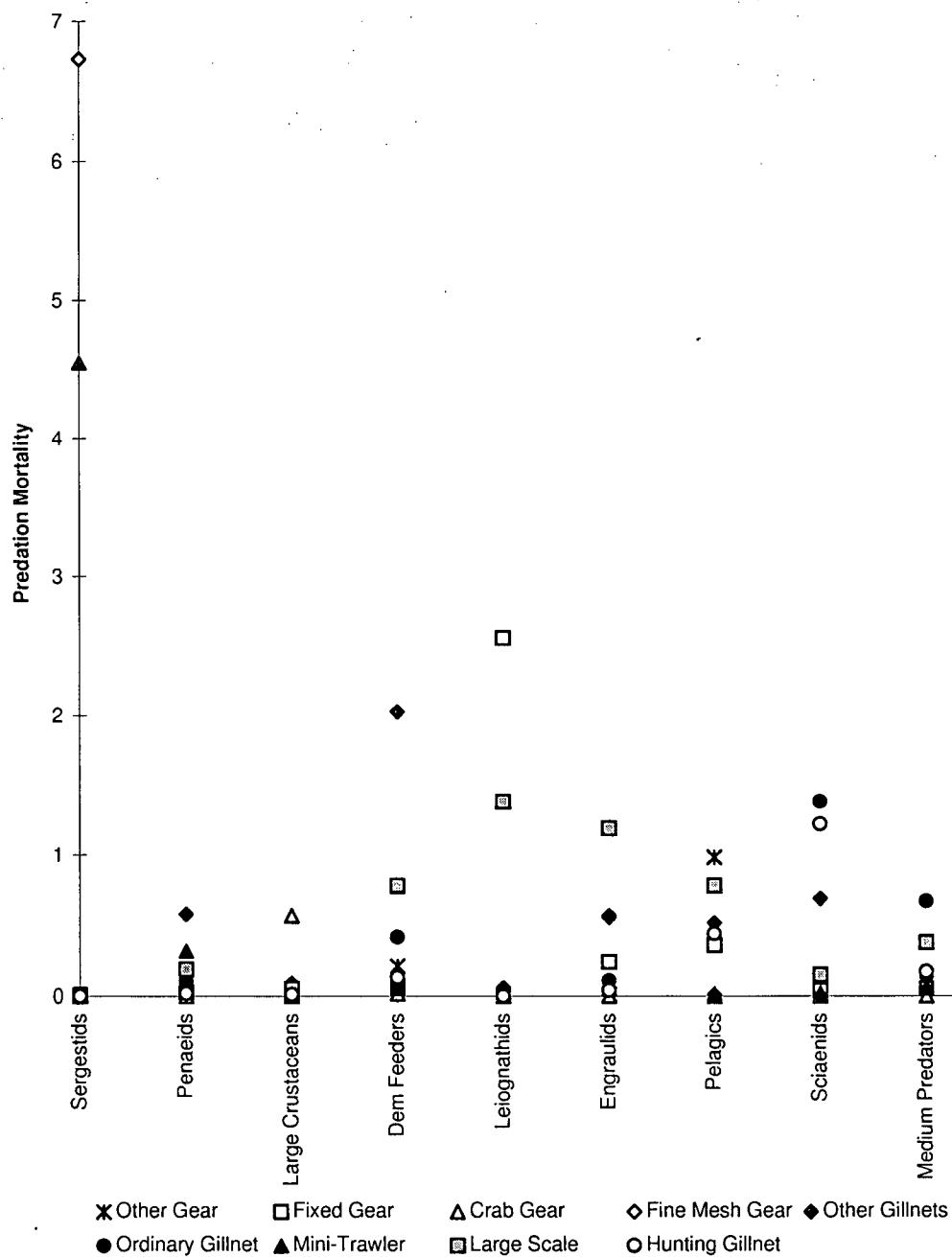


Figure 3.7 Comparison of the "predation mortality" (= fishing mortality) imposed by each "fishery predator" (= fishing gear) on the fished eco-groups.

These results are based on the data input to ECOPATH. By comparative standards, the data requirements of an ECOPATH model are small. However, as the first part of this Chapter attests, the parameter requirements are still quite demanding. The parameters used in this ECOPATH model have mixed origins and some were derived by, perhaps, somewhat convoluted means. A few were estimated directly from empirical data (some P/Bs, most Q/Bs, few diet compositions), some were estimated from empirical data and data from the literature and some were derived entirely from the literature. A skeptic could be forgiven for asking whether a model so constructed bears any resemblance to reality. This quasi-empirical approach has been used in many of the ecosystem models (Christensen and Pauly (1993a) and is recommended for ecosystems with insufficient data. As far as possible, the empirically derived parameters were not altered in the balancing of the model, thus ensuring that the model was fitted to the local data. In addition, the sensitivity analysis indicated that the most sensitive parameter is the P/B ratio and the EE. The P/B estimates from San Miguel Bay were comparable with other ecosystems, particularly the Gulf of Thailand. Certainly with respect to the pattern of fishing that has occurred in San Miguel Bay over the last two decades, they are feasible. The EE values were those recommended in Christensen and Pauly (1992a).

The time period of the San Miguel Bay ECOPATH model is 1 year. It was noted in Chapter 2 that there are seasonal effects in the fishery. Arguably, the dynamics of the ecosystem would be better represented by two models, one for the months of October to March, during the Northeast monsoon, and one from April to September, which would include the Southeast monsoon. Unfortunately there were insufficient data to estimate separate parameters for these two seasons. This was particularly so for the diet composition, most of which was drawn from

comparable models in the literature. It is therefore assumed that the annual model describes the Bay sufficiently.

Hypotheses About the San Miguel Bay Fishery and Resource

On the premise that the ECOPATH model is a useful construct, several hypotheses can be made about the fishery of San Miguel Bay. There is a fundamental query about San Miguel Bay. In the 1979-1982 study of the Bay, the fishery was declared highly/over-exploited. More than 10 years later, the fishery is still highly/over-exploited. The total catch declined by 17%. However, the fishery is still viable, perhaps more viable than would have been predicted in 1982. This is despite the fact that many fish are caught before maturity (Chapter 2). So the fundamental query is, "How is this resiliency possible?"

The trophic impact analysis of the small-scale sector showed that these gears exploit all trophic levels, niches and eco-groups in San Miguel Bay using an array of fishing methods. It was noted in Chapter 2 that there has been species succession, with "K" strategists being replaced by "r" strategists. The resilience of the fishery could simply be due to there still being room for the fishery to be fished further down the food web. The small-scale sector is able to respond to the changes caused in the resource through succession and continues to do so. The question then becomes, "What is the end point?"

A complimentary explanation may be found in the important role that detritus plays in the ecosystem. In Figure 4.4, all eco-groups, except the phytoplankton, responded positively to an increase in detritus, including the fishery. Could detritus be sustaining the fishery? The level of detritus in San Miguel Bay may have increased. Siltation in San Miguel Bay, from the river systems, loss of mangroves, and mining and quarrying activities has increased over at least the last few decades (Mendoza and Cinco 1995). The main influence on the siltation is the Bicol River. If it is assumed that the silt carries organic matter, then this would lead to an increase in detritus. This steady increase in detritus may have promoted productivity in the fishery, especially in those eco-groups which are sustained by the detritus flows, as suggested by Pauly (1982a). However, Mendoza and Cinco (1995) also note a large list of negative effects of siltation such as modification of bottom topography and clogging and abrasion of gills. In reality then, the positive impact of an increase in detritus may be modified by the negative physical effects of siltation.

A more parsimonious explanation for the continuance of the fishery at current levels, and one that bears some consideration, is that the species in San Miguel Bay are continually replenished by larvae from outside the Bay. That is, San Miguel Bay is not a closed system. Pauly (1982b) suggested that San Miguel Bay acts as a nursery area. Pauly cites several pieces of evidence to support this hypothesis, which include the lack of larvae in plankton trawls during likely spawning periods (Weber 1976). Length data also indicated that larger fish are caught outside the Bay than inside (Pauly 1982b). Pauly schematically suggested that mature fish emigrate from the Bay, spawn outside the Bay, the larvae are carried into the Bay by tidal currents and gyres, they migrate towards the shallow coastal or nursery areas where the

juveniles mature. As the juveniles mature they move into deeper water until they eventually leave the Bay as adults (1982b, Figure 4).

There is also other evidence to support the emigration/immigration hypothesis. In the catch data presented in Chapter 2, a large proportion of the sciaenids and engraulids, and to a lesser extent the leiognathids in the catch were below the length of first maturity. If fish are caught before they are mature, it is difficult to conceive how they could reproduce if there was no import from outside the Bay⁷¹. However, there were also mature fish caught, indicating that not all mature fish emigrate from the Bay, or that they do not immediately emigrate. If emigration occurs, it could account for the high estimates of apparent Z in Chapter 2.

Omori (1975) suggests that there is immigration by sergestids since they swarm and go offshore. Some penaeids also move offshore to spawn (Garcia 1988, Motoh 1981). Tiews and Carces-Boya (1965) conjecture, on the basis that no spawning leiognathids are caught by the fishery in Manila Bay, that mature leiognathids go offshore to spawn. However, there is also evidence that some species do not migrate offshore to spawn. In the Ragay Gulf (Philippines) for example, berried females of *Portunus pelagicus* are found inside the Gulf. (Ingles and Braum 1989).

Johannes (1978) discusses the reproductive strategies of tropical marine fish at some length. He identifies five types of spawners, including migrating spawners. Carangids, sphyraenids, serranids, lutjanids and leiognathids, all fish species found in San Miguel Bay, are described as

⁷¹ However, an alternative explanation for this is that since M is very high below L_{mat} , the fishery is only taking fish that would have died anyway.

migrating spawners. Johannes further delineates a size category. Fish greater than 25cm are likely to migrate to spawn and those less than 25 cm are likely to spawn inshore. The L_{∞} of the engraulid and leiognathid species in San Miguel Bay is less than 25cm. However, Johannes emphatically states that he has not considered the tropical fish of predominantly sandy and muddy continental shelves “because too little is known about their reproductivity” (1978:65).

This leaves a question mark concerning the issues of immigration and emigration. It is more likely than not that immigration and emigration occur, at least to some extent, for some species. The presence or absence of imports to the Bay profoundly affects our thinking about the fishery. If fish spawn outside the Bay, there is a continual supply of eggs into the Bay⁷². Thus the Bay is stocked from outside, allowing high production and catches. What are the implications for the San Miguel Bay ECOPATH model? And what are the management implications?

An essential assumption of any ECOPATH model is that there is greater similarity within the defined system than between the defined system and an outside system. In other words, the interactions within the system should add up to a greater flow than the interactions between the system and the outside (Christensen and Pauly 1992a). The sum of all flows in San Miguel Bay, that is, the throughput, is comparable to other systems. It could be argued, if emigration occurs, that fishing mortality is so high inside San Miguel Bay that the fishery competes for the flows to the outside. That is, the fishery catches the mature fish before they emigrate. Thus export is retained within the model as fishing mortality, and there is no or little flow to the

⁷² The area outside the Bay might thus be considered a refuge.

outside.

It is not feasible to model larval import or immigration using ECOPATH. This is because ECOPATH is a biomass model and import is modelled as negative export. Thus to model import alone, a negative value would be input for the export. But the biomass of the larval import would be very low, perhaps negligible. To model import and export, the difference between the biomass of the import and export would have to be input. This would be effectively the same as entering export only.

On the premise, then, that fishing captures most mature fish before they emigrate from San Miguel Bay (which is likely), that if any fish do “escape” the sum of the interactions between San Miguel Bay and the export is less than the sum of the interactions within San Miguel Bay, and that import cannot be effectively modelled using ECOPATH, it is assumed that the models described above are still useful and relevant, even if immigration and emigration do occur.

In summary, the ECOPATH models indicate that predation and competition in the biological community are important determinants within the dynamics of the ecosystem. The results also indicate that competition and predation interact with the effects of fishing mortality. Thus not all species which are caught by a fishing gear necessarily decrease as a result of an increase in fishing mortality. Different fishing gears have different impacts on the fishery and these are not simply a linear consequence of the fishing activity.

It is not possible to investigate the effects of predation, competition and fishing mortality further with ECOPATH. ECOPATH represents a static system. It is not dynamic and it is not quantitative. The trophic impact routine can only indicate direction of change. It does not indicate the degree of change necessary to produce a response, nor the degree of the response, nor the time dynamics of the response. The trophic impact routine cannot predict how biomass will change with time. These questions are investigated further using a fully dynamic ecosystem model, ECOSIM (Walters *et al.* 1997) in Chapter 4. With ECOSIM it is possible to quantitatively analyse the community dynamics and fishing dynamics in San Miguel Bay and to explicitly address the interactions between them. The question of immigration and emigration is returned to in Chapter 5.

Chapter 4

Dynamic Multispecies Modelling of San Miguel Bay

“With the rising status of the Third World and rapid growth of its tropical fisheries, the limitations of the classical single species approach are becoming more apparent”
Sugihara *et al.* 1984:132

Introduction

Tropical fisheries are reknown for their complexity and the multiplicity of species which co-exist in these productive marine ecosystems. The problems involved in assessing and managing these multispecies, multigear fisheries were introduced in Chapter 2, and an equilibrium biomass ecosystem model, ECOPATH II was used in Chapter 3 to gain an understanding of the fishery of San Miguel Bay as a fishery of an ecosystem. However, a static representation of an ecosystem does not allow the asking of “what if?” questions. For example, what would happen in San Miguel Bay if the trawling sector were banned completely? How many years would the ecosystem take to recover? ECOPATH was pushed to its limits to try to answer these types of questions, but the equilibrium nature of the model restricted the answers to mere indications of the likely direction of change. It was not possible to determine how species interactions would behave through time and how these might effect outcomes for the fishery. In order to address these types of “what if?” questions, a multispecies dynamic modelling approach is necessary.

Multispecies approaches have been minimally applied in fisheries management, despite the recognition by many that a multispecies approach to fisheries assessment is critical (Walters *et al.* 1997, Christensen 1996, Hilborn and Walters 1992, Daan and Sissenwine 1991, Kerr and Ryder 1989, Pauly and Murphy 1982, Larkin and Gazey 1982, and Mercer 1982). There is however, a range of multispecies approaches. They include: multispecies biomass models (eg, Kirkwood 1982, May *et al.* 1979); aggregate production models (eg, Ralston and Polovina 1982, Pope 1979); multispecies yield per recruit analysis (eg. Murawski *et al.* 1991, Wilson *et al.* 1991), and multispecies VPA (eg, Pope 1991, Pope and Macer 1991, Sparre 1991) and empirical methods (eg., Sainsbury 1991, 1988, Saila and Erzini 1987).

The aggregate production model, simply totals the biomass of all species, ignores interactions between species and treats them as though they were a coherent whole. Ralston and Polovina (1982) vary this by first using cluster analysis to identify species assemblages and then treating each assemblage as a whole. This approach requires only catch and effort data for the fishery. The other multispecies methods listed above all have large parameter demands. This includes the multispecies biomass model which requires interaction terms between species: the more species in the model, the greater the demands.

But multispecies modelling is a data hungry exercise. The sparse use of a multispecies approach in fisheries science and management is due in large part to their data demands⁷³. One of the few, if not the only multispecies VPA, is of the North Sea (Pope 1991): compared to tropical fisheries, this is a relatively simple ecosystem. None of the above models are

⁷³ It is also due to the economic and political consequences of multispecies assessment (May *et al.* 1979), which can confuse and confound an already complex management situation.

applicable to the type of data available from San Miguel Bay. There is no time series of catch and effort data with which to fit production or biomass models and the data simply do not exist for the other methods. The direct empiricism of Sainsbury (1991, 1989), based upon the principles of adaptive management, (Walters 1986) is a possible approach and is discussed in Chapter 5.

As early as 1979, May *et al.* demonstrated the potential effects of species interactions on yield estimates. They used direct interaction terms between species, and assumed a Lotka-Volterra model of predation. However, the Lotka-Volterra model is not the only model of flow control. Indeed, there is no unifying paradigm of how energy flow between trophic levels of an ecosystem is governed. Two theories, top-down control with trophic cascades (Carpenter *et al.* 1985, Carpenter and Kitchell 1993) and bottom-up, donor control (Hall *et al.* 1970, Hunter and Price 1992), have dominated the ecological literature (Matson and Hunter 1992). However, the notions of bottom-up and top-down control are not well-defined.

Operationally, a bottom-up regime is defined by a response to changes in productivity at the lowest trophic levels. An increase in productivity at the bottom of the trophic system leads to an increase in the productivity and abundance at all higher trophic levels. Each trophic level is food limited (Power 1992), not predator limited. This is an important concept in bottom-up control. That is, an increase in the abundance of a predator species, does not lead to an increase in the mortality rate on the prey. Predation is proportional to the biomass of the prey, flow is controlled by prey abundance and the mortality rate is stable over time. An increase in the

abundance of a top predator would not impact the rest of the ecosystem because the mortality rate would remain stable.

The operational definition of top-down control, is that an increase in the abundance of a top-predator leads to the prediction of a trophic cascade. That is, an increase in the abundance at trophic level 4, leads to an increase in the rate of predation mortality on trophic level 3, and thus a reduction in its biomass. This produces a decreases in the predation by trophic level 3 on 2, and thus to an increase in the biomass of trophic level 2. And so on. Here, flow is predator controlled. An increase in productivity at the bottom would lead only to an increase in biomass at the top of the food chain. The biomass of lower levels would not increase because the mortality rate incurred by them would increase due to an increase in the abundance their predators. Only their productivity would increase.

The terms top-down and bottom-up control, are used here in the sense described above (see also Walters *et al.* 1997). Much of the debate over top-down versus bottom-up control has occurred in the self-contained world of fresh water lake systems. In the marine world, the picture is no less murky. Until recently, few marine studies contributed much further to the top-down, bottom-up debate. Those that did commonly supported the top-down control theory. However, some recent work has shown evidence for bottom-up control in some systems, such as benthic marine communities and rocky shores (Menge 1992).

The debate has moved on and it is recognised that top-down and bottom-up control are both likely to act on ecological communities (Neill, *in press* , Hunter and Price 1992, Matson and

Hunter 1992, Menge 1992, Powers 1992). Questions are being posed instead as to the nature of the links between bottom-up and top-down influences. Hunter and Price (1992) propose that the variation in flow types observed in ecosystems is a consequence of heterogeneity of communities, ecosystems and species interactions.

The nature, or type, of flow dynamics in San Miguel Bay, is unknown. San Miguel Bay is a large muddy bay with a broad mixture of species, demersal and pelagic, vertebrate and invertebrate. The species composition of the Bay has changed over the last few decades (Chapter 2), and it is possible that the flow dynamics have also changed. To date though, no empirical studies have been undertaken to examine this issue. The ECOPATH model in Chapter 3 showed evidence for both flow hypotheses.

Despite the relative paucity of data with which to make a multispecies analysis and assessment of San Miguel Bay, this has been made possible with the development of a new multispecies model, ECOSIM (Walters *et al.* 1997). ECOSIM was developed from a simple mass-balance trophic model. ECOSIM proposes mechanisms for top-down and bottom-up control theories and it requires only a few more parameters than the ECOPATH model used in Chapter 3. ECOSIM is used here to explore the community dynamics of San Miguel Bay, the interactions between fishing, the ecosystem and species interactions and to examine certain “what if?” questions. Throughout the analysis, simulations were made for both top-down and bottom-up assumptions and the results were contrasted.

Methods

ECOSIM

ECOSIM (Walters *et al.* 1997) was developed from a mass-balance model. Walters (1996) recognised that the linear equations which describe the trophic fluxes in mass-balance, equilibrium assessments of ecosystems (for example, ECOPATH II) could be replaced by dynamic equations once the model was balanced. When an equilibrium model is balanced (see Chapter 3), the linear equations can be re-expressed as differential equations equivalent to changes in biomass through time. Perturbations to the equilibrium state are made by changing the exploitation regime. Thus without any extra parameters, the equilibrium ecosystem model is transformed into a dynamic ecosystem model and the impact of changes in exploitation can be examined for each component of the ecosystem.

The key step in the elaboration of ECOSIM was the replacement of the static consumption flows with functional relationships between consumption and biomass of predators and prey. The basic ECOPATH II linear equation is,

$$B_i.PB_i - \sum_j B_j.QB_j.DC_{ji} - PB_i.B_i(1 - EE_i) - F_iB_i - EX_i = 0$$

where,

B_i = Biomass of (i), PB_i = Production/Biomass ratio of (i), QB_i = Consumption/Biomass ratio of (i), DC_{ji} = Proportion of (i) in the diet of (j), $(1-EE_i)$ = other mortality of (i), EX_i = Export of (i).

This is equivalent to,

$$g_i \sum_j Q_{ij} - F_i B_i - M_o B_i - \sum_j Q_{ji} = 0$$

where,

Q_{ji} = consumption of (i) by (j) and

$$g_i \sum_j Q_{ij} = B_i . PB_i$$

and, making the equation a differential,

$$\frac{dB_i}{dt} = g_i \sum_{j=1}^n c_{ij} . (B_i, B_j) - M_o B_i - F_i B_i - \sum_{j=1}^n c_{ji} (B_i, B_j)$$

where,

$c_{ij}(B_i, B_j)$ is a function used to predict consumption, Q_{ij} , from the biomass of the prey (i) and the predators (j).

Initially, for simplicity, in the development of ECOSIM, the Lotka-Volterra mass action equation was used as the functional relationship, where

$$c_{ij}(B_i, B_j) = a_{ij}B_iB_j$$

and,

a_{ij} is the instantaneous mortality on the prey (B_i) caused by one unit of predator biomass, (B_j), or, in ecological terms, the rate of effective search.

The parameter a_{ij} is determined by solving for a_{ij} using the initial equilibrium model parameters. However, the Lotka-Volterra equation has only top-down control assumptions, and it does not take spatial or behavioural limiting mechanisms into account. In reality, there is likely only to be a certain amount of prey available to a predator at any given time, due to predator avoidance behaviour by prey. In ECOSIM this is modelled as “vulnerability to predation”, and provides one specific way to model bottom-up control of biomass flow.

In this case, the functional relationship is

$$c_{ij}(B_i, B_j) = \frac{a_{ij}v_{ij}B_iB_j}{2v_{ij} + a_{ij}B_j}$$

where,

v_{ij} = the maximum instantaneous mortality rate that B_j can exert on B_i .

Two extra parameters, a_{ij} and v_{ij} , are required in addition to the initial equilibrium model parameters. The maximum instantaneous mortality rate, v_{ij} , is calculated from user input and the equation then solved for a_{ij} .

With this functional relationship, ECOSIM is able to emulate the two energy flow control theories, top-down or bottom-up control. Top-down relationships are simulated by making v_{ij} , the “vulnerability factor” large, meaning that a large part (or all) of the prey biomass is vulnerable to predation. Bottom-up, donor control is simulated by making the value of v_{ij} very low. This limits the amount of prey that are vulnerable to predation (thus keeping the mortality rate close to stable). In ECOSIM, when bottom-up, donor control is simulated by making vulnerability very low, the mortality rate is largely independent of the abundance of the next trophic level and for top-down control, the mortality rate is strongly dependent on the abundance of the next trophic level.

The user thus determines the nature of the flow relationships within the ecosystem, and can experiment with the effects of different assumptions. When consumer biomass is low, the relationship is reduced to a Lotka-Volterra mass action flow and when consumer biomass is high, the relationship is donor control type.

In addition to the main functional relationship for consumers, functional relationships were developed for the producers in the ecosystem and for consumption and production by the detritus (see Walters *et al.* 1997 for further details).

The Delay-Differential Model

In the ECOSIM model described, each pool is composed of all age classes from juveniles to adults. Early work with the model unveiled a considerable shortcoming of this approach. Top

predators were seen to increase unrealistically in response to decreased fishing mortality.

These groups, first fully consumed the adult food groups, then were sustained by the invertebrate food of the juveniles. The model had not accounted for the different diets of juveniles and adults, that is, trophic ontogeny.

This was addressed by including the option to split pools into adults and juveniles (Walters *et al.* 1997). In the split pools, numbers of individuals are tracked in addition to biomass. The adult pool receives numbers and biomass from the juvenile pool and numbers of juveniles are recruited from the adults. Each juvenile/adult pair is represented by five differential equations, based on Beverton and Holt type equations. In essence the model is akin to the delay-difference equations of Deriso(1980): the juvenile pool has a mean age and weight and the juveniles grow to the age of maturity when they become adults.

Three additional parameters are required for the “delay-differential” model, W_k , the weight at which juveniles becomes adults, T , the age at which juveniles become adults and K , the von Bertalanffy growth parameter. The advantages of the split pool model is that it allows for the effects of both trophic ontogeny and cannibalism by adults on juveniles.

Running the Model

All that is required to run ECOSIM is the output from a balanced equilibrium mass balance model such as ECOPATH II (and the split pool parameters). The model has two main modes of operation, the “Equilibrium Fishing Routine” and the “Dynamic Run Routine”.

The Equilibrium Fishing Routine.

Fishing mortality, F , is varied incrementally and the equilibrium biomass of each pool in the model is estimated for each value of F . A Newton search method is used to find the equilibria by searching for zeros in the equations outlined above. F can be varied in three ways: on a single pool, for selected fishing gears or over total F in the fishery. The equilibrium catch for one pre-specified group is also calculated and both biomass and catch are plotted against F (see figure 4.2).

The Dynamic Run Routine.

The biomass of each pool in the ecosystem is estimated through time in response to user imposed changes in the fishing pattern (see figure 4.9). The user can change the fishing mortality of an individual group or an individual gear or of all gears in the fishery. This is done graphically. Biomass is calculated using a fourth order Runge-Kutta numerical integration scheme.

Analyses

The community dynamics of San Miguel Bay and the impacts of fishing on the ecosystem were examined in several ways using ECOSIM. Throughout, the results of the top-down and bottom-up assumptions about energy flow are contrasted. Bottom-up control was simulated using a vulnerability factor (vf) of 1.5, and top-down control by a vf of 10. Intermediate values were obtained using a vf of 4, and extreme top-down values obtained using a vf of 50. The scale and legend for Figures 4.2 - 4.10 and 4.14 are given in Table 4.1.

Impacts on Equilibrium Biomass of Changing Fishing Mortality Across All Fishing Gear

The Equilibrium Fishing Routine was used to investigate the gross effects of total fishing mortality on the ecosystem and its response to change in F . Total fishing mortality (that is, fishing mortality summed over all gears) was incrementally reduced and increased from the current value. In addition to the top-down and bottom-up runs, two additional runs were made for an intermediate assumption of flow control and an extreme top-down assumption.

The Equilibrium Yield Curves, Species Interactions and Flow Dynamics

In single species analyses, yield curves have played an important role in determining the status of fisheries and in helping to direct management. Here, three types of yield curves are compared: the single species yield curves from Chapter 2; yield curves produced by the Equilibrium Fishing Routine that include the effects of fishing on the whole fishery and yield curves produced by the Equilibrium Fishing Routine that only include the effects of fishing on one pool.

Yield curves which include the effects of fishing on the whole fishery (yield curve - *plus fishing*) are produced by the routine described above. The shape of the yield curve is influenced by the fishing mortality to which the pool is subject, indirectly by the fishing mortality that the other pools are subject to via species interactions, plus the assumptions made about flow dynamics in the ecosystem. The fishing mortality exerted on all pools changes simultaneously. Yield curves which include only the effect of fishing on one pool

(yield curve - *no fishing*) are produced when one pool is selected and fishing mortality varied only on that pool. The shape of this yield curve is influenced by the specific fishing mortality for that pool and the assumptions made about flow dynamics. The fishing mortality on all other pools remains at the current fishing mortality rate. For the single species yield curves from Chapter 2, the only process effecting the shape of the yield curve is the direct effect of fishing mortality on that species.

Multispecies Dynamics in the San Miguel Bay Ecosystem

The flow diagram of the San Miguel Bay (Figure 3.1) portrayed a hierarchically organised food web, ranging from trophic level 1 to 4. It also showed that many of the pools were clustered around trophic level 3, indicating likely competitive interactions as well as predation links between trophic levels. The extent and strength of these interactions, and thus their dynamics, could only be conjectured in Chapter 3. Here, it is possible to examine these interactions using the biomass results of the single pool runs of the Equilibrium Fishing Routine above, under both bottom-up and top-down assumptions. It is then possible to question and examine the association of these interactions with the effects of fishing.

The Effects of Fishing in San Miguel Bay

The impact of fishing was directly examined for every gear using the Dynamic Run Routine. The fishing mortality of each gear was reduced to zero over the first 2 years and the simulation allowed to run for another 8 years (Figure 4.1), by which time most biomass transients had

stabilised (in cases where they had not, the simulation was run for 20 years as a check). Their impacts on the ecosystem was assessed in two ways, (i) the number of pools impacted by the gear and the magnitude of that impact, and (ii) the effective change in total biomass in the ecosystem.

Consistency Checks

A routine within ECOSIM allows the user to check the effect of the above perturbations on the consumption/biomass ratios and the mean weights of the split pools. This is a means to check the validity of results. Extreme values of either parameter would indicate that there is internal inconsistency in the model.

The simulations were ran as described above with the exception of the Equilibrium Yield Curves-*no fishing*. The effect on QB and mean weight of changing the biomass of individual pools was checked using the Dynamic Runs routine. The fishing mortality on each pool was reduced to zero over the first 2 years of a 10 year simulation, as in the dynamic runs where fishing effort was reduced to zero. This method was used for efficiency and because it gives the results as transients through time as opposed to equilibrium values.

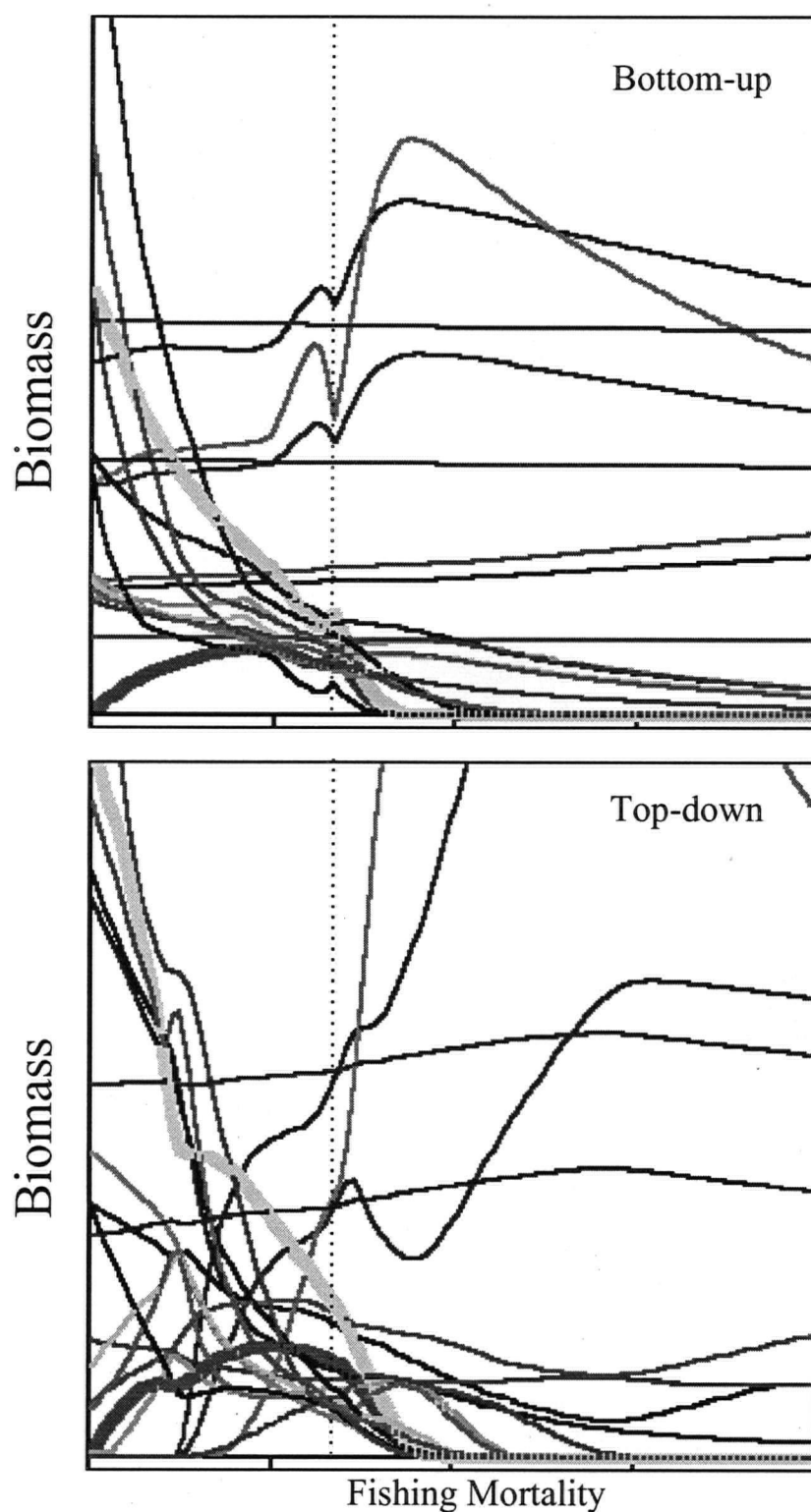


Figure 4.2: Equilibrium simulation of changing total fishing mortality for bottom-up and top down control. Thick red line indicates yield curve (for juvenile scieanids), coloured lines, biomass, dotted black line, current fishing mortality. See text for more details and Table 4.1 for legend and scale.

The Input Parameters

The initial data used for the ECOSIM model was the output runfile from the ECOPATH model used in Chapter 3. To examine the effects of fishing by different gear types in the fishery, an additional file containing a matrix of the fishing mortality imposed on each pool by each gear type was also necessary. The twenty six gear types listed in Table 2.14 were combined into nineteen and included in the model. The large, medium and baby trawlers were combined simply as large scale trawlers (in contrast to the mini-trawlers) and the ring net, pullnet, stationary tidal weir, beach seine and spear gun were included as “others”. Fishing mortality was calculated as catch (of pool a by gear x) over biomass (of pool a).

Splitting Pools into Adults and Juveniles

Early work with ECOSIM revealed that a “split pool” approach would be required. Of the 16 groups described in the San Miguel Bay ecosystem, eight are fish groups where the differences in diet of adults and juveniles of the same species might impose differential mortality effects on trophic groups lower in the ecosystem. However, only three of these, the sciaenids, the medium predators and the large predators are likely to show any significant trophic ontogeny. Since the number of pools in the ECOPATH model is already quite large, only these three groups were split in to adult and juvenile pools.

For the three split pools, each of the ECOPATH input parameters had to be re-calculated for both the adults and the juveniles, that is production/biomass, consumption/biomass, ecotrophic efficiency, exports/harvests, and diet. The new ECOPATH model was then rerun and balanced. Since there were no explicit data available for the juveniles distinct from the adults, some assumptions had to be made on the basis of the available data - full details are given in Appendix 2. Finally a file containing the split pool parameters, W_k , T and K for each of the split pools was required.

Results

Impacts on Equilibrium Biomass of Changing Fishing Mortality Across All Fishing Gear

The overall impact on the San Miguel Bay fishery of simultaneously increasing or decreasing fishing pressure over all fishing gears is shown in Figure 4.2. The results clearly show that, regardless of assumptions made about the control of energy flow, the biomass of almost all fished groups in the ecosystem is currently less than 50% of their biomass when no fishing mortality is incurred. The most striking feature of both plots is the profusion of coloured biomass lines on the left hand side of the graph, indicating abundance and complexity compared to the much simpler array on the right. The results predict that if fishing mortality were further increased, the fish groups would be replaced by the crustacean groups. The fishery would thus be reduced to essentially a crustacean fishery (a highly productive

Table 4.1 Legend and Scale for Figures 4.2 - 4.10 and 4.14 - 4.15.

| <i>Ecosim Pool</i> | <i>Equilibrium Graph Maximum</i> | <i>Dynamic Run Maximum</i> |
|---------------------------|----------------------------------|----------------------------|
| Zooplankton | 4.0 | 10 |
| Meiobenthos | 40.0 | 1 |
| Macrobenthos | 40.0 | 1 |
| Sergestidae | 0.4 | 100 |
| Penaeidae | 4.0 | 10 |
| Large Crustaceans | 4.0 | 10 |
| Juvenile Sciaenids | 4.0 | 10 |
| Juvenile Medium Predators | 4.0 | 10 |
| Juvenile Large Predators | 0.4 | 100 |
| Demersal Feeders | 4.0 | 10 |
| Leiognathids | 4.0 | 10 |
| Engraulids | 4.0 | 10 |
| Pelagics | 4.0 | 10 |
| Sciaenids | 4.0 | 10 |
| Medium Predators | 4.0 | 10 |
| Large Zoobenthos Feeders | 4.0 | 10 |
| Large Predators | 0.4 | 10 |
| Phytoplankton | 40.0 | 10 |
| Detritus | 40.0 | 10 |

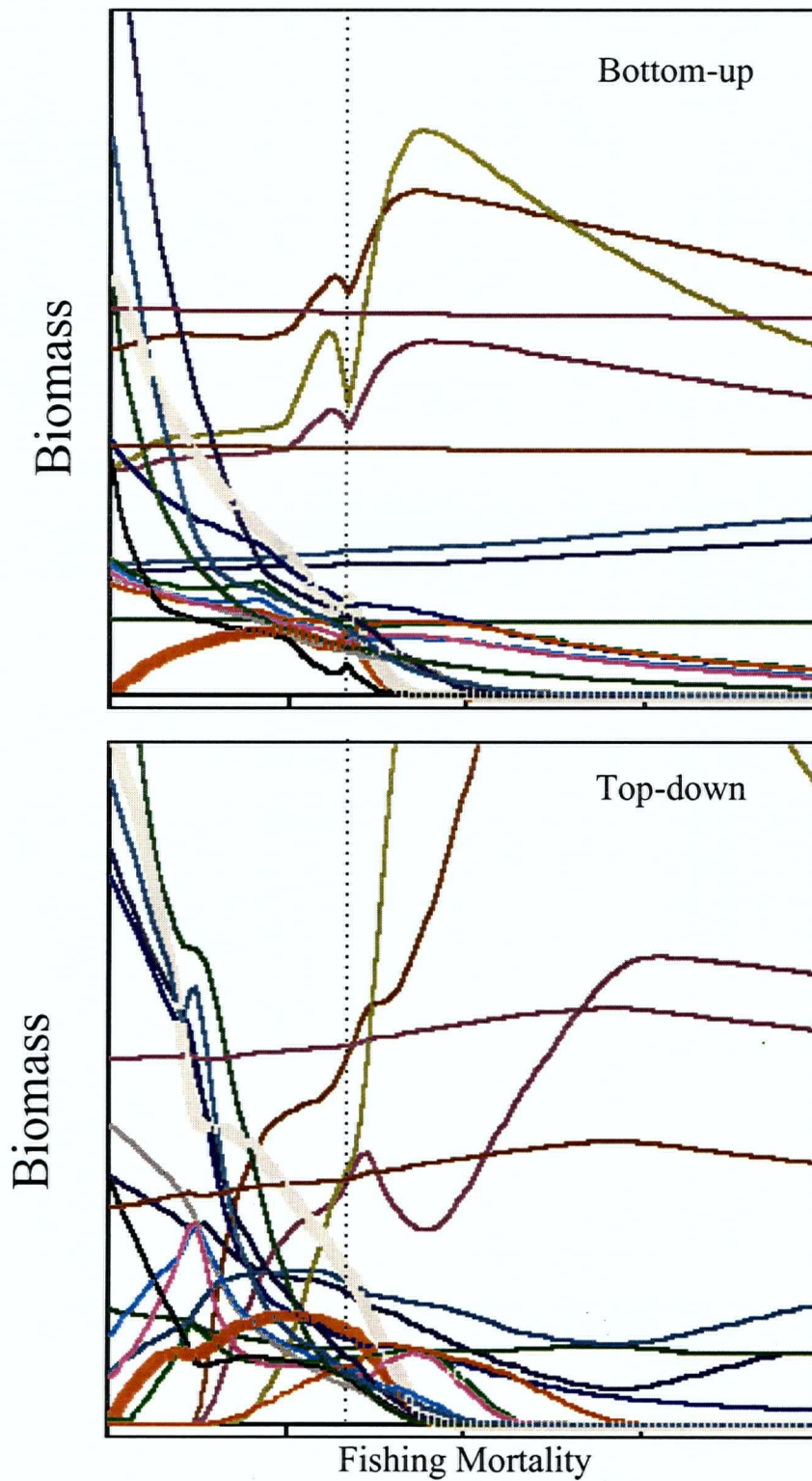


Figure 4.2: Equilibrium simulation of changing total fishing mortality for bottom-up and top down control. Thick red line indicates yield curve (for juvenile scieanids), coloured lines, biomass, dotted black line, current fishing mortality. See text for more details and Table 4.1 for legend and scale.

crustacean fishery in the case of the top-down assumption). These results once more emphasise that San Miguel Bay is suffering from ecosystem overfishing. The results also show that a reduction in fishing mortality would enable recovery of biomass across most groups.

Although the broad equilibrium response to fishing mortality is similar for the bottom-up and top-down control assumptions, there are differences. There are also differences in the responses of individual pools.

The top-down assumption produces more drastic changes in biomass than the bottom-up assumption: rates of biomass change are faster and the range of fishing mortality over which these changes occur, narrower. Thus biomass increases faster and higher, and decreases faster and lower. For some groups, such as the large crustaceans, the sergestids, the penaeids and the engraulids, this means that for low values of fishing mortality, their biomass is negligible. On the other hand, at high fishing mortality, the three crustacean groups are very productive. They suffer minimal predation, since all higher groups are virtually fished out. Pauly (1979a) predicts this increase in benthic invertebrates as a consequence of overfishing. This crustacean biomass increase is also seen, although less dramatically, under the bottom-up control assumption. Their response is different however when fishing mortality is reduced. Here, these crustacean groups remain at relatively high biomass levels even when the biomass of their predators also increases in response to the reduction in fishing mortality.

Bottom-up control produces a more stable system that is more resilient to change. Generally, when fishing mortality is reduced, the biomasses do not rise to the levels seen in the top-down

simulation^{74,75}. However, the biomass of the leiognathids, the pelagics, the demersal feeders, the engraulids and the large zoobenthos feeders persist over a greater range of high fishing mortality. This is because, under the bottom-up assumption, production at lower levels controls abundance at higher levels. All of these groups feed on the crustacean pools which increase in abundance as fishing mortality is increased. Thus the biomass of these groups persist, despite increased fishing pressure. The lack of persistence of these pools under the top-down control assumption is due to their intolerance to predation mortality and the increased fishing mortality. The biomass curve of the pelagics and the demersal feeders has a small second peak (top-down control) to the right of current F . These small peaks could be due to a release from predation pressure by the predator pools.

Under both control scenarios, the biomass of the sciaenids, the medium predators and the adult large predators is reduced to an extremely low level when fishing mortality is increased much above the current fishing mortality. Conversely, when fishing mortality is reduced, their biomass increases quite rapidly. Surprisingly, the juvenile large predators do not follow this pattern. Their biomass persists at high levels of fishing mortality.

⁷⁴ In the top-down equilibrium fishing simulation, the biomass plots of the leiognathids, demersal feeders and pelagics initially increase in response to decreased fishing mortality, but subsequently, as F is further decreased, their biomass decreases. This would appear to be the result of predation by the higher trophic groups which continue to increase their biomass. However, trawl data (Warfel and Manacop 1950) indicate that the leiognathids comprised over 60% of the trawl biomass in the late 1940s (Table 2.6), when fishing effort was at a comparatively low level. It seems unlikely then that at low fishing mortality, the predators in the ecosystem would increase to such a high biomass as predicted by ECOSIM. Walters *et al.* (1997) note that the ECOSIM is unlikely to reasonably predict the ecosystem behaviour when extreme values of F are used since the model parameters originate in an equilibrium state assumption. The extreme results of the ECOSIM simulations should thus be interpreted with caution.

⁷⁵ When the equilibrium fishing simulation was run under the bottom-up control scenario, a runtime occurred. This was due to the sciaenids. The biomass of the juvenile sciaenids, after steadily increasing when F was decreased, suddenly dropped at very low F , crashing vertically whilst the adult sciaenids biomass exponentially increased. Further explanation is given in the text below of this phenomenon. In Figure 4.2, an intermediate value of $\nu_f=4$ was used for the juvenile sciaenids, in order to conduct the bottom-up simulation.

The pools at the lowest trophic levels, the phytoplankton, zooplankton, meiobenthos.

macrobenthos and detritus are very stable when bottom-up control is simulated and respond to predation by pools higher in the food web when top-down control is simulated.

For comparative purposes, two more simulations were made, one assuming that neither top-down or bottom-up control predominates, but that in reality, there is an intermediate level of control. The other simulation assumes much stronger top-down control. The intermediate control simulation produced results, Figure 4.3 (a) which are in fact a kind of half-way house between those described above. The strong top-down simulation, Figure 4.3(b), results in a biomass plot which resembles Figure 4.2, but shows less stability. For example, the penaeids decrease almost to zero and then take-off at high levels of fishing mortality. Generally there is greater variation in biomass.

The results of this multispecies equilibrium analysis demonstrate that there is not simply an increase in all groups in the ecosystem when fishing mortality is decreased and a decrease in all groups when fishing mortality is increased. In addition to the effects of fishing mortality, the effects of species interactions (predation and competition) and the assumptions made about flow dynamics are evident. The interplay of fishing mortality, species interactions and flow dynamics has profound implications for fisheries assessment and management.

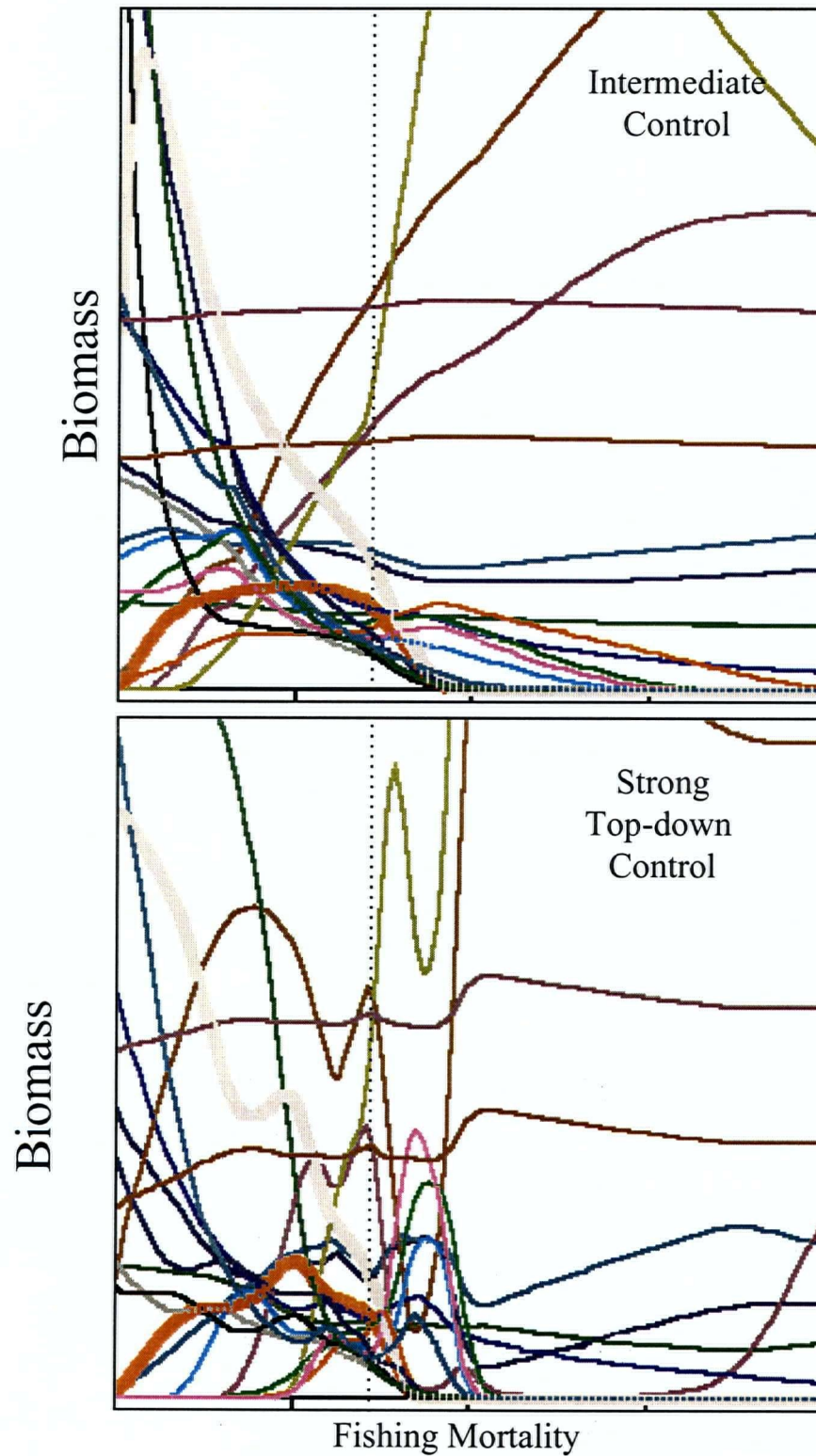


Figure 4.3: Equilibrium simulation of changing total fishing mortality for intermediate control and strong top-down control. Thick red line indicates yield curve (for juvenile scieanids), coloured lines, biomass, dotted black line, current fishing mortality. See Table 4.1 for legend and scale.

Equilibrium Yield Curves, Species Interactions and Flow Dynamics

The interactive effects of fishing mortality, species interactions and flow dynamics are perhaps even more evident in the equilibrium yield curves. Yield curves were analysed with and without the effects of fishing on other pools in the ecosystem, in addition to the effect of multispecies interactions. Yield curves without the effects of fishing on other pools (yield curve-*no fishing*) were produced by varying fishing mortality on only one pool. Yield curves that include the effects of fishing on other pools (yield curve-*plus fishing*) were produced by varying fishing mortality across the whole fishery. The equilibrium yield was calculated from the equilibrium biomass for each pool in Figure 4.2. The latter is the more realistic scenario in a multispecies, multigear fishery. Although the yield curve-*no fishing* allows the analysis of the effect of fishing mortality on one pool, it would not be possible in practice to selectively fish on only one pool. Table 4.2 compares results of the multispecies equilibrium yield curves with the single species curves (Chapter 2).

Some of the pools in the model exhibit significant differences in the shape of their yield curves, depending on whether fishing on other pools is included or not.

The (yield curve-*no fishing*) for the pelagics indicates a severely overfished fish group when top-down flow dynamics are assumed (Figure 4.4b), as does the single species yield curve for *Scomberomorus commerson* (41% of pelagic trawl biomass). A bottom-up assumption results in the conclusion that current fishing mortality is at around the optimum level⁷⁶. However, the

⁷⁶ Here optimum fishing mortality is interpreted as the area from the left of the top of the yield curve to the top of the curve, roughly comparable to the MEY and MSY points.

Table 4.2 Comparison of yield curves and current fishing mortality produced by a single species yield-per-recruit analysis and a multispecies analysis.

| ECOSIM Group | Single Species yield-per-recruit | Multispecies Yield Curves | | | |
|--|--|--|----------------------------------|---------------------------------|---------------------------------|
| | | Yield Curve-no fishing | | Yield Curve-plus fishing | |
| | | Bottom Up | Top Down | Bottom Up | Top Down |
| Sergestids | n/a | Fcurr. < Fopt | Fcurr. ~ Fopt | Fcurr. < Fopt | Fcurr. < Fopt |
| Penaeids | n/a | Fcurr. < Fopt | Fcurr. ~ Fopt | Fcurr. < Fopt | Fcurr. < Fopt |
| Large Crustaceans | n/a | Fcurr. < Fopt | Fcurr. ~ Fopt | Fcurr. < Fopt | Fcurr. < Fopt |
| Demersal Feeders | n/a | Fcurr. ~ Fopt | Fcurr. > Fopt. | Fcurr. < Fopt | Fcurr. < Fopt |
| Leiognathids <i>Leiognathus splendens</i> <i>Leiognathus bindus</i> <i>Secutor ruconius</i> | Fcurr. > Fopt. Fcurr. > Fopt. Fcurr. ~ Fopt. | Fcurr. ~ Fopt. | Fcurr. > Fopt. | Fcurr. < Fopt | Fcurr. > Fopt. |
| Engraulids (<i>Stolephorus commersonii</i>) | n/a | Fcurr. ~ Fopt. | Fcurr. > Fopt. | Fcurr. < Fopt | Fcurr. < Fopt |
| Pelagics (<i>Scomberomorus commerson</i>) | Fcurr. > Fopt. | Fcurr. ~ Fopt. | Fcurr. > Fopt. | Fcurr. < Fopt | Fcurr. < Fopt |
| Sciaenids (<i>Otolithes ruber</i>) | Fcurr. > Fopt. | J. ? A. Fcurr. ~ Fopt. | Fcurr. > Fopt. Fcurr. ~ Fopt. | Fcurr. > Fopt Fcurr. ~ Fopt. | Fcurr. > Fopt Fcurr. ~ Fopt. |
| Medium Predators (<i>Trichiurus haumela</i>) | Fcurr. > Fopt | J. Fcurr. ~ Fopt. A. Fcurr. ~ Fopt. | Fcurr. ~ Fopt. Fcurr. ~ Fopt. | Fcurr. > Fopt. Fcurr. > Fopt | Fcurr. > Fopt Fcurr. > Fopt |
| Large Zoobenthos Feeders | n/a | Fcurr. > Fopt | Fcurr. > Fopt | Fcurr. > Fopt | Fcurr. > Fopt |
| Large Predators | n/a | J. Fcurr. ~ Fopt A. Fcurr. < Fopt | Fcurr. > Fopt Fcurr. ~ Fopt | Fcurr. < Fopt. Fcurr. > Fopt | Fcurr. > Fopt Fcurr. > Fopt |

Fcurr = current fishing mortality, Fopt = optimum fishing mortality

yield curve-*plus fishing* produces quite different conclusions. For either flow assumption, the curve describes two peaks, one at a fishing mortality below the current fishing mortality and one at a fishing mortality higher than current fishing mortality (figure 4.4). The second peak results from the small increase in biomass at higher fishing mortality described above. If the optimum fishing mortality is associated with the highest peak, current fishing mortality is below optimum fishing mortality for both flow assumptions. The implication of this is that if fishing effort were increased across the fishery, there would be a greater return of pelagics. A comparison of the pelagic catch between 1979-1982 and 1992-1994 (Chapter 2) indicates that the level of catch has not changed. Both plots in Figure 4.4 have a relatively flat area to the left of the current fishing mortality. So, at a lower total fishing mortality, such as there was in 1979-1982, a similar total catch of pelagics would be predicted. The demersal feeders and engraulids have a similar response, although the latter does not show the double peak. The catch of the latter was also higher in 1979-1982 than in 1992-1994. In summary, these three pools are unlikely to be able to sustain further mortality if the effects of fishing mortality are examined on them alone. However, if the effects of fishing on other pools in the ecosystem are included in the analysis, they would be able to sustain further fishing mortality.

Their revitalisation results from the decrease in their predators, and therefore a reduction in predation mortality, caused by further fishing.

This is more dramatically shown by the crustacean pools (sergestid, penaeids and large crustaceans). The yield curves for the penaeids are shown in Figure 4.5. For the yield curve-*no fishing*, current fishing mortality is either below optimum fishing mortality (bottom-up) or

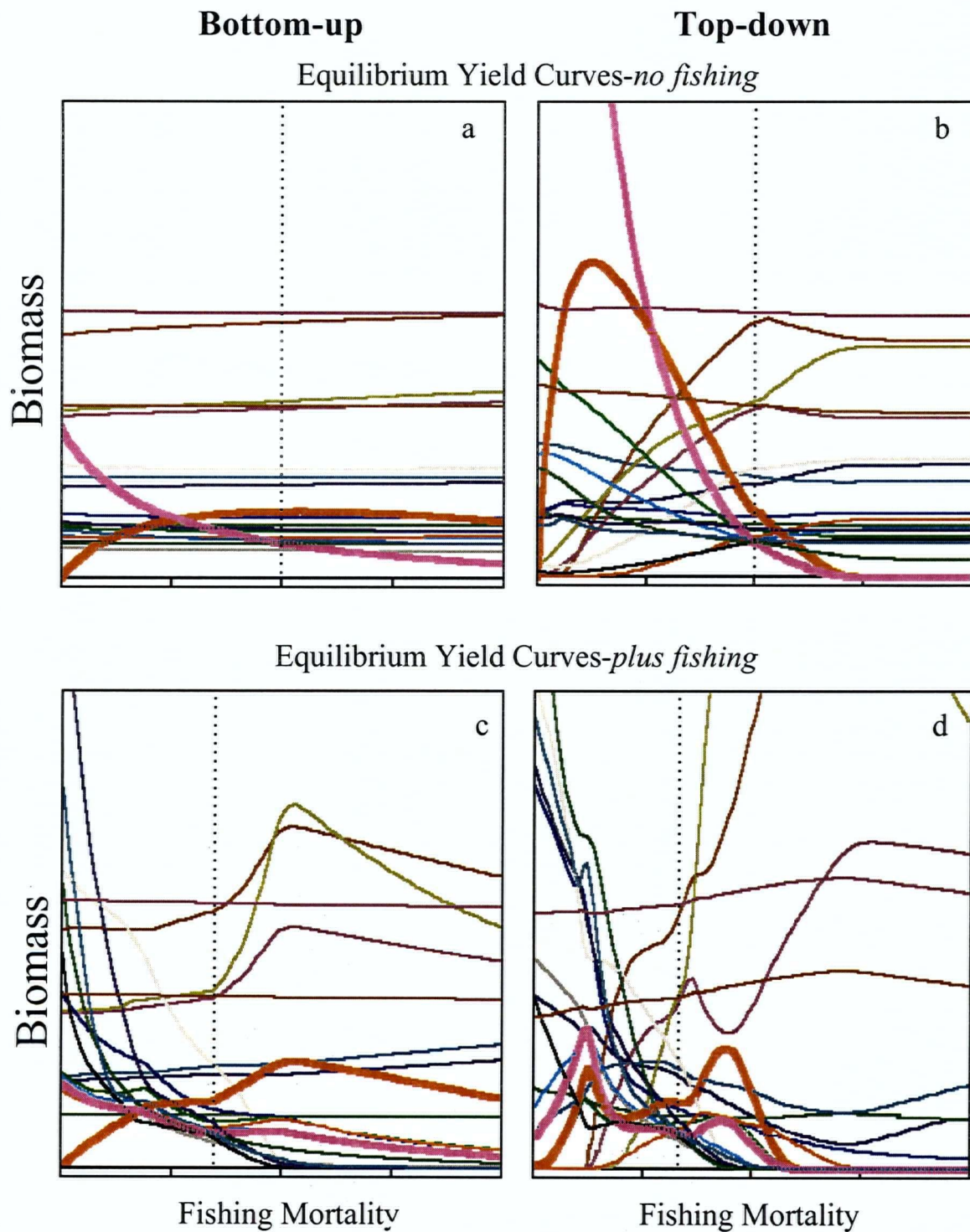


Figure 4.4. Equilibrium yield curves for the pelagics. In (a) and (b) fishing mortality is varied on only the pelagics. In (c) and (d) total fishing mortality is varied as in Figure 4.2. Thick red line indicates yield curve, coloured lines, biomass, dotted black line, current fishing mortality. See Table 4.1 for legend and scale. For further details see text.

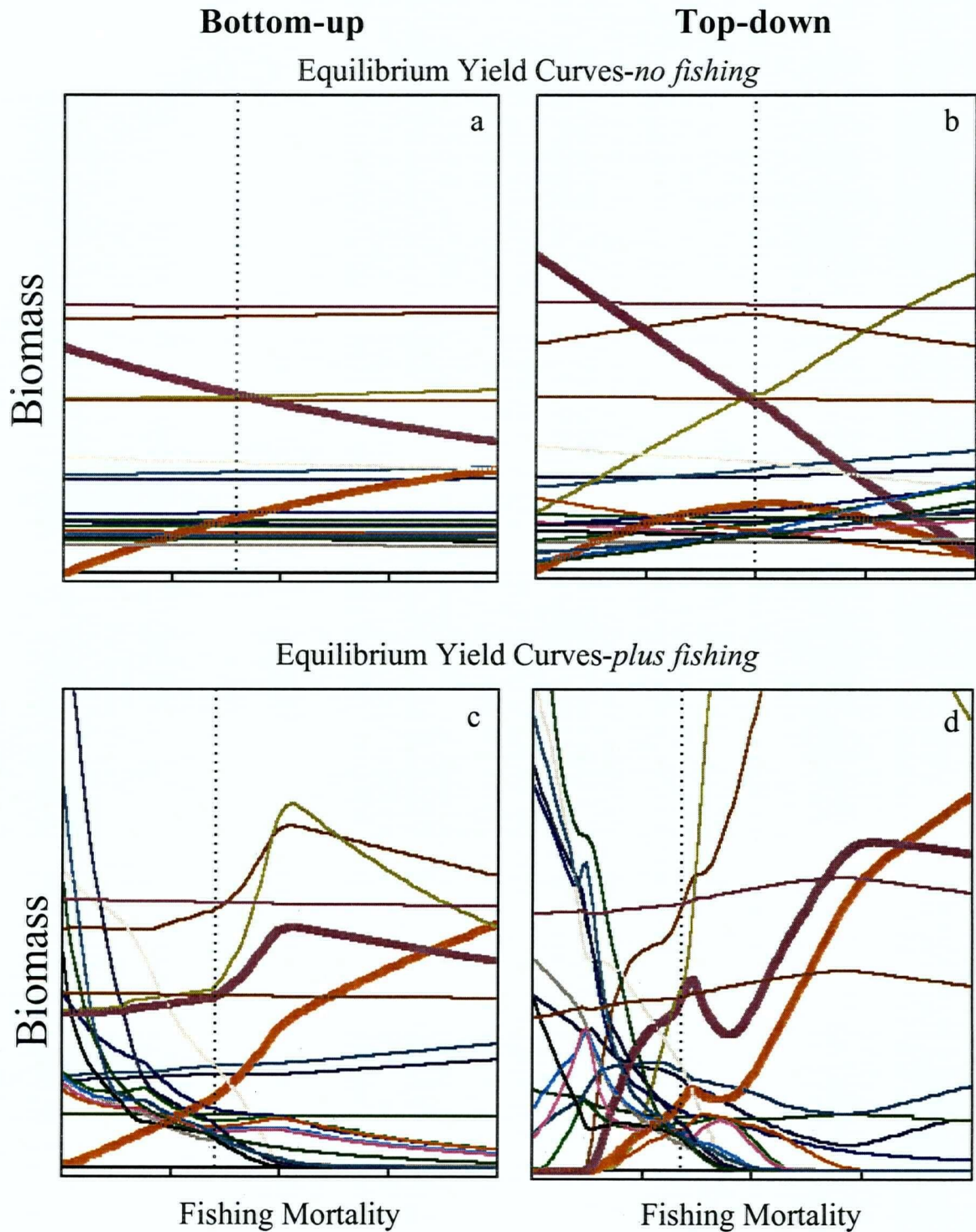


Figure 4.5. Equilibrium yield curves for the penaeids. In (a) and (b) fishing mortality is varied on only the penaeids. In (c) and (d) total fishing mortality is varied as in Figure 4.2. Thick red line indicates yield curve, coloured lines, biomass, dotted black line, current fishing mortality. See Table 4.1 for legend and scale. For further details see text.

at optimum fishing mortality (top-down). However, for the yield curves-*plus fishing*, current fishing mortality produces a yield far below the potential optimal yield produced when fishing mortality is increased and the other pools in the ecosystem are severely reduced by over-fishing. Again this is the case regardless of the flow assumption, although a much stronger response is seen for the top down assumption.

Some pools exhibit the opposite behaviour. For example, the adult medium predators yield curves-*no fishing* and the single species yield curve for *Trichiurus haumela* (62% medium predators trawl biomass) indicate that current fishing mortality is at about optimum fishing mortality. The yield curves-*plus fishing* indicate that current fishing mortality is substantially greater than the optimum fishing mortality (Figure 4.6). In the latter case, the optimum fishing mortality occurs at a much lower fishing mortality. This is because, when fishing mortality is reduced across the whole fishery, the biomass of many other groups also increases, including prey species. The medium predators are by definition predators and thus an increase in prey abundance plus a decrease in fishing pressure enables a substantial increase in their biomass. The optimum fishing mortality corresponds with high biomass and the return from fishing mortality. For the yield curve-*no fishing*, all predator groups are at low biomass levels: decreasing fishing mortality only allows the increase in biomass of the predator, not its prey. In fact, with the top-down assumption, the prey of the medium predators decrease further with the increase in biomass of the medium predators. Thus the medium predators are much more productive when fishing is reduced across the whole fishery and the abundance of their prey is increased. Notably it is the pools at the highest trophic levels in the ecosystem, that is the adult large predators, medium predators and the juvenile medium predators, that show

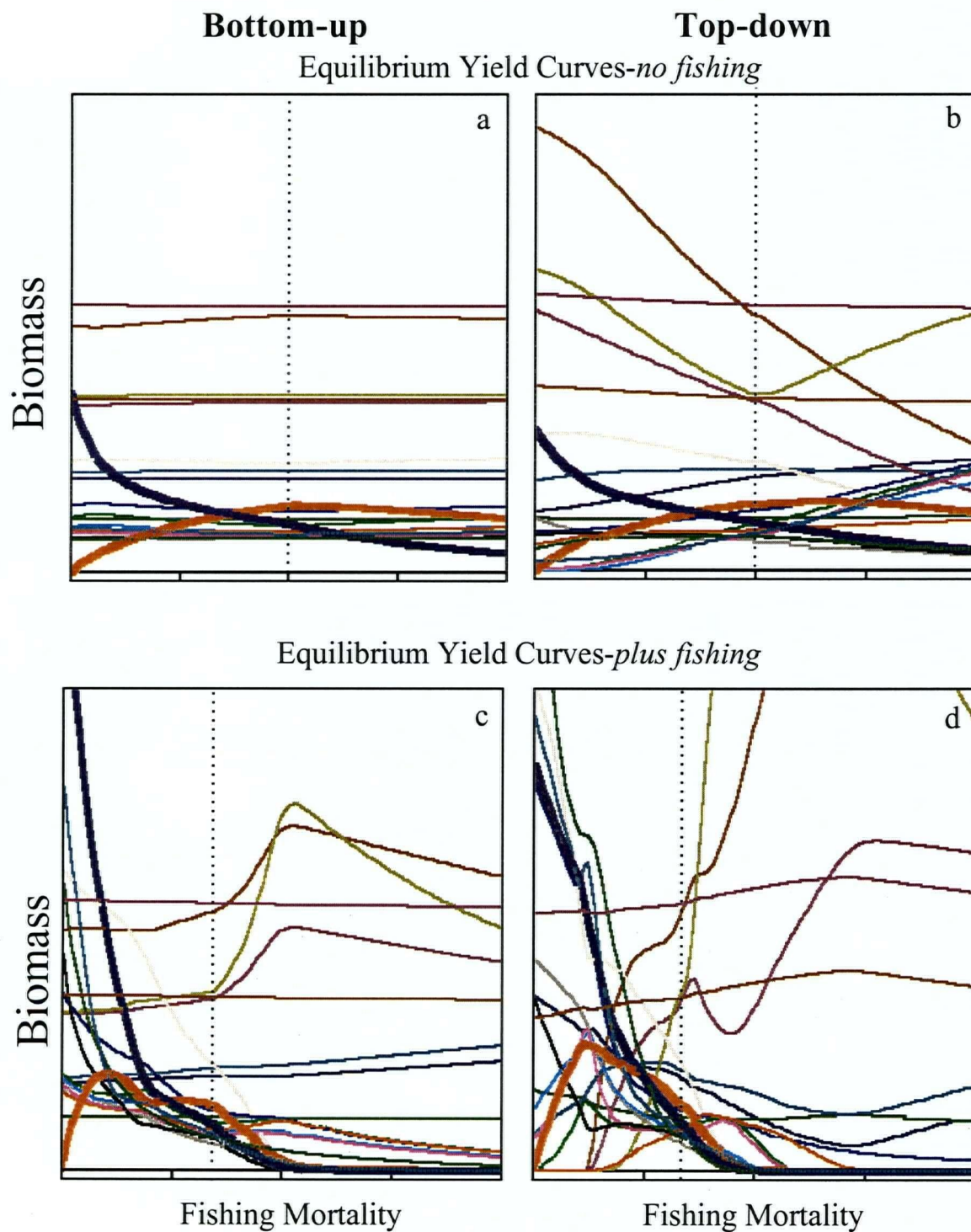


Figure 4.6. Equilibrium yield curves for the adult medium predators. In (a) and (b) fishing mortality is varied on only the medium predators. In (c) and (d) total fishing mortality is varied as in Figure 4.2. Thick red line indicates yield curve, coloured lines, biomass, dotted black line, current fishing mortality. See Table 4.1 for legend and scale. For further details see text.

this response. The yield curves of the juveniles medium predators are very similar to the adult medium predators suggesting that their abundance, at least in part, is a consequence of the adult abundance.

The leiognathids and juvenile large predators respond similarly with both types of multispecies yield curve, but differently with the assumptions made about flow dynamics. When a top-down assumption is made about flow dynamics, current fishing mortality is above the optimum fishing mortality, that is, the leiognathids are overfished. With bottom-up control, they are not overfished: current fishing mortality is below optimum fishing mortality for the yield curve-*plus fishing* and at about fishing mortality for the yield curve-*no fishing*. The single species yield curves for *L. splendens* and *L. bindus* show these two species, which comprise 55% of the trawl biomass of the leiognathids, to be overfished. However, the yield curve for another leiognathid, *Secutor ruconius* (32% trawl biomass) indicates that current fishing mortality is at the optimal rate.

Finally, some pools give the same results regardless of type of yield curve or the assumptions made about flow dynamics. The juvenile sciaenids and the large zoobenthos feeders are always overexploited and the adult sciaenids are always just at optimum fishing mortality. The single species yield curve for *Otolithes ruber* (65% trawl biomass of sciaenids) indicates that this species is highly overfished. Since most of the catch consists of juveniles (see chapter 2) this result is in accordance with the multispecies results.

The results from the two types of multispecies yield curve analyses underline the critical importance of the assumptions made about flow dynamics. Generally, the bottom-up assumption results in a current fishing mortality that is much closer to the optimum fishing mortality than the top-down assumption. Thus the predictions from a bottom-up analysis indicate, for many pools, a fishery in a reasonably healthy state. However, the top-down assumption results often indicate quite the opposite, where pools are seriously overfished. Clearly in this case, sustainable management would require measures that would act to conserve these pools and reduce fishing effort. Since it is not known what type of flow dynamics prevail in San Miguel Bay, making the top-down assumption would be the more cautious approach to managing the fishery. This would be consistent with the precautionary principle (FAO 1995a,b). The results from the bottom-up analysis could lead to the conclusion that there was potential for expansion in some sectors of the fishery.

The analyses also demonstrate the contrasting results obtained from examining the yield curves when fishing mortality is held constant on all pools other than that pool being examined (yield curve-*no fishing*) and the yield curves when fishing mortality is varied across all gears in the fishery. Fishing other pools in the fishery affects the shape of the yield curve. This can mean that the pool will sustain more fishing than the yield curve-*no fishing* indicates, it may imply that less fishing can be sustained, or the two analyses may reach the same conclusions. In a multigear, multispecies fishery then, it is vital to examine yield curves produced by changing fishing patterns across the whole fishery, or selected fishing gears, rather than examining the yield curve produced by changing fishing mortality on one pool alone. In this analysis, fishing mortality was varied across all gears simultaneously: in reality, increases in

fishing mortality would occur differentially across gears. This was demonstrated in Chapter 2 where the changes in gear composition across the fishery and their implications were described. The consequences of differential changes in fishing effort and fishing gear in San Miguel Bay are explored below.

The overall picture is that the crustacean pools would withstand further exploitation and indeed lead to a great increase in productivity, if the biomass of the other pools in the ecosystem were reduced further by increased fishing mortality. The pools at the top of the ecosystem are overfished no matter which way you look at it, with the exception of the medium predators which may not quite yet be overfished, although they are maximally fished. The state of the pools at an intermediate trophic level, that is the demersal feeders, leiognathids, engraulids, and pelagics is less clear. These groups respond to both changes in their predators and prey and so their status, like that of the crustacean groups, depends more heavily on what is happening in the rest of the fishery. Certainly the suggestion from the yield curve-*plus fishing* analysis is that they could sustain a higher fishing mortality. However here caution should be exercised since the yield curve-*no fishing* so clearly contradicts this.

The results of the multispecies yield curves broadly agree with the single species yield curves, (see Table 4.2). They give the similar relationships between current fishing mortality and optimum fishing mortality. They also produce similar parabolic shaped curves, although the shape of some are distorted by the effects of other pools. However, Figures 4.4-4.6 clearly show that changing the biomass of one pool in the ecosystem can strongly impact on other pools in the ecosystem. The single species analysis can give no indication of how changes in

one pool might affect other pools. This was briefly discussed above and will now be examined in more detail.

Multispecies Dynamics in the San Miguel Bay Ecosystem

The equilibrium yield curves-*no fishing* illustrate the changes in equilibrium biomass of all pools when the biomass of one pool is changed (by incrementally increasing and decreasing fishing mortality on that pool). When bottom up control of flow dynamics is assumed, most pools have little effect on the biomass of the other pools in the ecosystem. Since the other pools are not subject to changes in fishing mortality and are limited only by food availability, their biomass would not be expected to change very much, unless their food availability changes. However, when top down control is assumed, because of strong predator influences, there are often quite dramatic responses to the changes in biomass of just one pool. Table 4.3 maps these responses for top-down control.

Surprisingly, the large predators, at the top of the ecosystem have very little impact on the rest of the ecosystem, regardless of flow assumption. The juveniles have more effect than the adults because they form the prey of other pools. However, no pools consume the adult large predators, and because of their low biomass, they do not eat much of anything else.

In contrast, changes in biomass of the sciaenids and medium predators have profound effects on the other pools in the ecosystem. Figure 4.6 a,b illustrates this for the adult medium

Table 4.3. Impact of decreasing the biomass of one pool on the other pools in the San Miguel Bay ecosystem when top-down control is assumed. . The biomass of the pools in the left hand column are decreased. The pools across the table respond to the decrease.

Biomass Response

| <i>Biomass Decrease</i> | Serg | Pen | LC | J_Sci | J_MP | J-LP | DF | Leiog | Eng | Pel | Sci | MP | LZB | LP |
|-------------------------|------|-----|------|-------|------|------|-----|-------|-----|-----|-----|----|-----|----|
| Serg | | - | -+ | -- | ~ | + | - | + | + | ++ | - | ~ | + | + |
| Pen | +- | | ++ | - | ~ | ++ | + | ++ | -- | ~ | - | + | ++ | ++ |
| LC | ++ | ++ | | - | ~ | - | ~ | - | + | - | ~ | -- | -- | -+ |
| J_Sci | ++ | +- | ++ | | - | -+ | +~+ | +--+ | + | +~+ | -- | - | -- | -+ |
| J_MP | -- | -- | --++ | +- | | ++ | ++ | ++ | + | ++ | +- | - | +- | + |
| J-LP | - | - | -- | + | + | | - | + | - | + | + | + | + | - |
| DF | ++ | ++ | +-- | ++ | - | -- | | +--+ | ++ | -- | ++ | -- | ++ | -- |
| Leiog | ++ | -+- | +-- | ++ | - | -- | ++ | | ++ | -- | ++ | - | ++ | -- |
| Eng | ++ | ++ | -- | ++ | -- | -- | ++ | +-- | | +-- | ++ | -- | -- | -- |
| Pel | ++ | ++ | ++ | ++ | ~ | - | -- | -- | ++ | | ++ | - | -- | - |
| Sci | + | +- | ++ | -- | - | -+ | -+ | -- | ++ | ++ | | - | -- | ~+ |
| MP | -- | -- | --++ | -- | - | ++ | ++ | ++ | + | ++ | - | | ~ | ++ |
| LZB | -- | ++ | -- | ++ | + | -- | ~ | - | + | + | + | ++ | | - |
| LP | ~ | ~ | ~ | ~ | ~ | -- | ~ | ~ | ~ | ~ | ~ | ~ | ~ | |

+ = positive response, ++ = strong positive response, - = negative response, -- = strong negative response, ~ = no change.

Serg = Sergestids, Pen = Penaeids, LC = Large Crustaceans, J_Sci - Juvenile Sciaenids, J_MP = Juvenile Medium Predators, J_LP = Juvenile Large Predators, DF = Demersal Feeders, Leiog = Leiognathids, Eng = Engraulids, Pel = Pelagics, Sci = Sciaenids, MP = Medium Predators, LZB = Large Zoobenthos Feeders, LP = Large Predators.

predators. When medium predator biomass is decreased from the left to the right of the plot (and F is increased), the biomass of the leiognathids, the demersal feeders, the pelagics and the large predators all increase substantially. The biomass of the engraulids also increases but more slightly. These pools are all prey for the medium predators. The sciaenids decrease with the medium predators, as do the sergestids and the penaeids. The large crustaceans decrease then increase. Essentially then, increasing the biomass of the medium predators (either adults or juveniles) leads to a decrease in the biomass of most of the other fish pools in the ecosystem. Under the bottom-up assumption there is little change in any group other than the adult medium predators. However, when the biomass of the juvenile medium predators is decreased, there is a small increase in the biomass of the leiognathids, the demersal feeders, the engraulids, the pelagics, the sciaenids and the large predators. These increases might be due to the release from competitive pressure for food by the juvenile and adult medium predators.

The sciaenids also exhibit a repressive effect on other pools in the ecosystem when top down control is assumed. In this case, the engraulids, the pelagics, the large predators, the sergestids and the large crustaceans are most affected and increase when the biomass of sciaenids is decreased (Figure 4.7 d). The leiognathids, the medium predators and the large zoobenthos feeders decrease, while the demersal feeders increase slightly when the biomass is increased or decreased. This repressive effect is much weaker when bottom-up control is simulated however (Figure 4.7c). In this case the three crustacean groups all increase when the biomass of the sciaenids is reduced. Most of the fish pools also increase, but only very slightly.

The plots for the juvenile sciaenids are less easy to interpret (Figure 4.7 a,b). Both bottom-up and top-down interpretations present problems. Under the top-down assumption, the medium predators decrease with decreasing biomass of sciaenids and the basic trend for the leiognathids and the large zoobenthos feeders is also to decrease. This result is qualitatively the same as for the adult sciaenids. The engraulids, sergestids and large crustaceans increase. However, the demersal feeders, the pelagics, the large predators and the penaeids have a more complicated response. The trend in the biomass of these pools changes at the point of current fishing mortality, and it changes again at a much lower value of fishing mortality on the juvenile sciaenids.

When the juvenile sciaenids are subject to the bottom-up assumption, the model crashes as it incrementally tracks its way down the fishing mortality range from current fishing mortality (Figure 4.7a). As fishing mortality is decreased, the biomass of the juvenile sciaenids and the adult sciaenids increase. However, before the fishing mortality reaches zero, the biomass of the juveniles suddenly and rapidly crashes while the biomass of the adult increases exponentially. At this point the model fails. The failure is caused by a combination of factors. One is that the numerical procedure for tracking the movement of the system equilibrium breaks down when a parameter (including fishing rate) combination is encountered for which the system has either no stable state or a bifurcation to several possible dynamic patterns

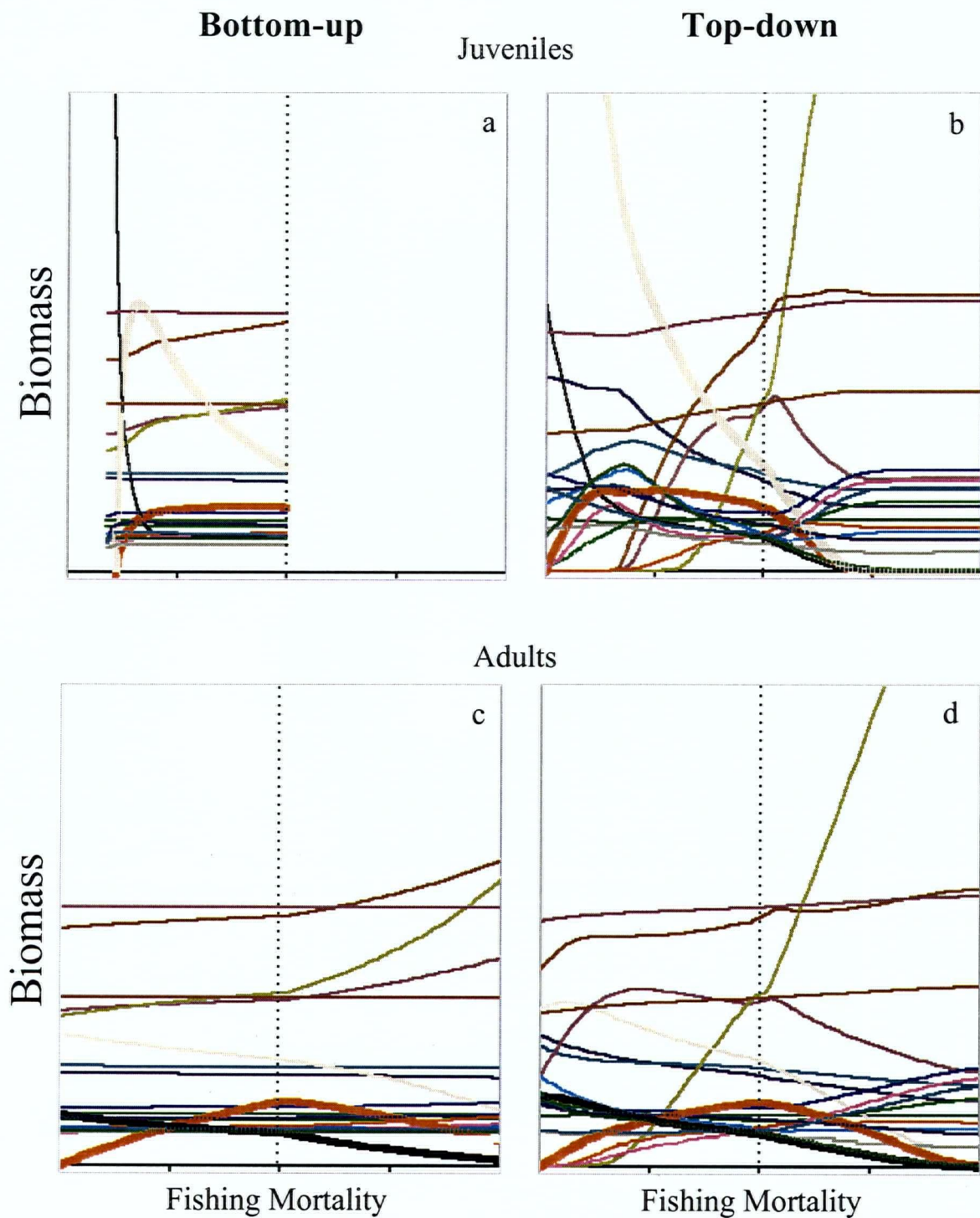


Figure 4.7: Equilibrium yield curves-*no fishing* showing ecosystem impacts of the sciaenids. Thick red line indicates yield curve, coloured lines, biomass, dotted black line, current fishing mortality. See Table 4.1 for legend and scale. For further details see text.

(C. Walters, *pers. comm.*). However, in addition to this there is a more serious and fundamental problem.

This is a problem with the way that the two-pool delay differential model is structured.

ECOSIM explicitly models the flow from adults to juveniles and juveniles to adults. The flow of adult sciaenids to juveniles has reasonable values in the above simulation (high adult biomasses produce high flows into the juvenile pool as new recruits). However, the flow of surviving juveniles moving to the adult pool at presumed size W_k , takes values that are much too large. The equation which calculates this flow is based on a steady-state approximation of juvenile flow to adults. It is this approximation that fails, whereby juveniles would not be able to reach size W_k in the time predicted by the model, or would reach that size very much sooner than predicted. Such failure is caused by extreme changes (or very rapid changes) in the biomasses (C. Walters, *pers. comm.*)⁷⁷.

The model is thus unable to predict, when bottom-up flow dynamics are assumed, what would happen when the biomass of the sciaenids, juveniles and adults, is increased through a decrease in fishing mortality. The increased abundance would presumably lead to increased competition for food and space. One hypothesis (C. Walters, *pers. comm.*) is that increased competition would lead to an increase in foraging time which would lead to increased vulnerability. Since bottom-up control is modelled as low vulnerability, it would not make any sense to assume bottom-up control when the juveniles are so abundant that they have trouble getting enough food to reach body size W_k within a time span set in the input data. This

⁷⁷ A new version of the model, ECOSIM II is now available to correct these problems for future studies; it was not available in time for this analysis.

approach was adopted here and in the bottom-up simulations, the juvenile sciaenids are given an intermediate vulnerability factor (see footnote 74)⁷⁸.

Biomass changes of the pools at the middle of the trophic range, that is the demersal feeders, leiognathids, engraulids and the pelagics (Figure 4.4) also impact strongly on other pools in the ecosystem under top-down control. If their biomass is decreased, the biomass of the sergestids and the sciaenids increases while the medium predators and the large predators decrease. The former are largely competitors for food, the latter, predators on these four pools. The large zoobenthos feeders increase when the demersal feeders and the leiognathids are decreased but decrease when the engraulids and pelagics are decreased. Within this groups of four pools, the pelagics always decrease when the biomass of the other three pools is decreased and the engraulids increase. The demersal feeders and the leiognathids both decrease when the pelagics decrease. Thus these groups also have strong influences on the rest of the ecosystem.

The crustacean groups have less of a direct impact on the higher trophic pools in the ecosystem (Figure 4.8), even under the bottom-up assumption (one might think that large increases or decreases in the biomass of these lower trophic groups would have impacts for those groups higher in the system if their biomass is controlled from the bottom up). Under the bottom-up assumption, the three pools have almost identical effects on the other pools in the ecosystem (Figures 4.5 and 4.8). These crustacean pools are more indicators of change than determinants of change. However, under the top-down assumption, these pools have some impact on the

⁷⁸ However, there are other possible explanations for this response. For example, a density-dependent compensatory growth response may occur. An experimental approach, reducing fishing mortality in order to create this condition would begin to provide some insight into what occurs on reality (see Chapter 5).

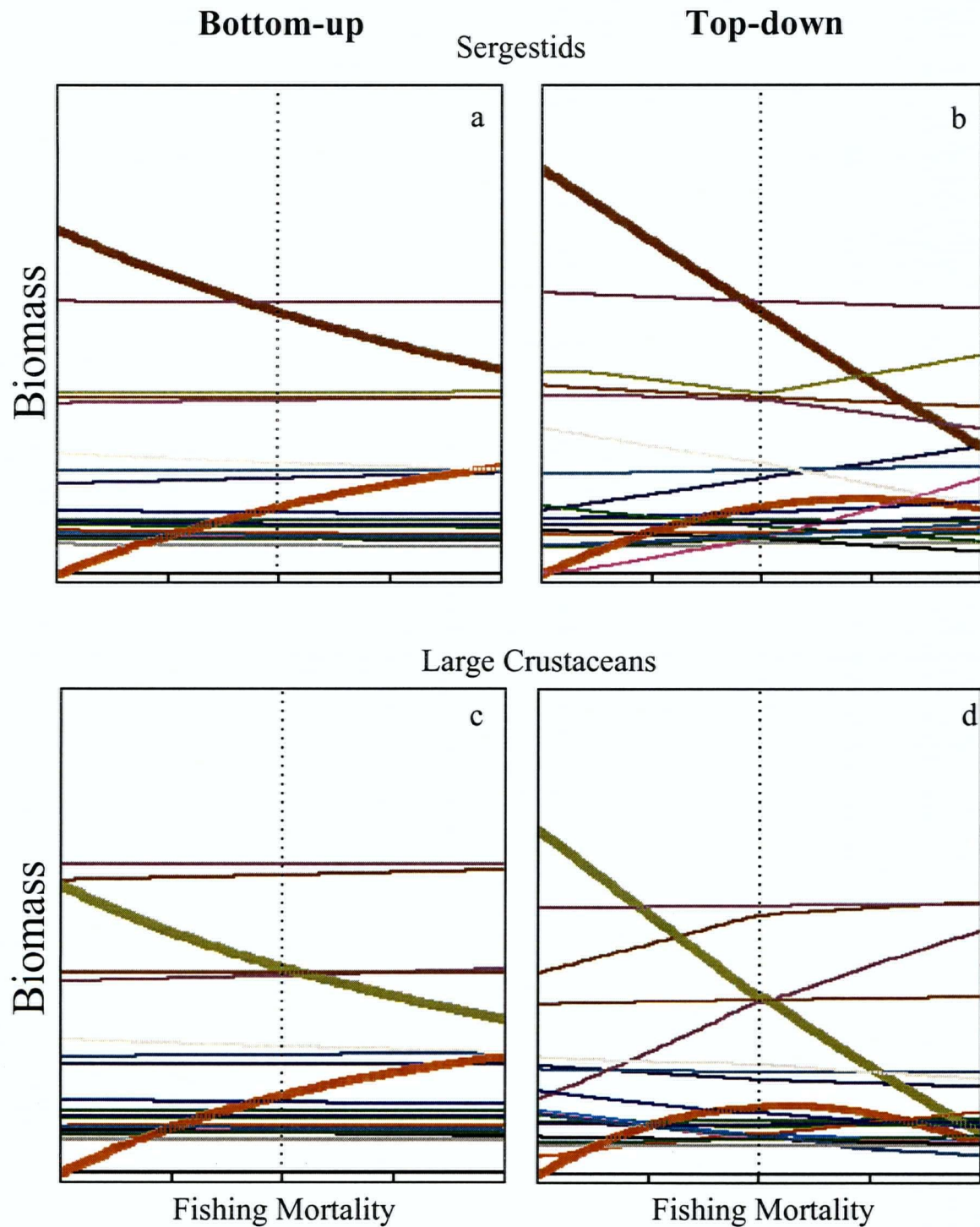


Figure 4.8. Equilibrium yield curves-*no fishing* showing ecosystem impacts of the sergestids (a) and (b) and the large crustaceans, (c) and (d). Thick red line indicates yield curve, coloured lines, biomass, dotted black line, current fishing mortality. See Table 4.1 for legend and scale. For further details see text.

middle pools in the ecosystem, such as the leiognathids, demersal feeders, engraulids and pelagics and on the juvenile pools.

Table 4.3 maps the responses of all the pools to one another, under the top-down assumption. There are some patterns to be noted. The medium predators usually decrease when the biomass of another pool decreases. The sciaenids and the medium predators are positively linked in their response to one another, that is when the biomass of one is changed by changing fishing mortality, the biomass of the other also changes in the same direction (see Figures 4.6 and 4.7). They also both exert a strong repressive effect on the other pools in the ecosystem. However, while the sciaenids increase when the biomass of the demersal feeders, leiognathids, engraulids and pelagics increases, the medium predators decrease. The penaeids and the large crustaceans are competitively linked: an increase in the biomass of one produces a decrease in the biomass of the other. This is shown quite clearly in Figure 4.5, where the biomass of the penaeids decreases across the plot while the biomass of the large crustaceans increases (top-down control). Essentially, Table 4.3 portrays a complexity of response across the San Miguel Bay food web. It is difficult to break this web of response down into smaller, more easily understood components. There are many interactions between species and these may not be simple hierarchical interactions. This conclusion agrees with the findings of Walters *et al.* (1997). They point out that with complex food webs such as those found in the tropics, it can be difficult to make simple and clear predictions about ecosystem response to changing fishing patterns.

These results demonstrate that it is not possible to cause change in one part of the ecosystem without effecting the rest of the interconnected pools in that system. This is the situation whether bottom-up or top-down control is assumed, although the impacts under the latter are much greater. However, in fisheries science this is largely what is done. That is, the effects of fishing on one pool or species in the ecosystem are directly analysed and the rest are ignored. The results described here underline the need to change the way that science is done. However, these results also show that it is no easy matter to model the ecosystem or to predict outcomes. Responses to perturbations are neither simple nor straightforward.

In fisheries management it is not possible to manipulate just one pool or species in a multispecies fishery directly because fishing gears are usually not that selective. The above analysis of the effect of single pools in the ecosystem model allowed patterns of response to be discerned, where possible, and the complexity and interconnected nature of the ecosystem to be fully realised. In practice though, it is fishing effort and fishing pattern that is manipulated by management strategies.

The Effects of Fishing in San Miguel Bay

All results indicate that the current rate of fishing mortality and the fishing pattern is excessive in San Miguel Bay and that it has led to growth, recruitment and ecosystem overfishing. Figure 4.2 illustrated the effect on the equilibrium biomass of all pools of increasing or decreasing fishing mortality. These results indicated that the conservative, and perhaps only wise

approach to management in San Miguel Bay, would be to reduce fishing effort over the whole fishery, although under certain conditions there might be room for expansion of some sectors. The analysis of the pool dynamics in San Miguel Bay demonstrated that changes in the biomass of individual pools can have considerable effect on the species composition of the ecosystem. In this section the effects of fishing in conjunction with the ecosystem dynamics are examined.

Of the nineteen gears represented in the model, four account for over 54% of the total 1992-1994 catch (see Table 2.15). The effect that each of these gears has on the ecosystem was analysed by running a time simulation (the "Dynamic Run" of ECOSIM described above) over 10 years. Over the first 2 years, effort was linearly reduced to zero and then kept at that level for the next 8 years. The results are shown in Figure 4.9 for top-down control and for bottom-up control in Figure 4.10 (pools impacted by the gear). The overall biomass changes are given in Table 4.4 below (the effective change in total biomass in the ecosystem).

Once more, what is most striking about the biomass plots is that there is a great deal of change in the pool composition, particularly when strong top-down assumptions are made. The changes are both unidirectional and multidirectional. Of the four gears, the changes of greatest magnitude for individual pools are caused by the ordinary gillnet and the hunting gillnet, which landed 15% and 10% of the total 1992-1994 catch respectively. It is no coincidence that these gears target the sciaenids and the medium predators, noted above for their repressive influence on the rest of the ecosystem. The biomass of these pools increases substantially, by over 100% in the case of the sciaenids and 90-100% for the medium

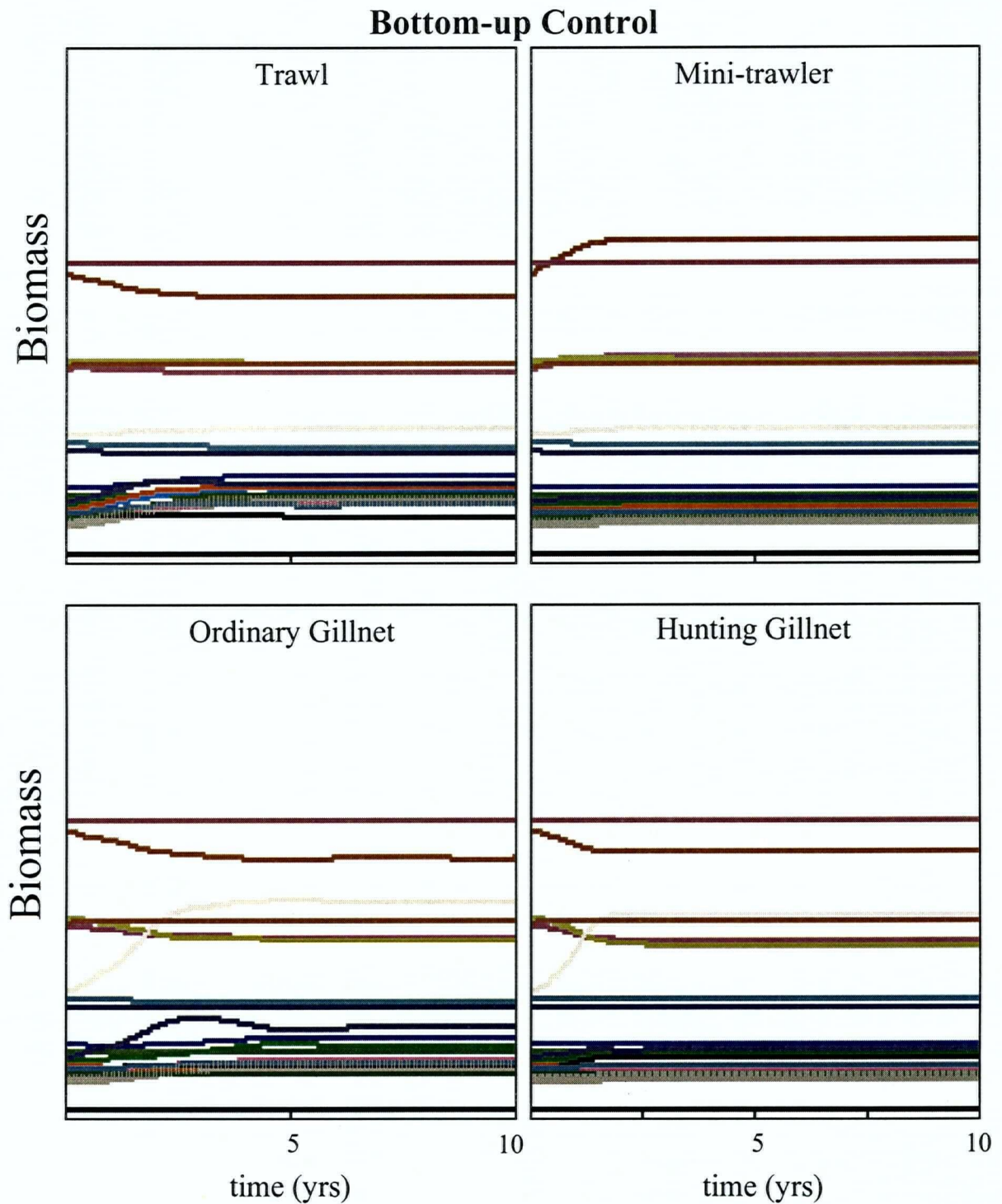


Figure 4.10. Dynamic simulations reducing the fishing effort of each gear to zero over the first 2 years of a 10 year simulation under the bottom-up control assumption. The coloured lines are biomass, the thick line the juvenile medium predators. See text for further details, Figure 4.1 for fishing pattern and Table 4.1 for legend and scale.

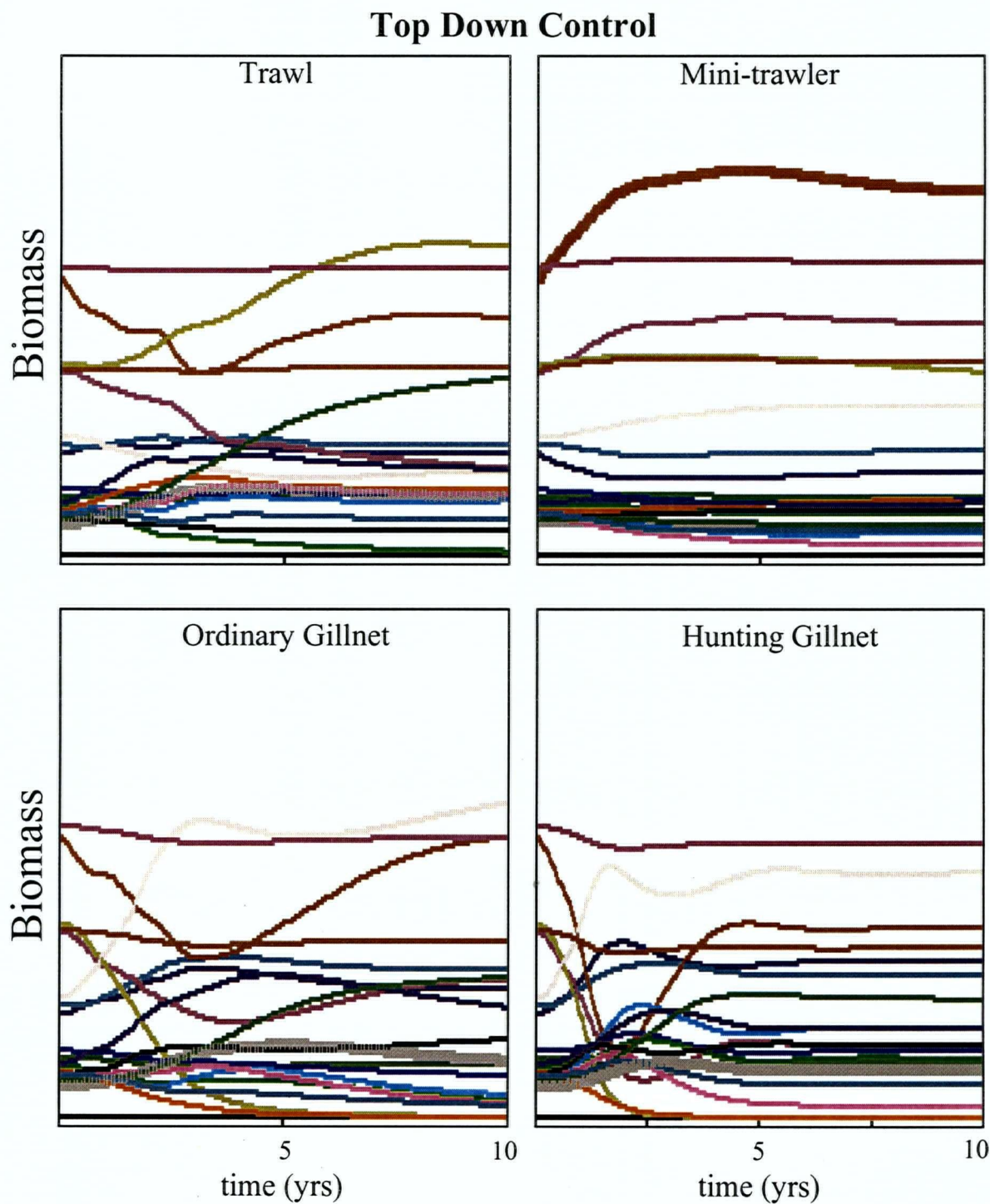


Figure 4.9. Dynamic simulations reducing the fishing effort of each gear to zero over the first 2 years of a 10 year simulation under the top-down control assumption. The coloured lines are biomass, the thick line the juvenile medium predators. See text for further details, Figure 4.1 for fishing pattern and Table 4.1 for legend and scale.

Table 4.4 Percentage change in total fished biomass, for each type of fishing gear after 10 years and a reduction in fishing effort to zero.

| | <i>Trawl</i> | <i>Mini- trawler</i> | <i>Ordinary Gillnet</i> | <i>Hunting Gillnet</i> | <i>Shrimp Gillnet</i> | <i>Surface Gillnet</i> | <i>Bottom Gillnet</i> | <i>Shark Gillnet</i> | <i>Crab Gillnet</i> | <i>Other Gillnet</i> |
|----------------------------------|--------------|--------------------------|-----------------------------|----------------------------|---------------------------|----------------------------|---------------------------|--------------------------|-------------------------|--------------------------|
| | <i>t</i> | | | | | | | | | |
| Top Down | 21.7 | 4.4 | 6.0 | -3.4 | -1.8 | 3.3 | 4.1 | 0.6 | 9.8 | 5.5 |
| Bottom Up | 14.8 | 4.0 | 15.8 | 7.6 | 3.1 | 1.1 | 1.1 | 0.1 | 6.2 | 6.2 |
| % of total catch landed | 17 | 12 | 15 | 10 | 6 | 1 | 2 | <1 | 4 | 8 |

Table 4.4 (cont.) Percentage change in total fished biomass, for each type of fishing gear after 10 years and a reduction in fishing effort to zero.

| | <i>Liftnet</i> | <i>Crab Liftnet</i> | <i>Longline</i> | <i>Hand line</i> | <i>Filter Net</i> | <i>Fish Corral</i> | <i>Scissor Net</i> | <i>Fish Trap</i> | <i>Other</i> |
|----------------------------------|----------------|-------------------------|-----------------|----------------------|-----------------------|------------------------|------------------------|----------------------|--------------|
| Top Down | 28.4 | 0.3 | 15.3 | 0.9 | 6.8 | 0.1 | 2.8 | 0.6 | 0.2 |
| Bottom Up | 6.0 | 0.2 | 6.6 | 0.2 | 1.9 | 1.0 | 1.1 | 0.1 | 0.3 |
| % of total catch landed | 6 | <1 | 3 | <1 | 5 | 2 | 7 | <1 | 1 |

predators (Figure 4.9). The large zoobenthos feeders increase by almost three fold. However, other pools decline. In the case of the hunting gillnet, the large crustaceans and engraulids decrease by, essentially, 100%, the penaeids, and pelagics decrease by over 60% and the sergestids decrease by 33% over the 10 year period (all biomass levels flatten out after this). When the ordinary gillnet effort is decreased to zero, all the above pools similarly decrease (except the sergestids, which initially decrease then regain their biomass) plus the demersal feeders, leiognathids and the large predators decrease by between 36 and 73%. These changes occur when top-down control is assumed.

By contrast the trawlers account for 17% of the total catch and the Mini trawlers 12%, yet their impact on the rest of the ecosystem does not appear to be as great as either of the above gillnets. For the trawlers, the large zoobenthos feeders again increase greatly and the large crustaceans increase by 62%, the engraulids and pelagics by 50% and the leiognathids by 27%. The demersal feeders decrease by 93%, the penaeids by 52% and the sciaenids by 30% of their start biomass. The magnitudes of biomass change for the mini-trawler are considerably less. The sergestids, penaeids, sciaenids, juvenile medium predators and engraulids all increase by between 3 and 32%, while the large crustaceans, large predators, demersal feeders, leiognathids, pelagics, adult medium predators and large zoobenthos feeders decrease (4 - 71%). The mini-trawler has the greatest negative impact on the pelagics, leiognathids and large predators and the greatest positive impact on the sergestids.

The bottom-up simulations (Figure 4.10) produce a much flatter biomass response to reduced effort. The sciaenids still increase when the ordinary gillnet and the hunting gillnet are

reduced to zero, but the response of the other pools differs. All biomass pools other than the three crustacean pools and the engraulids (hunting gillnet only) increase. The results for the trawl simulation are similar. Reducing the trawl effort to zero results in biomass increases across all the fish pools, with only slight ($<10\%$) decreases in the three crustacean pools and the adult sciaenids (-0.04). Here the juveniles sciaenids and the demersal feeders increase where they decreased in the top-down simulation. For the mini-trawler, all pools increase except for a small decrease (-0.24) in the pelagics.

These percentage differences between start and end biomass for each of the exploited pools over the 10 year period are shown in Figure 4.11 This figure shows, for each of the gears, the direction and magnitude of the biomass response of each pool. The figure directly contrasts the results for the top-down and bottom-up assumptions. It shows quite clearly that under the bottom-up assumption, the magnitude of the biomass response to reduced fishing is generally considerably less than under top-down control and that it is basically positive for all pools other than the crustacean groups.

It also confirms that, under top-down control, much of the change in the biomass of individual pools is effected by the two gillnet gears. Under bottom-up control, the trawl and the ordinary gillnet have the greatest effect on individual pools. Here the hunting gillnet has less impact than with top-down control. In both cases the mini-trawler has the least effect of the four gears.

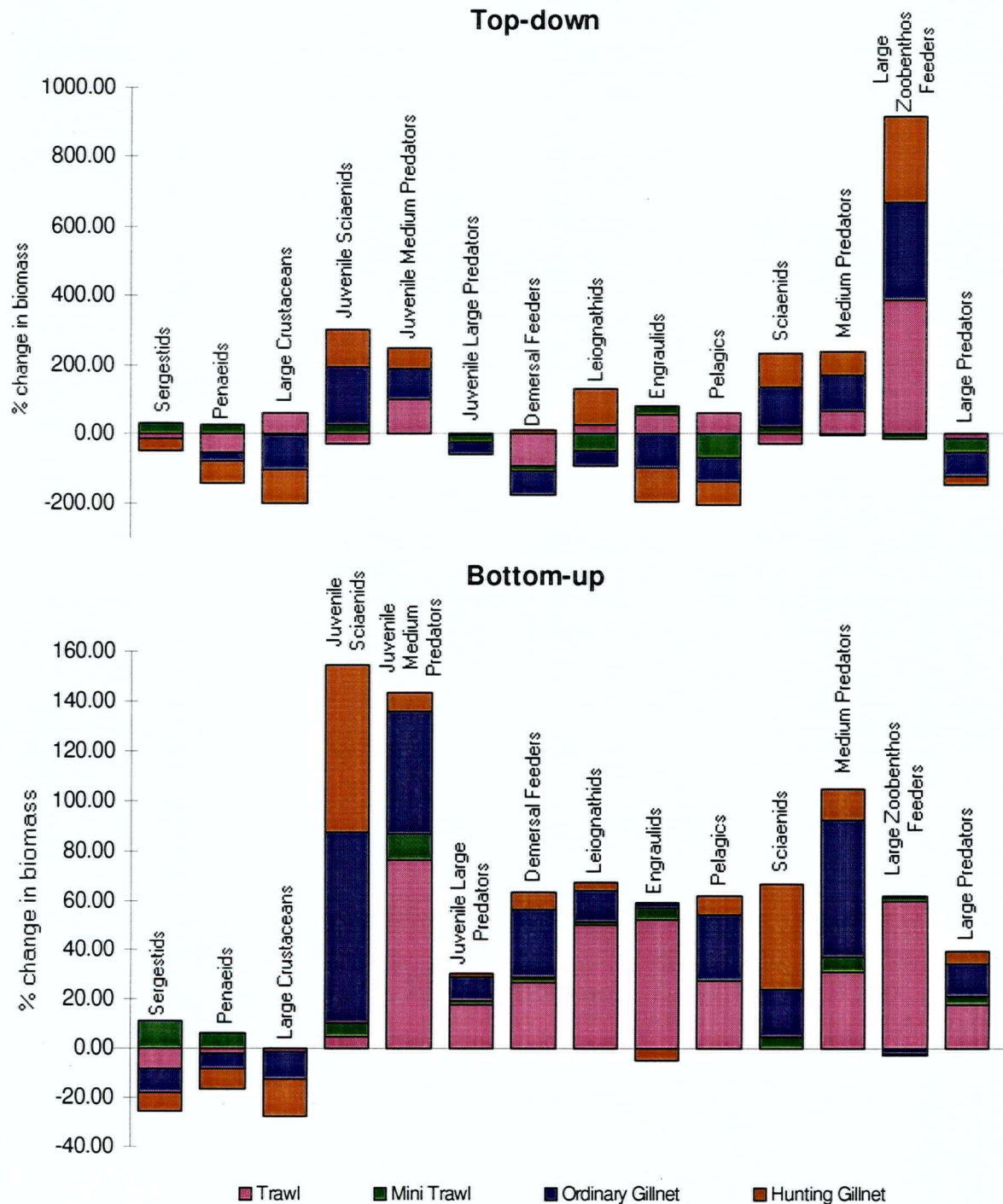


Figure 4.11 Percentage change in the biomass of each pool after 10 years for (a) top-down control and (b) bottom-up Control. The effort of each gear is reduced to zero as described in the text and in Figure 4.1. Gears shown are the trawl, mini trawl, ordinary gillnet and hunting gillnet.

When the effect on total biomass is examined (Table 4.4), a different picture emerges. Under top-down control, the trawl reduction produces a total biomass increase which is much greater than that produced by the ordinary gillnet, hunting gillnet and mini-trawler combined. Additionally, for the hunting gillnet, there is a reduction in total fished biomass. The results for bottom-up control reflect the results on the individual pools described above. The ordinary gillnet and the trawl produce the greatest increases in total biomass, followed by the hunting gillnet and then the mini-trawler.

A similar analysis was conducted for the other gears in San Miguel Bay. These gears are all small-scale gears and are usually more selective than the four gears just discussed. Most do not produce the magnitude of biomass changes seen above. Figures 4.12 and 4.13 show the % differences in biomass of each exploited pool for top-down and bottom-up control. Again the differential effects of the top-down and bottom-up assumptions are apparent: bigger biomass changes are generally seen with the top-down assumption. The gears which have greatest impact on the ecosystem, by the two measures described above are the liftnet, set longline, other gillnets, filter net, crab gillnet and shrimp gillnet.

The liftnet and the set longline have the greatest impact on individual pools. As pointed out in Chapter 2 however, the catch composition for the liftnet was taken from only one sample. Thus the large increase seen in the biomass of leiognathids and pelagics should be interpreted with caution. The results from the set longline gear are more compelling. The set longline targets the large predators and the large zoobenthos feeders and these pools benefit greatly from a reduction in fishing effort. Indeed, all pools are impacted, in some way, by the set

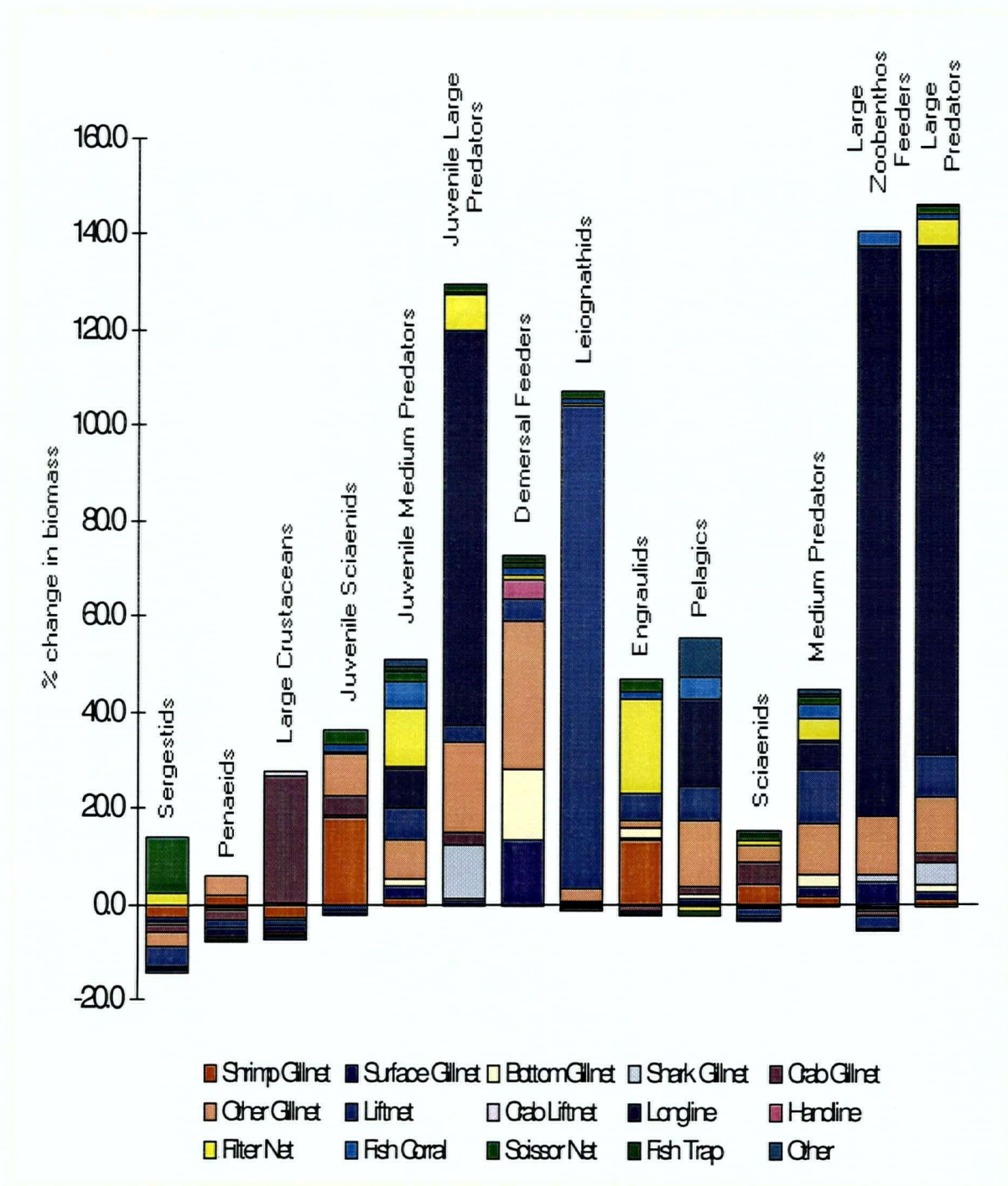


Figure 4.13 Percentage change in the biomass of each pool after 10 years when bottom-up control is assumed for all other gears. The effort of each gear is reduced to zero as described in the text and in Figure 4.1.

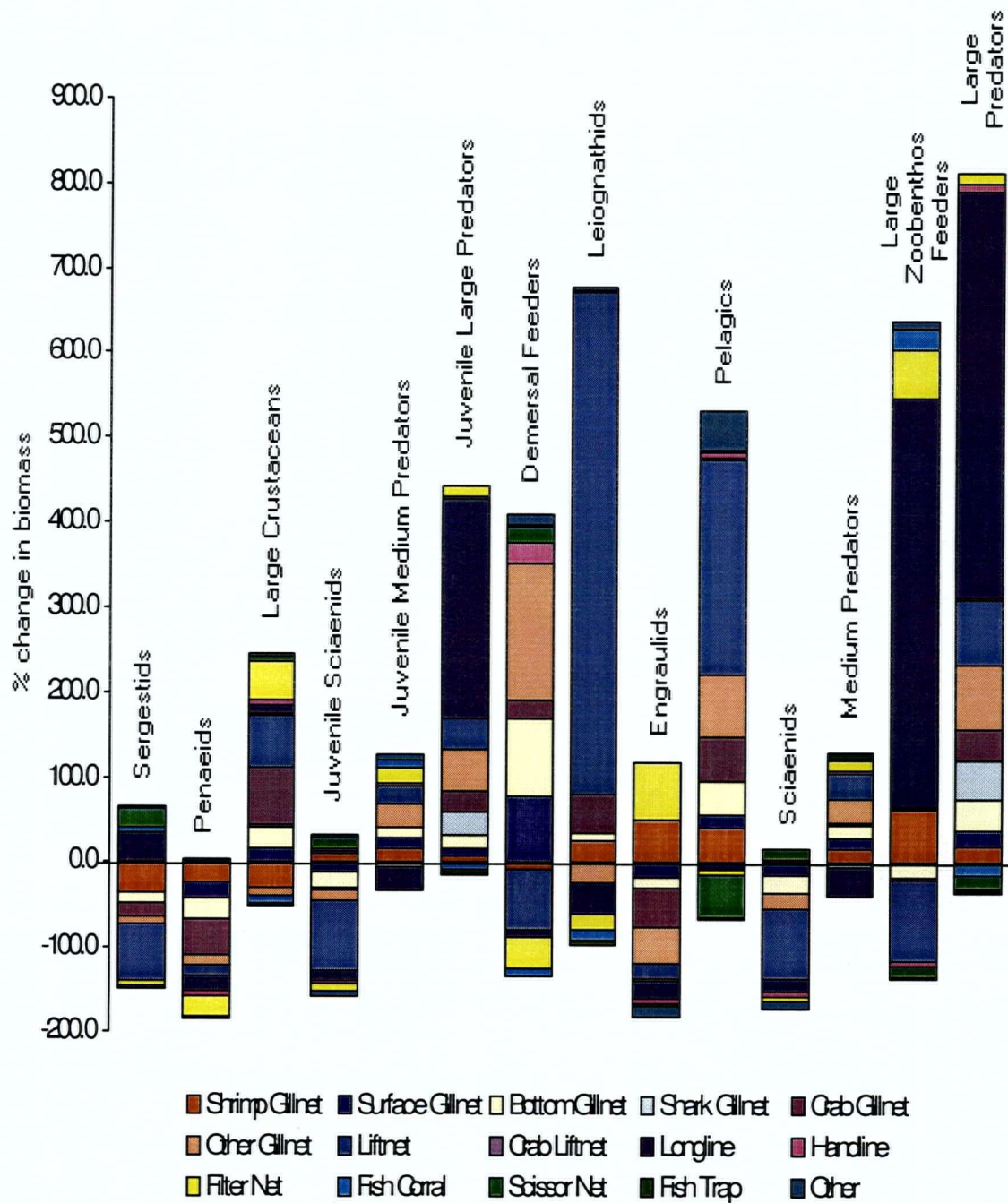


Figure 4.12 Percentage change in the biomass of each pool after 10 years when top-down control is assumed for all other gears. The effort of each gear is reduced to zero as described in the text and in Figure 4.1.

longline gear with top-down control. The leiognathids, demersal feeders, engraulids and sciaenids decrease by between 10% and 30% although none are caught by the set longline. They may decrease due to competition with the large zoobenthos feeders for food, and by predation and competition with the large predators. Interestingly, the medium predators decrease by 33% with top-down control, but increase by 6% with bottom-up control; the pelagics decrease by 5% with top-down control but increase by 18% with bottom-up control. Thus some pools respond in opposite directions to the two energy flow assumptions. With bottom-up control, the set longline only affects the pools that are caught by the gear: the biomass of the other pools do not change by more than 2%.

The other gillnets and the shrimp gillnet also have an impact on all other pools. The former has its greatest impact on the demersal feeders. However the other gillnet category is a composite group of gears and will not be discussed further here. Reduction in effort of the shrimp gillnets surprisingly leads to decreases in the biomass of the sergestids, penaeids and large crustaceans (the latter two are caught by shrimp gillnets). The biomass of the juvenile sciaenids and the engraulids, which are also caught by the gear, increase, by 50% in the case of the engraulids. Biomass increases are also seen for the juvenile medium predators, the juvenile large predators, the leiognathids, the pelagics, the medium predators, the large zoobenthos feeders and the large predators, none of which are caught. The picture is different for bottom-up control. Here the engraulids and the sciaenids increase and there is little change in any of the other pools.

Reducing effort of the filter net positively impacts the medium predators, large predators, engraulids and the sergestids under bottom-up control and produces a loss in biomass for the pelagics. This is a fine meshed gear that catches juvenile fish in addition to the targeted sergestids and engraulids. With the top-down assumption, every pool is affected by a reduction in effort by this gear. The crab gillnet similarly effects all pools under the top-down assumption, although this gear almost exclusively targets large crustaceans, with a small catch of demersal feeders, pelagics, adult sciaenids and the large zoobenthos feeders. With bottom-up control, there are small effects on other pools, but the main impact is the increase in the biomass of the large crustaceans and the sciaenids.

The changes that these gears produce in the total fished biomass lead to similar conclusions about which gears have most impact (Table 4.4). There are some exceptions though. Under top-down control, the shrimp gillnet effects very little overall change in total fished biomass. Effectively, the changes seen in the individual pools cancel one another out. The decrease in biomass of the large crustaceans and the penaeids is compensated by the increase in biomass of the other pools. Under the bottom-up assumption, the shrimp gillnet and the filter net have little effect on the total biomass.

Also shown in Table 4.4 is the proportion of the catch caught by each gear. Although intuitively one might assume that those gears which catch the greatest proportion of the catch have the biggest effect on the fishery, this is not necessarily the case. The results here show that the set longline, for example, causes the third greatest change in biomass and yet accounts for only 3% of the catch (top-down control). The crab gillnet, taking only 4% of the catch,

which is mostly large crustaceans, effects the fourth greatest change. With bottom-up control, the magnitude of the changes are generally less, but again gears which catch small proportions of the catch can cause relatively large changes in the total biomass. On the other hand, the mini-trawler, which accounts for 12% of the catch, produces smaller biomass changes than the crab gillnet, other gillnet or set longline. The trawl gear however, lands the greatest proportion of the catch and effects one of the greatest changes in biomass. Thus the impact of the various gear types in San Miguel Bay is not only related to their total catch, but also to the composition of their catch.

Throughout the description of the impacts of fishing gears on the ecosystem, there have been two recurring outcomes: (i) the biomass of pools which are not caught by a specific gear may be considerably affected by a change in fishing effort of that gear and (ii), the assumptions made about tropho-dynamic control can produce quite opposite predictions about direction of change of biomass.

(i) the biomass of pools which are not caught by a specific gear may be considerably affected by a change in fishing effort of that gear

The scissor net targets sergestids and these increase by 22% (top-down) and 12% (bottom-up). Additionally, the demersal feeders increase by 17% and the pelagics decrease by 50% under top-down control. The results in Table 4.3, show that the pelagics increase and the demersal feeders decrease when the biomass of sergestids is reduced. Thus the decrease in the pelagics

and the increase of the demersal feeders can be explained by the increase in the sergestid biomass.

However, not all responses are as clear cut. For example, the large zoobenthos feeders show considerable increases in biomass when the ordinary gillnet, hunting gillnet, shrimp gillnet and filter net effort are reduced to zero. The first two gears effect a very small level of fishing mortality on the large zoobenthos feeders ($F=0.003$), the latter two do not catch them. So what causes such large increases in large zoobenthos feeders? The only biomass response that all four gears have in common (when F is reduced to zero) is a decrease in the biomass of sergestids and the penaeids. The large zoobenthos feeders do increase in response to decreased biomass of the sergestids and penaeids (Table 4.3), but simulations changing the biomass of these two pools alone do not produce the increases seen in the large zoobenthos feeders biomass. The large zoobenthos feeders also increase when the leiognathids and demersal feeders biomass is decreased. But not all of the four aforementioned gears create this condition: the leiognathids and demersal feeders decrease when the ordinary gillnet and the filter net are reduced to zero, with the hunting gillnet they increase and only the demersal feeders decrease (slightly) in the case of the shrimp gillnet. This demonstrates that the interactions between pools and between pools and fishing gear can be complex and difficult to predict.

(ii) the assumptions made about tropho-dynamic control can produce quite opposite predictions about direction of change of biomass

The opposite reactions of some pools to the two tropho-dynamic assumptions are well illustrated in Figures 4.11-4.13. Most of the pools increase when effort is reduced under the bottom-up control assumption, while a more mixed response is produced with top-down control. This result was anticipated from the results of the ecosystem analysis above and is caused by the differential responses of the ecosystem components to top-down and bottom-up control. It was seen, for example, that the medium predators and the sciaenids exerted a strong repressive effect on lower pools under top-down control (Figures 4.6b, 4.7d), but a much weaker effect with bottom-up control. In Figure 4.11 (top-down control), when the ordinary gillnet effort is reduced to zero, the sciaenids and medium predators increase and the lower pools decrease. Under the bottom-up assumption, where the theory states that these pools are not limited by predation, the sciaenids and the medium predators increase when ordinary gillnet effort is decreased, and the lower pools also increase.

This pattern is observed not only for the ordinary gillnet. For all gears, some of the pools exhibit opposite responses to bottom-up and top-down assumptions. In many cases, under bottom-up control, the change in biomass is less than 1% while under top-down control the change can be as much as 200-300% (large zoobenthos feeders). For example, under top-down control, the large predators, demersal feeders, leiognathids, and engraulids decrease by up to 15% when the fish corral effort is decreased to zero, while these groups barely increase with the bottom-up assumption. But in other cases, both negative and positive changes are of

greater magnitude. This is the case for the trawl, ordinary gillnet, hunting gillnet, liftnet and set longline. For example, when trawl effort is reduced to zero under top-down control, the juvenile sciaenids, large predators and demersal feeders decrease: under bottom-up control they increase. The reverse is true for the large crustaceans. Clearly then, it is important to understand the dynamics of an ecosystem and what governs tropho-dynamic flows.

Three broad conclusions can be drawn from these results:

- (i) fishing gears effect a wider part of the ecosystem than simply the pools which they target,
- (ii) the assumptions made about tropho-dynamic control have considerable impact on the results,
- (iii) the impact a gear has on the fishery is strongly related to its catch composition in addition to the size of its catch.

A fourth conclusion can also be drawn from these results. The reduction or exclusion of only one gear from the fishery is unlikely to produce a biomass increase of sufficient magnitude nor over a sufficient breadth of pools to realise a sustainable and healthy fishery. In Figure 4.2, the equilibrium biomass is shown over a range of total fishing mortality. The biomass which corresponds with current fishing mortality is very low in almost all cases. Thus although a 50% increase in biomass may numerically sound like a large increase, in practice this may only increase the biomass by a small amount relative to pristine or desirable biomass. So there needs to be considerable reduction in fishing mortality in order to regain biomass and biodiversity. This is born out by the results from Chapter 2 where it was shown that trawl CPUE in 1992-1994 had decreased to around 28% of the 1947 value. From this perspective

then, the percentage changes in biomass shown in Table 4.4 are clearly inadequate, on their own, to effect much change. The implication is that a management plan, to reduce fishing effort in San Miguel Bay and improve the status of the fishery, would require reduction of two or more fishing gears.

When fishing effort is reduced completely in San Miguel Bay, much greater changes are effected (Figure 4.14). Here the time simulation is carried out for a 50 years period. With the bottom-up assumption, all pools increase with the exception of the three crustacean pools, which decrease slightly. Over the first 2-3 years, all the fish pools increase. The demersal feeders, leiognathids, pelagics and engraulids then stabilise and their biomass remains more or less the same over the remainder of the simulation. Other pools such as the sciaenids, the large predators and the juvenile medium predators take a little longer to stabilise. The medium predators and the large zoobenthos feeders increase exponentially to extremely high biomass levels. In the top-down simulation the large zoobenthos feeders also increase to very high biomass levels. The pools at the higher trophic levels in the ecosystem increase to greater biomass levels than in the bottom-up simulation. The pools at lower trophic levels achieve a lower biomass. The demersal feeders, pelagics and the three crustacean pools immediately decrease while the leiognathids and the pelagics decrease after an initial increase. All pools stabilise after about 30 years.

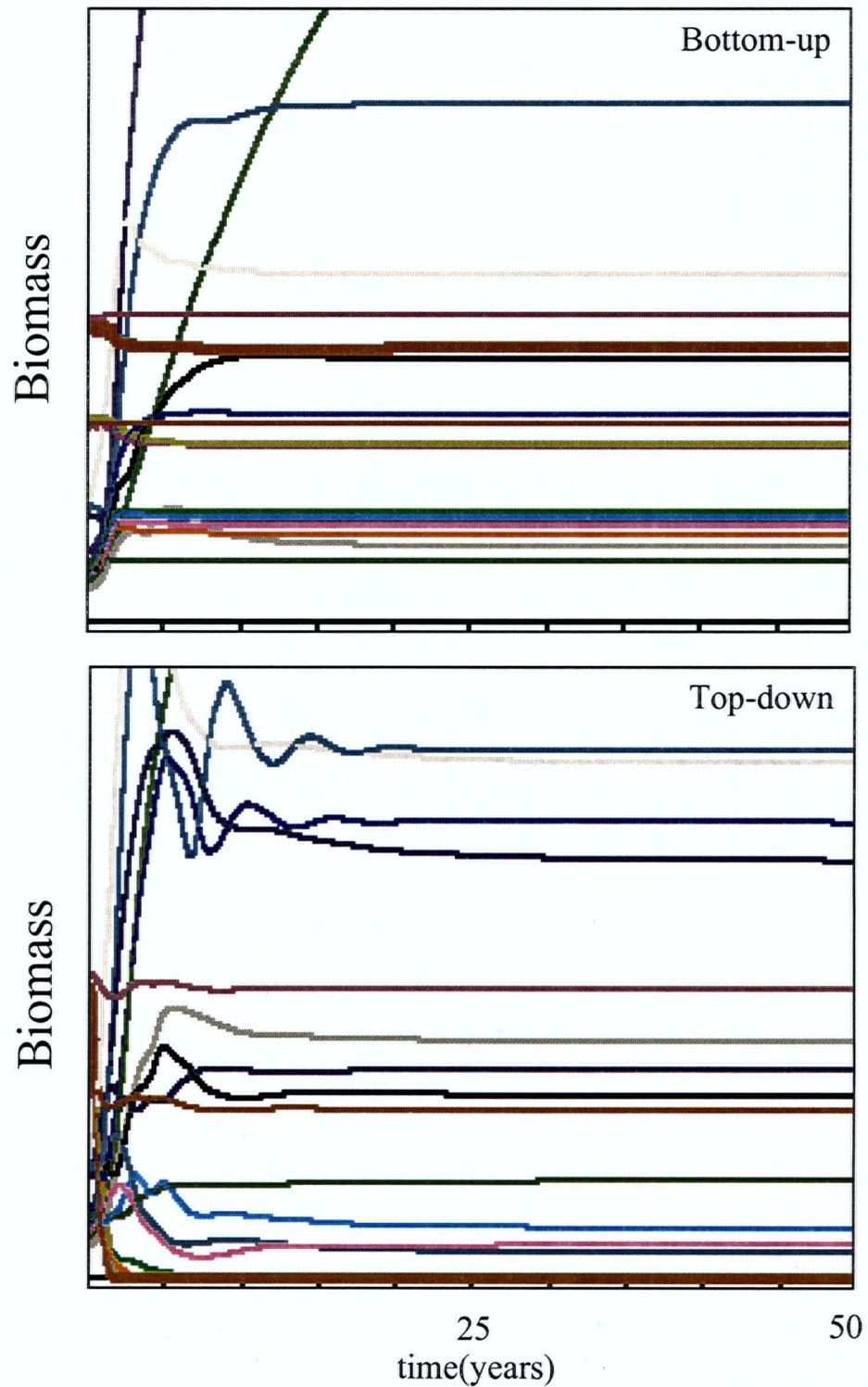


Figure 4.14. Dynamic simulations reducing total effort to zero. Effort is linearly reduced over the first 2 years and the simulation allowed to run for 50 years. The coloured lines are biomass, the thick line the Sergestids. See text for further details, Figure 4.1 for fishing pattern and Table 4.1 for legend and scale

Consistency Checks

The consumption/biomass (QB) ratios of all pools and the mean weights of the split pools were checked for individual gears and for individual pools using the Dynamic Run routine.

Impacts on Equilibrium Biomass of Changing Fishing Mortality Across All Fishing Gear

When the top-down assumption is made the QB ratios of most pools do not change by more than approximately plus or minus 50%. There are a few exceptions. The QB of the sergestids increases after the fishing mortality is reduced by about one third. It does not increase by more than 100%. At the same time, the large zoobenthos feeder's QB decreases. The high QB of the sergestids corresponds with a very low biomass (Figure 4.2). This could mean that there is surplus food available to other pools, pools which may not be represented in the model. On the other hand, the low QB of the large zoobenthos feeders corresponds with a high biomass: it is questionable whether, with such a low consumption rate the large zoobenthos feeders would in reality be able to persist, or if they would switch prey.

When fishing mortality is increased the QB of the juvenile sciaenids and large predators increase and the QB of the large predators decreases. These do not reach extreme values until fishing mortality is double its current value. Since fishing mortality is already very high, such extreme values may be considered unlikely to occur in reality.

The mean weights of the split pools show some variability: the juvenile pools and the sciaenids decrease when fishing mortality is decreased; when fishing mortality is increased, they increase initially. They do not differ from the initial values by more than a factor of 2. However, the results for the adult medium predators and the large predators are more concerning. When fishing mortality is decreased these mean weights rapidly increase to values several times higher than their initial values: when fishing mortality is increased, the opposite occurs, that is they decrease to several times their initial values.

A similar pattern occurs for the mean weight of the adult medium predators and large predators results under the bottom-up assumption. In addition, the other split pools also show extreme variation. When fishing mortality is decreased, they rapidly decrease. When fishing mortality is increased, they increase then decrease and flatten out (by this point their biomass has reached minimal levels). With the exception of small changes in fishing mortality, in the range of its possible levels, all split pool mean weights take on extreme values.

The QB results for the bottom-up assumptions are similar to those described for the top-down assumption; most pools do not vary by more than 50% from original values. The crustacean pools respond a little differently though. When fishing mortality is decreased none of the pools change (cf. the sergestids above): when fishing mortality is increased they all decrease a little. The QB of all the other pools decrease when fishing mortality is decreased and increase when fishing mortality is increased. The large zoobenthos feeders follow this pattern but have more extreme values.

The Effects of Fishing in San Miguel Bay

When fishing mortality is reduced to zero for each gear, no QB ratios vary by more than +/- 100% and in most cases the variation is considerably less. The mean weights of the split pools are generally within +/- 100% variation, with two exceptions. For the ordinary gillnet (bottom-up assumption), the mean weight of the adult medium predators increases by several fold. For the liftnet (top-down assumption), the mean weight of the adult medium predators also increases by several fold and the adult large predators increase to 2-3 times their initial mean weight.

When all gears are reduced to zero, more serious changes result in both parameters. Under the top-down assumption, the QB of all fished pools decreases, the large zoobenthos feeders by the greatest amount (this coincides with a large increase in biomass of large zoobenthos feeders seen above: so although the biomass increases, consumption is not able to keep up with it). The mean weights of adult medium predators and the large predators increase by several fold. The mean weight of the juvenile pools and the sciaenids decreased slightly. For the bottom-up simulation, QB remains fairly consistent. The mean weights all pools decrease by not more than 100%.

Reducing Fishing Effort on Each Pool to Zero

The results for the QB ratios are consistent, with one exception. The QB values change dramatically for the juvenile sciaenids when bottom-up control is assumed. Here, the QB of

the juveniles sciaenids increases several fold while the QB of the adults decreases by several fold. QB increases occur when sciaenid biomass decreases and QB decreases occur when biomass increases. The problems with the sciaenids and the bottom-up assumption were discussed above.

The results for the mean weights are much less consistent and the mean weights of the adult medium predators and large predators frequently rapidly increase or decrease when fishing mortality is reduced to zero on an individual pool. For top-down control, this occurs when the fishing mortality of the juvenile sciaenids, the juvenile medium predators, the adult medium predators, the demersal feeders, the leiognathids, the engraulids and the pelagics is reduced to zero. That is 50% of the 14 fished groups, or 7 out of 11 of the fish pools. For the bottom-up simulation, the same pools are affected, plus the juvenile large predators but not the engraulids.

Implications of the Consistency Checks

It is noted above (see footnote 2) that predictions made at the extremes of the model are the least reliable. This is the take home message for the results of the QB and mean weight analysis. That is, when dramatic changes are made away from the initial parameter system, predictions are less reliable (see also Walters *et al.* 1997).

The mean weights of the adult medium predators and large predators are particularly effected when the fishing mortality on individual pools is reduced to zero. However, initial fishing

mortalities are high and a reduction to zero is a substantial change. Simulations where fishing mortality is reduced, but not as far as zero, produce mean weight values which are more consistent. In addition the simulations where the effort of fishing gears was reduced to zero produced more reasonable results. Thus care needs to be exercised when modelling extreme situations and the values of these parameters should be checked.

The results of the mean weight analysis for the Equilibrium Fishing simulation are more cause for concern. Arguably they indicate that when total fishing mortality is increased or decreased by more than a relatively small amount from the current fishing mortality value, the model predictions begin to break down. Certainly the perturbations are extreme when fishing mortality by all gears is changed simultaneously. However, when an extreme top-down assumption is made (as in Figure 4.3), the response of the mean weights is less extreme. In the dynamic run above where the total fishing mortality was reduced to zero, the mean weights also responded in an extreme manner.

Of the three split pools, the results for the adult medium predators and large predators are the most uncertain. This could be due to the nature of the split pools and the amount of data available for their representative species. Both pools are more diverse than the third split pool, the sciaenids (see Chapter 3). In addition, there are less empirical data for these groups and more assumptions had to be made for the input parameters, both to the mass balance model (Chapter 3) and the split pools parameters (Appendix 2). This clearly indicates the need to obtain better data for these two groups. Future modelling should experiment with splitting the medium predators group.

Discussion

ECOSIM captures both thermodynamic consistency and the effects of harvesting on the components of the food web. It transports the “humble” mass-balance model into a dynamic system which has the ability to make predictions for the future. It also demonstrates the complexity of dealing with multispecies and multigear fisheries. The results described above give a glimpse of its potential application. However, there are some limitations to the approach.

ECOSIM begins with the assumption of equilibrium. In the mass-balance model, it is assumed that total production minus total loss is equivalent to zero for each pool. This is ECOSIMs departure point. It is this assumption, in part, which limits the usefulness of predictions that are far from the current fishing mortality and pattern. Walters *et al.* (1997) identify this as “perhaps the most worrisome” weakness of ECOSIM.

ECOSIM shares with ECOPATH the restrictions that grouping species into pools incurs. That is, in order to make the model tractable for a multispecies fishery with a broad range of species, it is necessary to group species together into pools. The pools are then parameterised using representative values, either from one representative species or from an average of values (see Chapter 3). In ECOSIM, this has two implications. The first is that when modelling an ecosystem dynamically, the assumptions made about the pool parameters are replicated throughout the simulation. So for each time step, the assumption is made. This creates a situation for potential addition or multiplication of errors and inaccuracies. Unfortunately this

is a necessary feature of the model if it is to be tractable. One way to circumvent this problem is to put species of particular importance, for example, the sciaenids in to separate pools. A parallel approach is to obtain better parameter definition.

This problem may be particularly relevant to the diet composition. Models with well-founded, empirical diet compositions perform better than models where the diets are derived at least in part from the literature on comparative species (Walters, *pers. comm.*). This is certainly one area where the San Miguel Bay model could be improved.

The second implication is that ECOSIM cannot model the changing composition of a pool. In the 1947 trawl survey for example, (Warfel and Manacop, 1950), the medium predators consisted mainly of Ariidae and Syphraenidae. The pool now consists largely of Trichiuridae. ECOSIM cannot simulate conditions where the Ariidae and Syphraenidae are specifically abundant unless they are modelled in separate pools. In the results above, the medium predators are a very successful pool, particularly when fishing mortality is reduced. However, this does not necessarily imply that there is an increase in the Ariidae or Syphraenidae.

Associated with the above is the inability of ECOSIM to predict prey-switching behaviour by pools (Walters *et al.* 1997). Diet is defined in the mass-balance model. In ECOSIM, the relative composition of the diet changes through time but pools cannot switch to another type of prey, not previously included in the diet. Similarly, ECOSIM cannot predict invasion by a "new" species into the ecosystem. In nature, species take-over do occur. For example, an ECOSIM model based on data from the 1947 trawl survey would not have predicted the

current biomass of the *Trichiurus* sp. because their biomass was insignificant then. With hindsight they could be modelled as a separate pool. Indeed, it could be argued that in the current model the *Trichiurus* sp. should be modelled separately.

As the pools are predators and competitors, so are the fishing gears. Just as a predator cannot suddenly consume a new prey, fishing gears cannot switch targets. The composition of their catch remains the same through time. This is somewhat unrealistic since fishers are known to switch targets, and that catch will vary with variation in the biomass composition.

The failure of ECOSIM to model the behaviour of the sciaenids under bottom-up control was perhaps the most vivid illustration of its limits. However, the alternative would be to develop a full age and size-structured model, with no approximations. This would re-introduce the problems of modelling stock-recruitment and density-dependent effects and mortality of juveniles when their biomass changes. In addition, it would re-introduce the big, parameter hungry model discussed earlier.

While recognising these limitations (see also Walters *et al.* 1997) ECOSIM is a fully dynamic multi-species ecosystem model. It is tractable, it can be used on many different types of aquatic ecosystems and it requires relatively few parameters. It can be used, as it was here, to make predictions about the ecosystem and its response to perturbations such as fishing or changes in fishing patterns. Its potential for future fisheries management is very exciting.

The results above confirm, complement and add to what is already known about the fishery of San Miguel Bay. They indicate that pool to pool interactions, the effect of fishing by different gear types on the ecosystem and the uncertainty about how trophodynamic flows are governed combine to make a challenging problem.

The plots in Figures 4.2 and 4.3 graphically illustrate the extent to which the fishery has been over-fished and the impact that this has had on the biomass of individual pools. Many are well below the commonly accepted standard of 30-50% unexploited biomass. Increasing fishing mortality much beyond its current values leads to the loss of most of the fish pools and large increases in the crustacean pools. The only pools that might withstand further general exploitation under either flow assumption are the crustaceans. The pools at the intermediate trophic levels might sustain further fishing effort if the bottom-up assumption is true, or if their predators are further reduced.

The analysis of individual pools showed that changes in the biomass of the medium predators, sciaenids, demersal feeders, leiognathids, engraulids and pelagics have strong impacts on the other pools in the model. The large predators, and surprisingly, the crustacean had little effect. The analysis also indicated that complex interactions involving competition and predation produce a web-like response to perturbations which is most extreme under the top-down assumption.

A comparison of the results from different types of yield curves indicated that it matters whether fishing mortality is considered on one species alone, or if fishing mortality is

considered across the whole fishery. Fishing on other pools can lead to conditions, whereby the biomass of the pool of interest changes (due to predation and competition), that result in a yield different from that which would be obtained from examining the effects of fishing mortality on that pool alone. The comparisons in Table 4.2 reveal that, to some degree, this occurs for almost all pools. Thus, not only does this analysis indicate the restricted nature of the traditional single species methods, it also demonstrates that it is vital to look further than just the species of interest.

These conclusions and their implications are reinforced by the analysis of the effects of fishing. It was demonstrated that fishing by one gear, be it a selective gear such as the scissor net or a less selective gear such as the gillnet can have a wide impact on the ecosystem. The biomass of pools which are not caught by a fishing gear can nonetheless undergo substantial changes. These changes are wrought by the change in biomass of the pools which are caught, through species interactions such as predation and competition. Moreover, the effect that an individual gear has on the ecosystem is not only related to its proportion of the total catch, but it is also strongly related to its catch composition. Although fishing gears can have quite dramatic impacts on the biomass of individual pools (Figures 4.11-13), their effect on total fished biomass is limited. It is suggested that in order to regain former biomass levels in San Miguel Bay, reduction in fishing effort across multiple gear types would be required.

Throughout all analyses, one feature was consistent: the assumptions made about trophodynamic flow have a large effect on predictions. When top-down control is simulated the biomass response is more varied and of greater magnitude and direction than when bottom-up

control is simulated. The results of the bottom-up analyses indicate a fishery in a healthier state than the top-down results. This is particularly evident in the results of the equilibrium yield curve analysis. When bottom-up control was assumed, most pools were either optimally or sub-optimally fished. The top-down analysis showed virtually all pools to be overfished. This result agrees with the findings of Walters *et al.* (1997), that is, that the bottom-up assumption gives significantly higher optimum fishing mortality values than the top-down assumption.

Bottom-up control was modelled in ECOSIM by setting the vulnerability to predation very low, thus making the mortality rate on prey nearly independent of the abundance of predators at the next trophic level, and top-down control was modelled by making mortality rate of prey strongly dependent on the abundance of predators at the next trophic level. For bottom-up control then, a change in the biomass of any eco-group will only impact on those ecogroups at higher trophic levels for which it is prey. The biomass of lower trophic levels will not be affected. For top-down control, a change in the biomass of an eco-group will impact in both directions, up and down the food web, as prey and as predator. Impacts of perturbation under top-down control, are thus more varied and larger than the impacts of perturbation under bottom-up control. This also explains why an ecosystem under bottom-up control is more stable than an ecosystem under top-down control.

Making the top-down assumption is in keeping with the precautionary principle (FAO 1995a,b). Since it is not known what kind of flow dynamics prevail in San Miguel Bay it is a more cautious approach to fisheries management to assume that top-down control occurs⁷⁹.

⁷⁹ However, top-down control can also give optimistic forecasts if certain outcomes are sought, such as an increase in crustacean biomass if fishing mortality was increased by another 100%.

However, bottom-up control is implicitly assumed in most fisheries assessment models, since predation mortality is held constant. Since the results of this analysis indicate that a top-down assumption is the more cautious approach to fisheries assessment, it would be wise to reconsider the sensibility of the assumption of constant predation mortality.

Fishing gears impact the ecosystem differently under different flow assumptions. For top-down control for example, a reduction in trawling leads to a decrease in the sciaenids, whereas for bottom-up control, the sciaenids increase. When the ordinary gillnet is reduced and top-down control simulated, the juvenile large predators, demersal feeders, leiognathids, engraulids and pelagics all decrease: when the assumption is bottom-up control, they all increase. Several examples of this type were given above. It is therefore quite critical to gain an understanding of both the effects of different flow assumptions, and insight into the nature of energy flow control in San Miguel Bay.

There is little basis on which to make a prediction about what governs energy flow in San Miguel Bay. Can the past give any indication? That is, when fishing mortality is reduced in the fishery in order to simulate the past, which assumption is best? A simulation was conducted using the fishing pattern of 1979-1982. The effort of some gears, such as the hunting gillnet, the shrimp gillnet and the other gillnets were reduced to zero, others such as the Surface gillnet and the Bottom gillnet were increased by two and four fold respectively. The results are given in Figure 4.15 and they were compared to the trawl biomass data from 1979-1982 and 1992-1994 in Table 2.6. Unfortunately they are mixed and do not give a clear picture. When top-down control was assumed, the three crustacean pools and the engraulids were reduced to very

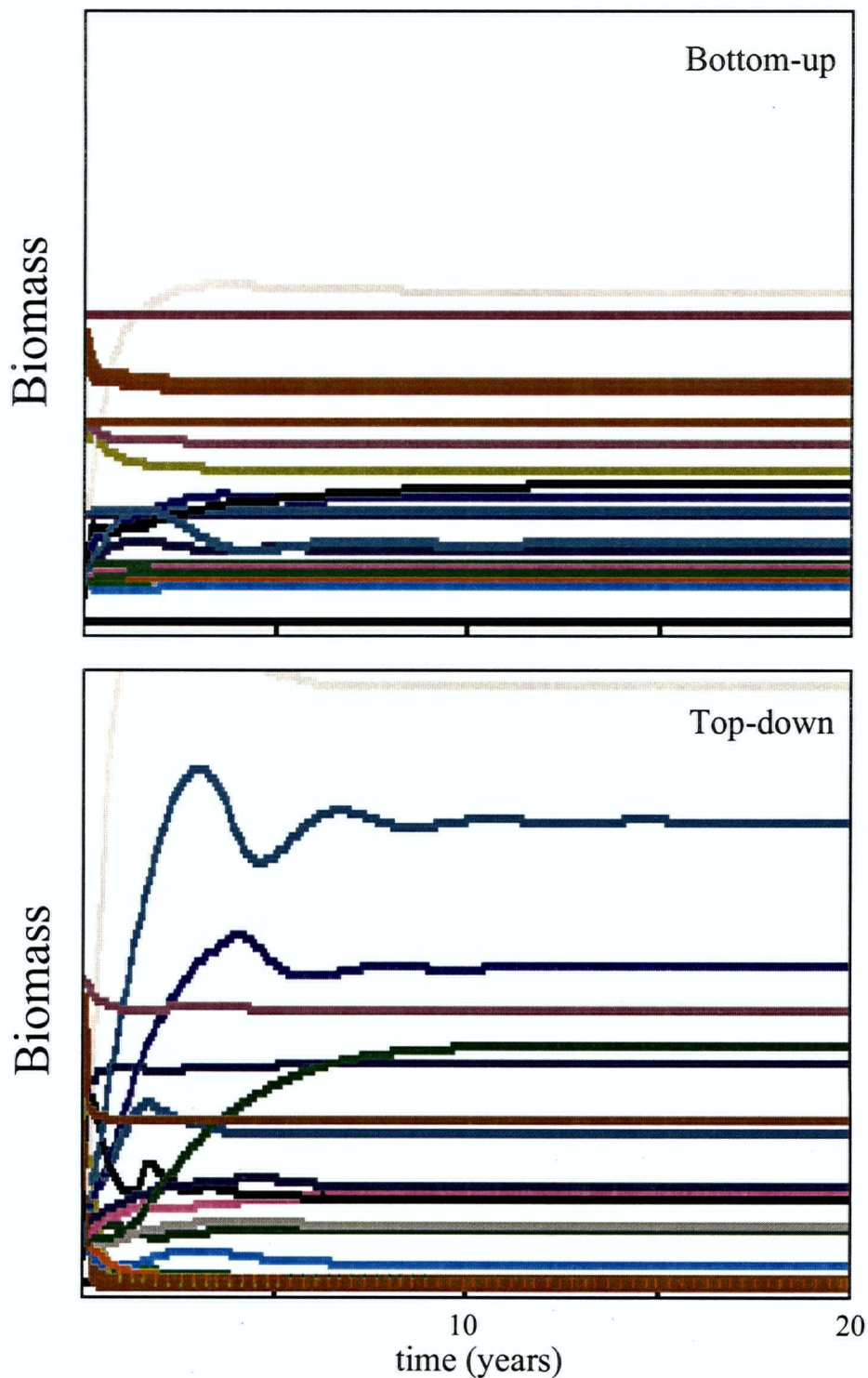


Figure 4.15. Dynamic simulation of the 1979-1982 fishing pattern. Effort is changed at the beginning of the simulation which is run for 20 years. The coloured lines are biomass, the thick line the Sergestids. See text for further details, Figure 4.1 for fishing pattern and Table 4.1 for legend and scale

low biomasses⁸⁰. However, they should increase, for hundreds of tons of these pools were caught in 1979-1982, and the biomass of the penaeids and engraulids was higher in 1979-1982 than in 1992-1994 (Table 2.6). It is not known what the biomass of the large crustaceans or the sergestids were in 1979-1982. In the bottom-up simulation, the biomass of the three crustacean pools decreased a little. The engraulids remained at about the same biomass. In Table 2.6, the biomass of the demersal feeders, the medium predators and the pelagics were less in 1979-1982 than 1992-1994. The decrease in demersal feeders is simulated using top-down control, but the decrease of the medium predators and pelagics are not reproduced by either assumption. In 1979-1982 the sciaenids and large predators were more abundant than in 1992-1994 and the leiognathid abundance was about the same. Both assumptions reproduce these results.

It is not possible to discriminate between the two flow hypotheses on the basis of these results. The lack of consistent comparability with the 1979-1982 trawl abundance data may be caused by several factors. The first is that simulation of the 1979-1982 fishing pattern, which involves reducing the effort of some gears to zero and increasing others by several fold, may be to take the model too far from the equilibrium starting point described above. A consistency check of the QB ratios and the mean weights showed some inconsistencies in both parameters. In addition, the fishing effort statistics for 1979-1982 may be less complete than the 1992-1994 data. That is, some gears recorded in 1992-1994 but not in 1979-1982 may have been present but recorded in a separate category. In Chapter 3 it was shown that there was not very good accordance between the pool biomasses estimated by ECOPATH and the biomasses from the

⁸⁰ This result is comparable to reducing all effort to zero. In Figure 4.14, these same pools were reduced to very low biomass when total effort was reduced to zero.

trawl survey. Since ECOSIM begins with the ECOPATH biomasses, then it may not be realistic to expect it to be able to reproduce the biomasses from 1979-1982. Finally, it remains to be seen whether it is indeed possible to reproduce a previous ecosystem. That is, is nature reversible? It was noted above that ECOSIM cannot simulate species replacement or takeovers. Certainly these have occurred in San Miguel Bay, as changes in habitat have occurred due to fishing.

The top-down assumption is a strong paradigm in the marine setting (Menge 1992). There is however growing evidence in support for some contribution from bottom-up influences. So far, there are no data to suggest that bottom-up control does not influence flow dynamics in San Miguel Bay. The Bay has a large detrital input from the 12 river systems that drain into it (Silvestre 1996). This may contribute to productivity in the Bay. Indeed, as suggested in the previous chapter, it could be this input that has enabled the fishery to be sustained to the extent that it has. In addition, San Miguel Bay has an important nursery role for many fish. It is as likely as not that juvenile fish are immediately affected by bottom-up control more than top-down control, in spite of predation by larger fish.

In conclusion, it is not possible on the basis of available data and knowledge to determine what governs energy flow in San Miguel Bay. However, the results clearly demonstrate that it is critical that some insight be gained into this uncertainty. This uncertainty makes it difficult to ascertain what the best management approach for the fishery of San Miguel Bay would be. In simple terms, if a long term sustainable fishery is sought, the answer is to reduce fishing mortality. However, other than reducing effort across the whole fishery simultaneously it is not

immediately clear from this analysis what this would entail. Therefore it would be necessary to devise both different management strategies for each type of control and to develop a means to discriminate between the two hypotheses. Assuming bottom-up control should be an easier task, since reductions in fishing effort lead to biomass increases in most cases. The top-down assumption makes the task more challenging. The top down effects of predation and trophic cascades lead to a variable biomass response. An adaptive management approach is used to address these questions in the next chapter.

Chapter 5

Management Strategies for San Miguel Bay

“The construction of a decision table to assess EVPI is useful beyond providing a test of whether further adaptive policy calculations are worthwhile. It forces an evaluation of the best policy implied by each alternative hypothesis and helps define a range of uncertainty about the best policy. It forces a preliminary assessment of what performance measures should be used to compare policy alternatives. With luck, it helps uncover a robust policy that will do well no matter what model is correct.”

Hilborn and Walters (1992)

Introduction

Three properties define the fishery of San Miguel Bay. It is (1) a multispecies fishery that is (2) over-fished by (3) a diverse multi-gear effort. When the fishery was assessed using length-based single species methods, over-exploitation was found to occur. In addition, a multispecies assessment using ECOPATH and ECOSIM indicated that when bottom-up control is assumed, the fishery is at an optimal to high state of exploitation. Moreover, when top-down control is assumed, almost all species are highly over-exploited. However, the multispecies assessments also demonstrated that interspecific interactions and assumptions made about the control of energy flow in the ecosystem, can profoundly affect predictions. Multispecies simulations of the impact of different fishing gears clearly indicated the differential effects of bottom-up and top-down assumptions. It was concluded that in order to better manage the fishery of San Miguel Bay, a greater understanding should be obtained of the dynamics of the ecosystem. This conclusion is explored further in this chapter.

To date, there has been little effective management of San Miguel Bay (see Chapter 2), and measures such as mesh size restrictions and even trawl bans have been largely unsuccessful. However, events in San Miguel Bay have overtaken both fisheries science and the lack of management. Sunderlin (1994) reports that, in response to lower incomes and increased pressure on the resource, continued competition with the large-scale sector, three initiatives towards community management have occurred in San Miguel Bay. First, the number of fisher's organisation in San Miguel Bay has increased substantially since 1979-1982 (Pomeroy *et al.* 1995, Pomeroy and Pido 1995, Sunderlin 1994). Second, two operations were established in 1992 to enforce laws against illegal trawling⁸¹. The third was the development, then establishment of the San Miguel Bay Management Council (SMBMC) in 1993⁸². Sunderlin attributes this move towards community management to the small-scale fishers, and suggests that, although the idea of the SMBMC was first presented by Smith *et al.* (1983) and elaborated by government planners, the SMBMC "could not have been implemented in its current form without the evident determination of Bay fishers to have a role in governing their livelihoods" (1994: 228).

Community management has been widely discussed (e.g., McGoodwin 1990, Bailey and Jentoft 1990, Pinkerton 1989). The origin of community management is a key factor for its success. If community management originates within the community, as in San Miguel Bay, there is a much greater chance for its success than for community management regimes which

⁸¹ The two agencies are called the *Bantay Dagat* (Defense of the Sea) and *Task Force Kalikasan* (Task Force for Nature). In the space of one year (mid 1993-mid 1993) they apprehended and confiscated the gear of more than 100 trawlers (Sunderlin 1994).

⁸² The San Miguel Bay Management Council is a hierarchical institution organised at four levels with horizontal linkages. At the top is the regional development council (government). Next is the SMBMC which consists of 31 members, including 11 fishers. At the next level is the municipal task force and at the bottom, the village level task forces (Pomeroy *et al.* 1995).

are imposed on a community from without, by the government, for example. An additional factor that is likely to enhance the success of community-based management in San Miguel Bay is the Local Government Code (LGC), which devolved authority to Local Government Units (LGU) in 1992 (Luna 1992). Through the LGC and several other initiatives, the Philippine government actively promotes community-based resource management (Pomeroy and Pido 1995, Abregana *et al.* 1996). San Miguel Bay is thus completely under the authority of local, or “municipal” government control. There are two direct consequences of the SMBMC. First, the committee brings together the the seven municipalities which border the Bay, and thus the Bay can be managed as a whole, instead of by the individual municipalities. Secondly, management can be focused locally instead of part of a grand plan for the entire Philippine archipelago.

To this end, the SMBMC formed an interim planning committee (IPC) to draft an integrated fisheries management plan for San Miguel Bay. The SMBMC also invited ICLARM to assist the IPC (Luna 1995a). The scope of the IPC was wide and included environmental, social and political considerations as well as biological and economic concerns. C. Luna, used Decision Analysis⁸³ to help develop an integrated management plan. In the development of this plan, the

⁸³ Decision analysis is a method developed to quantitatively discriminate between alternative decisions or options where multiple criteria or objectives impinge on that decision. There is an expansive literature on decision analysis but two theories, the Multi-Attribute Utility Theory (Keeney and Raiffa 1976) and the Analytic Hierarchy Process (Saaty 1980) are the main methods used for discrete multiple criteria decision making problems (Belton 1986). Simply described, decision analysis first deconstructs a problem into a series of hierarchically related objectives or attributes. It then discriminates between various alternative solutions to the problem based on the weighted objectives. It does this by scoring the alternatives against the objectives then adding the products across the range of objectives for each of the alternatives. This produces a rank ordering of the alternatives. Decision analysis is a group method whereby the views of the stakeholders are incorporated into the decision making process. It is they who decided on the relative weights of the objectives and score the success of alternatives against the objectives.

management recommendations of Smith *et al.* (1983), which completed the 1979-1982 ICLARM research in San Miguel Bay, were also considered.

The nine members of the IPC participated in the decision analysis. They consisted of two mayors, a commercial trawl operator, a "baby" trawl operator, a small-scale fisher, a university professor of fisheries, an NGO representative, and two government representatives, one from the Department of Agriculture and one from the Asian Development Bank (ADB) Fisheries Sector Program. Thus both stakeholders and decision makers were represented in the IPC. Luna (1995a) notes that some of these participants were major protagonists in the conflict between the large-scale trawling sector and the small-scale sector. He provides an example: "[f]or instance, the municipal fishers' representative and his group once tried to make a citizens arrest on the commercial trawler representative, allegedly for illegal trawling in the Bay. In turn, the latter filed a case of illegal arrest, resulting in the imprisonment of the municipal representative for a few days. During the trial, the NGO representative acted as legal council to the municipal representative. Meanwhile, at the northern end of the Bay, the mayor of Mercedes, who was conducting an enforcement campaign against trawlers, twice arrested boats owned by the commercial trawling representative" (Luna 1995a). Although achieving consensus amongst these old adversaries might be considered challenging, progress and consensus decisions were made (Luna 1995b). This can be regarded as a promising start to the community management of San Miguel Bay.

Management Objectives in San Miguel Bay

The committee agreed on an overall objective, the “optimal management of San Miguel Bay’s coastal fisheries”. The main management objectives identified to reach this goal were sustainable exploitation, habitat protection, maximise economic benefits, to promote equity, maximise the acceptability of interventions and to optimise the administrative side of management. Twelve alternative projects were scored against these objectives (Luna 1995b). As a result of the decision analysis, four projects were defined as urgent, (1) Mangrove reforestation, (2) Improvement of law enforcement capabilities, (3) Phase out of municipal trawlers and (4) Enactment of an intermunicipal ordinance to ban commercial trawling. The other projects were classified as either necessary, or, desirable but deferrable⁸⁴.

Only the third and fourth projects involve direct fishery interventions. Since the trawling sector is a non-selective mass harvester, it is expected that these interventions would lead to an increase in biomass available to the fishery. Indeed, the results of the multispecies assessment in Chapter 4 demonstrated this. When simulated trawling effort was reduced to zero for 10 years⁸⁵, the total fished biomass increased by 21.7% and 14.8% for the top-down and bottom-up assumptions about energy flow control. However, the top-down results mask a great deal of variability in the response of different components of the ecosystem. Some eco-groups, such as the sciaenids and demersal feeders, decreased after 10 years without trawling. For both

⁸⁴ The other alternative projects were: (5) Construction of a road link between Siruma and Tinambac, (see Figure 2.1), (6) Improvement of Mercedes fish port, (7) Upland reforestation, (8) Marine Sanctuaries, (9) Improvement of Calabanga jetty, (10) Stabilisation of critical agricultural lands, (11) Improvement of fisheries licensing system, (12) Establishment of ice plant and storage facility (Luna 1995b)

⁸⁵ This did not include the Mini Trawlers.

assumptions, the biomass of the three crustacean groups decreased. Economically, these are some of the most valuable species in the Bay (Padilla *et al.* 1995). Thus, a total reduction in trawling may not bring about all the rewards anticipated, especially if top-down control has more influence than bottom-up control.

Cruz-Trinidad and Garces (1997) used linear programming to examine optimum management strategies in San Miguel Bay. With an objective function to maximise net revenue in the fishery, they used linear modelling to determine the optimal fleet configuration. Their objective function included the costs of fishing in terms of material and fixed expenses and the cost of labour. Constraints were also included. The total catch was set not to increase beyond 14, 000 tonnes per annum, the catch was allocated between trawl gears, gillnets and fixed gears, effort by the trawl fleet had to be less than the current effort and wage rates had to be greater or equal to the opportunity cost of labour. The optimum fleet configuration produced by this method consisted of a six fold increase in the number of fish corrals and a four fold decrease in the number of baby trawlers. When the current fleet configuration was maintained, profits were still made in the fishery, but at one third of the level of the optimum scenario. Simulating this optimum fleet configuration with ECOSIM produced an increase of 5% in total fished biomass, assuming bottom-up control, and 16% assuming top-down control. Again, the latter figure masks a great deal of variation in the biomass change of individual eco-groups.

The IPC alternative projects, listed in Luna (1995a, b) and the optimum fleet scenarios of Cruz-Trinidad and Garces, do not address the multispecies nature of the San Miguel Bay

fishery. Nor do the IPC identify specific fisheries ecological objectives⁸⁶. The sustainability of the resource is at the top of the list of objectives, but it is not clear quite what this means, other than to maintain overall biomass. In this chapter, the “sustainable” management of San Miguel Bay is addressed from a multispecies perspective. Various management options, based as far as possible on the stated objectives of the IPC, are investigated using simulation modelling with a multispecies model, ECOSIM. The management options are examined for their effect on the total fished biomass in San Miguel Bay, the total catch in San Miguel Bay, the effect on the biomass of individual eco-groups, the effect on the catch distribution amongst gears, the total profit made in the fishery and the profit per small-scale gear.

However, two major uncertainties have been identified in San Miguel Bay. The first is the nature of tropho-dynamic flow. This was explored in Chapter 4 where it was demonstrated that assumptions of top-down or bottom-up control produced quite different results. The question is, “what are the management implications of this uncertainty?” A top-down assumption implies that predator-prey interactions in a multispecies system impact the system and therefore the fishery. A bottom-up assumption implies that predator-prey interactions are less important. In the multispecies model used here, a bottom-up assumption used in conjunction with assessment of fishery yields, resembles a series of single species analyses. Thus, the question is also, “are multispecies considerations important in the management of multi-gear fisheries which consist of many species?”.

⁸⁶ Luna (1995a) provides an Appendix, containing a much larger list of objectives than those which ended up in the decision tree. These are the results of the initial rounds of identifying the objectives in San Miguel Bay. Many of the objectives are focused on law enforcement, and reducing effort. However, there are a couple of objectives which address specific species in the Bay. One is to “Increase production of exportable species to earn foreign exchange”, the other to “Conduct studies of shrimp because these are the major species in San Miguel Bay fisheries.”

The second major uncertainty in San Miguel Bay is the extent to which the fish inside the Bay are replenished from outside, that is, the extent of immigration and then, emigration. This was discussed in Chapters 2 and 3. In Chapter 3 it was argued that the ECOPATH model, as it was constructed, allowed for the possibility of emigration and immigration and was valid.

However, the uncertainty over the role of immigration and emigration in the fishery and their contribution to the sustainability of the fishery remains. A fishery which has an immigration component is likely to be more stable and resilient than a self sufficient fishery, assuming that the immigrants are not subject to high exploitation elsewhere. In San Miguel Bay, the area outside the Bay is much less intensively fished than inside the Bay.

How important are these uncertainties? This question was addressed using adaptive management, a methodology which explicitly addresses uncertainty. Simulations were made for each of the management options, using bottom-up and top-down control assumptions, and assumptions about immigration.

Adaptive Management in San Miguel Bay

Adaptive management is a method which explicitly addresses our uncertainty about resource dynamics and uses feedback information to increase knowledge. In adaptive management, hypotheses or models are made about key uncertainties in the resource system (Walters 1986). Short term, experimental management policies are then developed to discriminate between the alternative models. After an experimental period where the fishery is carefully monitored for

response to the chosen policy, information from this period is fed into the decision making process for the next management step. If all goes well, it may be possible to place more certainty on one model than the other and proceed with a longer term strategy, based on the greater certainty placed on that model. Alternatively, it may be necessary to undergo another experimental or short term management period before a longer term management strategy is set. Adaptive management should be a continuous feedback system where the responses of the resource to management are used constructively to inform future management decisions.

There are two types of policy in adaptive management, non-adaptive or passive policies and adaptive or active policies. A passive policy is one that does not deliberately probe in order to reduce uncertainty, but is the best policy for a given model. An actively adaptive policy is one which is designed to evoke a specific response and thus help to reduce uncertainty. Adaptive management does not have to be active to be adaptive. The adaptive component of the term refers to the use of feedback information from actions to revise future management. In developing adaptive management strategies, a large amount of time is spent determining whether an active or passive approach is the most suitable. An active, experimental approach is not always called for.

In formal application of adaptive management, Bayesian statistics are employed to place initial probabilities on the uncertainties about different resource models or hypotheses (Hilborn *et al.* 1994, Walters and Ludwig 1994, Walters 1986, Lindley 1983). Simulation modeling of the models under different short and long term policies then produce updated probabilities for

the models. Statistical decision theory is subsequently used to work out which is the best overall strategy to follow, using criteria such as “expected value”.

The fishery of San Miguel Bay falls into Walters (1986) classification of a “complex problem”. Complex problems may be systems where there are too many variables, where consensus cannot be reached about a small set of alternative hypotheses about uncertainties, or where no single objective function will represent the conflicting interests of the various actors involved on management (1986:333). At least the first and last of these apply to San Miguel Bay. It is not possible to place posterior probabilities on the different resource models because there is insufficient historical data. In addition, the Monte-Carlo simulation approach required to generate updated probabilities is not appropriate for the multispecies model used here. There are a very large number of variables and the wood would indeed be lost in the trees. Finally, it has been suggested that no-one would believe the probabilities placed on the different resource models anyway (Carl Walters, pers. comm.). Thus it is assumed that all models are equally likely and the various policies options are deterministically simulated and their impacts determined.

There are six basic steps in adaptive management (Walters 1986, Hilborn and Walters 1992):

1. Identify alternative hypotheses or models of the resource;
2. assessment of whether further steps are necessary by estimating the expected value of perfect information (EVPI);
3. develop baseline policies;

4. develop adaptive, probing policy options;
5. develop performance criteria to measure success;
6. formal comparison of options.

In practice, these steps are not all distinct. In order to estimate EVPI, one needs a range of policies by which to judge the alternative hypotheses. Clearly, some of these will be included in the policies developed at step 3, and possibly at step 4. Again, in order to assess the EVPI, it is necessary to have a criterion against which the alternative models and policies are judged. This is not necessarily separate from step 5. Arrival at step 6 is an interactive process, which may be iterated several times.

Step 2 is a critical step in adaptive management (Hilborn and Walters 1992). It places bounds on the importance of learning by calculating the difference between knowing with absolute certainty which model is right, that is, having perfect knowledge, and not knowing which model is right. The calculation of EVPI is a calculation of the importance of learning. It is at this step that decisions are made about the utility of an experimental approach. Surprisingly often, according to Walters, "learning will not be as valuable as we intuitively expect" (1986:195)

Walters (1986) recommends using a decision table to determine the EVPI. The table contains the predictions of each model, assuming that there is perfect information, over of a range of policies. The best policy for each model (U^{**}) is determined. The best policy averaged over all models (U^*), that is, the best non-adaptive policy, is also determined. The EVPI is calculated

as the average of the best policy for each model minus the best non-adaptive policy. More formally, this is the difference between average gain using U^{**} instead of U^* , that is,

$$EVPI = \sum_i P_i (M_i) [V(U^{**}|M_i) - (V(U^*|M_i))]$$

where, M_i = model (i), P_i = probability of M_i , and $V(U|M_i)$ = expected value of policy U , given model M_i (Walters 1986:197).

The EVPI for the resource models and policy options in San Miguel Bay is calculated below. The uncertainties are first described and alternative models developed to represent these uncertainties. A range of likely long term policy options for San Miguel Bay are then outlined and various performance criteria are developed. The simulations were run for 20 years for each model and policy option. Decision tables were made for each of the performance criteria and the EVPI estimated.

Identify Alternative Hypotheses or Models of the Resource

The uncertainties in San Miguel Bay were discussed above. ECOSIM was used to directly model the key uncertainties in San Miguel Bay. Model 1 simulates bottom-up control and model 2 simulates top-down control. They were modelled as in Chapter 4. Immigration was modelled using a routine in ECOSIM. ECOSIM is programmed to include a very small rate of immigration, in this case, 1/1000 of the start biomass. Immigration was modelled by making

the immigration rate equivalent to the start biomass for all the fished groups except for the large crustaceans (see Chapter 3). However, it was not possible to make the immigration rate that high for the juvenile groups, because of the way their the initial equilibrium balance is calculated⁸⁷. The number of immigrants plus the number of juveniles recruits must equal the initial equilibrium juvenile mortality rate. If the immigration rate is too high, this implies negative juvenile recruitment (C. Walters, pers. comm.). This is an unfortunate consequence of the structure of ECOSIM. Thus the juvenile sciaenids were modelled with an immigration rate equivalent to 1/10 of their starting biomass, while the juvenile medium predators immigration rate was 1/100 of their starting biomass. It was not possible to model immigration for the juvenile large predators⁸⁸. Emigration is modelled as the immigration rate divided by the initial starting biomass. Model 3 simulated immigration with bottom-up control and Model 4 simulated immigration with top-down control.

Potential Management Policies in San Miguel Bay

To be fair to everyone, and to consider all major fish species in San Miguel Bay, it would be necessary to devise at least as many different management strategies for the fishery as there are people and species: it is a complex situation and there are many management permutations. In addition to the biological complexity of the resource and the diverse range of fishing methods used, management is confounded by equity considerations. The current social

⁸⁷ The split pools are accounted for in ECOSIM using both biomass and numbers, unlike the unsplit pools which are tracked using biomass alone. See Chapter 4 and Walters *et al.* (1997).

⁸⁸ A sensitivity analysis of the juvenile and adult parameters, K, T and W_k indicated that the large predators and medium predators were particularly sensitive to the T parameter, the age at which they mature into adults.

and local political situation is anti-trawling and trawlers are held to be responsible for many of the ills in the fishery (see Gorospe 1995, Pomeroy *et al.* 1995). However, it is not possible to optimise for every fishing gear and for every species, for every situation. A simple approach is more useful.

The eleven policies in Table 5.1 were developed as feasible long term options for the fishery. They are based on the results of the multispecies analysis conducted in Chapter 4, and on predictions of the likely effects of immigration and emigration in San Miguel Bay. They also reflect the stated management objectives in San Miguel Bay and include policies in line with current events in the Bay. Also included are a few, extreme policies, necessary for comparative purposes. Walters (1986) suggested that extreme policies are the most informative policies.

Policy 1 is the optimum policy developed by the IPC (Luna 1995b)⁸⁹. Since the SMBMC adopted the recommendations of the IPC and are going ahead with the plan to phase out all trawling (C. Luna and M. Pido, pers. comm), this may be regarded as the baseline policy.

Policy 2 is the optimum policy developed by Cruz-Trinidad and Garces (1997).

⁸⁹ However, it does not include banning mini-trawlers, for two reasons. First, their catch is targetted on the penaeid and sergestid shrimp, and although the mesh size used is small, the trawlers also fish at low speeds (they have 16 HP engines). Secondly, the impact of reducing the mini-trawler effort to zero was much less than either the ordinary gillnet, hunting gillnet or the large, medium and baby trawlers (Chapter 4). Since this sector is quite large (260 vessels) and still makes a profit (see Chapter 2 and Padilla *et al.* 1995), and does not have a large negative impact on the biomass or biomass composition it was retained as the one type of trawling allowed in San Miguel Bay.

Table 5.1 List of potential long term management options for San Miguel Bay.

| <i>Long Term Policy Number</i> | <i>Long Term Policy</i> |
|------------------------------------|---|
| 1 | Ban all trawling by large, medium and baby trawlers |
| 2 | Reduce trawling by baby trawlers from 50 vessels to 11 vessels and increase the number of fish corrals from 123 to 767. |
| 3 | Ban all trawling by large, medium and baby trawlers and increase the number of crab gillnets by 100%. |
| 4 | Ban all trawling by large, medium and baby trawlers and ban the use of the hunting gillnet. |
| 5 | Ban all trawling by large, medium and baby trawlers and ban the use of the filter net. |
| 6 | Ban all trawling by large, medium and baby trawlers and ban the use of the hunting gillnet and the filter net. |
| 7 | Ban all trawling by large, medium and baby trawlers and reduce effort by other gears by 25% (except the crab gillnet, crab liftnet, set longline, handline, fish corral, scissor net, fish trap and "others" gear). |
| 8 | Ban all trawling by large, medium and baby trawlers and reduce effort by all other gears by 50%. |
| 9 | Reduce effort of all types of gillnet by 50%. |
| 10 | Increase effort of all gears by 33% |
| 11 | Increase effort of all gears by 33%, except the crab gillnet which should be increased by 100%. |

In Chapter 4 it was concluded that reducing trawling alone was not sufficient to adequately address the over-fishing and over-exploitation problems in San Miguel Bay. In addition to the trawl ban, policies 3-8 involve a range of other management measures. Thus, policies 3-8 ban all trawling, with the exception of the mini-trawler.

Policy 3 increases the crab gillnet effort by 100%. The simulations in Chapter 4 showed that the biomass of the large crustaceans increased when trawling was reduced, particularly under top-down control. Policy 4 bans the hunting gillnet. Gillnets, especially the hunting gillnet and the ordinary gillnet have a substantial impact on the resource (see Figure 4.9). The hunting gillnet was banned in this policy rather than the ordinary gillnet because it is an active gear. In addition, they are a recent development in the Bay (Chapter 2) and for that reason, may be easier to ban. The ordinary gillnet is a traditional fishing method. The bottom-up assumption predicts that decreasing effort of the Hunting gillnet has a positive impact on the resource (Table 4.4).

Policy 5 bans the use of the filter net. This reflects opinion voiced by the IPC. One of their objectives was the ban of the filter net (Luna 1995b, Appendix 1). This ban was called for because the filter net, which is a fine meshed gear, is perceived to catch the juveniles of many species, and therefore have a negative impact on the biomass and on catches. The multispecies analysis of the filter net in Chapter 4 demonstrated that this gear does impact all eco-groups, although not all impacts are negative. In Chapter 2 it was noted that there has been an increase in the use of fine-meshed gears in San Miguel Bay. The filter net has seen the most substantial increase by over 300% from 60 units in 1979-1982 to 260 in 1992-1994 (Table 2.13).

In Policy 6 the effect of banning both the hunting gillnet and the filter net are examined.

Since an aim of the SMBMC is to reduce effort in San Miguel Bay, the next two policies are designed to investigate the impact of reducing effort further, that is by taking more drastic action. All effort is reduced by 25% in policy 7, except for the crab gillnet, crab liftnet, set longline, handline, fish corral, scissor net, fish trap and "others" gear. These were excluded for they are relatively benign gears and could be considered "environmentally friendly". Policy 8 is more extreme. All effort is reduced by 50%, in addition to the trawl ban.

Collectively the gillnets impose a high mortality in the fishery, they are very numerous and are at least as great a threat to the sustainability of the resource as the trawlers, as demonstrated in Chapters 3 and 4. For this reason, policy 9 was designed to examine the impact of reducing all gillnets by 50%. The impact of this reduction should be more beneficial under the bottom-up than the top-down assumption.

If fish immigration⁹⁰ is a significant factor in San Miguel Bay, then it should be possible to expand the fishery and to increase employment. The last two policies, 10 and 11, were designed to examine the impact of increasing effort in the fishery. The multispecies analysis in Chapter 4 also indicated that there would be a large increase in the biomass of the crustacean groups if effort were increased, especially under top-down control. In both policies, effort is increased by 33%. Policy 11 also increases the crab gillnet effort by 100%.

⁹⁰ The term fish immigration refers to both the fish and crustacean species.

Performance Criteria

There are many criteria which could be used as a measure of success in a multispecies fishery, such as total biomass, total catch, net revenue and number of jobs. It is clear from the objectives listed by the IPC that these factors are all concerns in San Miguel Bay. Performance criteria can also include the costs of management, monitoring costs, loss of revenue compensation costs, socio-economic costs or the estimated costs of not doing an experiment. If sustainability, multispecies issues and equity issues were a concern, a measure of species diversity and the equity of the distribution of the catch across gears could be used as performance criteria.

Sainsbury (1991) maximised the expected present value for the multispecies fishery of the Northwestern Shelf of Australia. Sainsbury's expected present value included the costs of fishing, the costs of doing the experiment and social costs. The expected present value is a useful performance criteria, but it does not distinguish between either the effect of policies on the biomass or its composition, or on the distribution of catch across gears. Walters and Hilborn (1976) used long term catch, discounted over time. But there may be no "best" performance criteria in a fishery such as San Miguel Bay where there are multiple management objectives. Hilborn and Walters (1992) recommend using a set of performance criteria if there is uncertainty as to which is best, then to conduct a series of assessments based on those criteria. This suggestion is followed here.

Usually, the performance criteria are discounted over a long time horizon, or to infinity. This method is not possible with the multispecies model used here. The following procedure was used. Each policy change was implemented over two years. Effort was either increased or decreased linearly over this time. Simulations were then performed to determine the number of years it took for the ecosystem to stabilise after the perturbation. By year 20, all biomasses had stabilised for all models and policies. Since the concern at this stage is to determine whether learning is beneficial to the long term future of the resource, the results of each policy were compared after 20 years. This ignores the path by which the final values after 20 years are reached. However, since all the policies are considered valid long term options, it is the long term view which is relevant, not the short term path.

The performance criteria that were developed reflect concerns about sustainability of the resource, concern about maintaining the diversity of the multispecies resource, concerns to maximise catch, to maximise equity in catch distribution and to maximise revenue. At this stage, no attempt is made to discriminate between the different criteria. The criteria are listed in Table 5.2.

The first criteria is the percentage change in "total fished biomass". An increase in biomass is beneficial to the fishery, a decrease is not. The second criteria was developed to measure the impact of policies on the diversity of the biomass. The rational is that more diversity is better.

Table 5.2 Performance Criteria for the first stage of the adaptive management

| <i>Performance Criteria</i> | <i>Units</i> |
|------------------------------|--|
| Total fished biomass | % change after 20 years |
| Biomass Diversity | Difference between the number of eco-groups that increased in biomass after 20 years and those that decreased. |
| Total Catch | % change after 20 years |
| Catch Distribution | Difference between the number of gears whose catch increased after 20 years and those that decreased. |
| Total Revenue | % change after 20 years |
| Revenue per small-scale gear | % change after 20 years/number of small-scale gears |

The criteria, “biomass diversity” is calculated from the difference between the number of eco-groups which decrease in biomass and the number of eco-groups which increase in biomass. All increases or decreases greater than 5% are regarded as equally good or bad⁹¹. Changes less than 5% were not included in the count. The best result is an increase in the biomass of all eco-groups, and conversely, the worst result is a decrease in the biomass of all groups. This criteria differs from the first criteria because an increase in total fished biomass could be due to a large increase in one eco-group and smaller decreases in many other eco-groups.

The third criteria is the percentage change in “total catch”. From the normal fisher point of view, increase in catch is good. The fourth criteria, “catch distribution” measures the distribution of the change in catch over all the gears in the fishery, including the trawling sector. As with the second criteria, the number of gears whose catch increases is subtracted from the number of gears whose catch decreases. This criteria measures the equity of the distribution of the catch benefits⁹².

The last two criteria are economic. They are simply based on the net revenue, that is, the total revenue from fishing minus the costs of going fishing plus fixed costs. Labour costs are not included. The economic data were estimated from Padilla *et al.* 1995. It is assumed that when effort is altered by x%, the costs correspondingly change by x%. Criteria 5, “total revenue”

⁹¹ This criteria could be more complex. For example, if an increase in the biomass, of say the penaeids and the large crustaceans was deemed desirable, then these could be given a greater weight when their biomass increased, and a greater negative effect on the criteria when their biomass decreased.

⁹² If it was desirable to increase the catch of specific gears, these gears could be weighted more heavily than the other gears.

measures the percentage change in total net revenue. Criteria 6 measures, “revenue per small-scale gear” measure the percentage change in net revenue per small-scale gear⁹³.

Assessment of Whether Further Steps are Necessary by Estimating the Expected Value of Perfect Information, EVPI

EVPI for all Policies

Decision tables showing the expected values for each of the performance criteria are given in Tables 5.3 to 5.8. The optimum policy for each option, assuming perfect knowledge, is marked **, and the best non-adaptive policy is marked *.

It is immediately clear from the results that two policies dominate the rest. For each of the alternative resource models, one policy is always the best. These dominant policies are both extreme policies. Policy 8, where the trawl gears are reduced to zero and all other effort decreased by 50%, is the dominant policy when total fished biomass and revenue per small-scale gear are the performance measures. Policy 11, where all effort is increased by 33% and the crab gillnet is increased by 100%, is the dominant policy when the performance criteria are total catch, total revenue or distribution of catch. Biomass diversity is the only performance criteria that shows a variable response to the different resource models. Policy 8 is the best non-adaptive policy in this case.

⁹³ The total net revenue is simply divided by the total number of small scale gears at time 0 and at time 20.

Table 5.3 Percentage Change in Total Fished Biomass after 20 years.

| <i>Management Policy</i> | <i>Model 1</i> | <i>Model 2</i> | <i>Model 3</i> | <i>Model 4</i> | <i>Average</i> |
|--------------------------|----------------|----------------|----------------|----------------|----------------|
| 1 | 15.8 | 24.0 | 11.1 | 10.7 | 15.4 |
| 2 | 3.9 | 15.5 | 2.6 | 4.0 | 6.5 |
| 3 | 11.5 | 18.5 | 7.7 | 6.8 | 11.1 |
| 4 | 25.3 | 18.0 | 19.1 | 15.3 | 19.4 |
| 5 | 20.3 | 40.2 | 14.4 | 14.9 | 22.5 |
| 6 | 28.3 | 21.5 | 22.4 | 18.0 | 22.6 |
| 7 | 29.7 | 21.9 | 19.1 | 15.3 | 21.5 |
| 8 | 79.2** | 113.6** | 60.9** | 56.3** | 77.5* |
| 9 | 23.2 | 15.4 | 18.9 | 15.3 | 18.2 |
| 10 | 7.2 | 43.5 | -7.9 | 5.5 | 12.1 |
| 11 | -2.9 | 17.1 | -11.5 | -0.8 | 0.5 |

** = optimum policy for each model,* = best policy averaged over all models

Table 5.4 Change in Biomass Diversity after 20 years

| <i>Management Policy</i> | <i>Model 1</i> | <i>Model 2</i> | <i>Model 3</i> | <i>Model 4</i> | <i>Average</i> |
|--------------------------|----------------|----------------|----------------|----------------|----------------|
| 1 | 8 | 0 | 8 | 5 | 5.3 |
| 2 | 5 | 4 | 4 | 3 | 4.0 |
| 3 | 7 | -1 | 7 | 3 | 4.0 |
| 4 | 8 | -2 | 8 | 6 | 5.0 |
| 5 | 9** | -1 | 9** | 5 | 5.5 |
| 6 | 8 | 1 | 8 | 7 | 6.0 |
| 7 | 9** | -1 | 5 | 8** | 5.3 |
| 8 | 8 | 2 | 8 | 8** | 6.5* |
| 9 | 9** | 6** | 6 | 5 | 6.5* |
| 10 | -1 | -4 | -8 | -9 | -5.5 |
| 11 | 0 | -3 | -10 | -8 | -5.3 |

** = optimum policy for each model,* = best policy averaged over all models

Table 5.5 Percentage Change in Total Catch after 20 years.

| <i>Management Policy</i> | <i>Model 1</i> | <i>Model 2</i> | <i>Model 3</i> | <i>Model 4</i> | <i>Average</i> |
|--------------------------|----------------|----------------|----------------|----------------|----------------|
| 1 | -5.3 | -16.9 | -8.8 | -15.3 | -11.6 |
| 2 | -2.7 | -6.3 | -3.0 | -6.2 | -4.6 |
| 3 | -3.3 | -14.2 | -6.3 | -12.8 | -9.2 |
| 4 | -6.8 | -19.2 | -6.8 | -19.2 | -13.0 |
| 5 | -8.9 | -23.2 | -12.7 | -21.9 | -16.7 |
| 6 | -12.2 | -26.1 | -16.5 | -21.6 | -19.1 |
| 7 | -13.3 | -26.1 | -20.57 | -26.5 | -21.6 |
| 8 | -28.5 | -31.5 | -31.4 | -35.5 | -31.7 |
| 9 | -5.9 | -5.3 | -11.1 | -11.3 | -8.4 |
| 10 | 24.5 | 24.2 | 18.1 | 23.0 | 22.5 |
| 11 | 26.3** | 29.3** | 19.4** | 26.9** | 25.5* |

** = optimum policy for each model,* = best policy averaged over all models

Table 5.6 Change in Catch Distribution after 20 years

| <i>Management Policy</i> | <i>Model 1</i> | <i>Model 2</i> | <i>Model 3</i> | <i>Model 4</i> | <i>Average</i> |
|--------------------------|----------------|----------------|----------------|----------------|----------------|
| 1 | 11 | -4 | 12 | 5 | 6.0 |
| 2 | 7 | -4 | 6 | 1 | 2.5 |
| 3 | 12 | -1 | 12 | 5 | 7.0 |
| 4 | 7 | -6 | 7 | 3 | 2.8 |
| 5 | 10 | -6 | 11 | 4 | 4.8 |
| 6 | 5 | -7 | 6 | 3 | 1.8 |
| 7 | 6 | -6 | 0 | -5 | -1.3 |
| 8 | -6 | -5 | -9 | -4 | -6.0 |
| 9 | -3 | 2 | -3 | -1 | -1.3 |
| 10 | 13 | 3 | 17 | -2 | 7.8 |
| 11 | 14** | 8** | 16** | 2** | 10.0* |

** = optimum policy for each model,* = best policy averaged over all models

Table 5.7 Percentage Change in Total Revenue after 20 years

| <i>Management Policy</i> | <i>Model 1</i> | <i>Model 2</i> | <i>Model 3</i> | <i>Model 4</i> | <i>Average</i> |
|--------------------------|----------------|----------------|----------------|----------------|----------------|
| 1 | -8.4 | -21.6 | -11.0 | -15.5 | -14.1 |
| 2 | 0.7 | -3.8 | 0.3 | -2.6 | -1.4 |
| 3 | -2.9 | -8.5 | -4.5 | -9.4 | -6.3 |
| 4 | -13.4 | -42.5 | -17.0 | -26.9 | -25.0 |
| 5 | -11.4 | -24.6 | -14.1 | -21.8 | -18.0 |
| 6 | -17.8 | -48.7 | -20.0 | -31.6 | -29.5 |
| 7 | -18.5 | -37.0 | -23.0 | -29.0 | -26.9 |
| 8 | -37.7 | -48.6 | -40.1 | -46.6 | -43.3 |
| 9 | -15.1 | -27.0 | -19.4 | -24.7 | -21.6 |
| 10 | 47.9 | 62.1 | 24.6 | 40.0 | 43.7 |
| 11 | 48.0** | 69.5** | 28.3** | 48.6** | 48.6* |

** = optimum policy for each model, * = best policy averaged over all models

Table 5.8 Percentage Change in Profit per Small-scale Gear after 20 years

| <i>Management Policy</i> | <i>Model 1</i> | <i>Model 2</i> | <i>Model 3</i> | <i>Model 4</i> | <i>Average</i> |
|--------------------------|----------------|----------------|----------------|----------------|----------------|
| 1 | 58.3 | -20.8 | 42.5 | 15.0 | 23.8 |
| 2 | 28.3 | 5.0 | 26.7 | 10.0 | 17.5 |
| 3 | 46.7 | 15.8 | 38.3 | 10.8 | 27.9 |
| 4 | 65.8 | -119.2 | 42.5 | -20.0 | -7.7 |
| 5 | 53.3 | -30.0 | 36.7 | -11.7 | 12.1 |
| 6 | -30.8 | -157.5 | 37.5 | -40.8 | -47.9 |
| 7 | 116.7 | -14.2 | 80.8 | 42.5 | 56.5 |
| 8 | 255.0** | 125.0** | 226.7** | 149.2** | 189.0* |
| 9 | 75.8 | -10.0 | 52.5 | 15.0 | 33.3 |
| 10 | 55.8 | 151.7 | -29.2 | 39.2 | 54.4 |
| 11 | 35.8 | 141.7 | -31.7 | 47.5 | 48.3 |

** = optimum policy for each model, * = best policy averaged over all models

There is thus no need to estimate EVPI, for the EVPI equals zero. This means that there is no value in future learning. This result occurs because the changes in fishing effort incurred by these policies are so large that they force the resource in a particular direction, regardless of the assumptions about the resource. That is, the impact of the fishery on the resource, under these policies is much greater than the differences between the different resource models. This does not mean that there is no value in trying to learn more about the uncertainties for different policy options, but it does mean that for each of these resource policies, the particular model used to model the resource is not critical.

It is difficult to judge whether such extreme policies would be acceptable to the SMBMC, and therefore whether they are valid management options. Since the actions of the SMBMC are in favour of reducing effort in the fishery and improvement of the resource base, the two policies, 10 and 11, which increase effort throughout the fishery and decrease the biomass of many of the eco-groups, are unlikely to be enacted. Following either of these options would be very risky, and would not be recommended without further testing the ECOSIM predictions. In the absence of directed management however, they could, in practise occur by default.

In terms of the sustainability and health of the resource, policy 8 is by far the best policy. It also gives the highest revenue per gear, although following this policy means a loss of 50% of the jobs in the fishery plus the loss of onshore jobs. One of the management objectives of the IPC was to "promote equity amongst coastal resource users" (Luna 1995a). Clearly this policy is not equitable for those who "lose out". However, the IPC objectives also include the "provision of alternative livelihoods" and many of the projects listed in the San Miguel Bay

Integrated Coastal Fisheries Management Plan (unpublished manuscript) address this concern. If it proved possible to provide those who lose out with alternative livelihoods, policy 8 would be feasible.

EVPI when Extreme Policies are Excluded

The results above suggest that learning more about the community dynamics in San Miguel Bay or the role of immigration is unimportant to fisheries management. However this may not be the case. Not all management actions will be as extreme as the two dominant policies, and thus may not overwhelm the resource dynamics. The decision tables are re-drafted in Tables 5.9 - 5.14, excluding the extreme policies. The optimum policy for each option and the best non-adaptive policy are marked as before.

The results show that there is still consistency in the pattern of a particular policy to be the optimum for most models. In the case of the “total revenue”, policy 2 is dominant (this policy was derived from the work of Cruz-Trinidad and Garces (1997), who used profits maximisation as the objective function), and the EVPI is zero. For “total catch”, “catch distribution” and “revenue per small-scale gear”, policies 2, 3 and 7 are optimum for all models with the exception of model 2, the top-down model with no immigration. The EVPIs are all low. For “total biomass”, there is a similar picture. Policy 6 is the optimum policy for models 3 and 4. Policy 7 is the best policy for model 1, but policy 6 is a close second. The EVPI for “total biomass” is considerably higher than the EVPIs for the other performance

Table 5.9 Percentage Change in Total Fished Biomass after 20 years.

EVPI = 5.02

| <i>Management Policy</i> | <i>Model 1</i> | <i>Model 2</i> | <i>Model 3</i> | <i>Model 4</i> | <i>Average</i> |
|--------------------------|----------------|----------------|----------------|----------------|----------------|
| 1 | 15.8 | 24.0 | 11.1 | 10.7 | 15.4 |
| 2 | 3.9 | 15.5 | 2.6 | 4.0 | 6.5 |
| 3 | 11.5 | 18.5 | 7.7 | 6.8 | 11.1 |
| 4 | 25.3 | 18.0 | 19.1 | 15.3 | 19.4 |
| 5 | 20.3 | 40.2** | 14.4 | 14.9 | 22.5 |
| 6 | 28.3 | 21.5 | 22.4** | 18.0** | 22.6* |
| 7 | 29.7** | 21.9 | 19.1 | 15.3 | 21.5 |
| 9 | 23.2 | 15.4 | 18.9 | 15.3 | 18.2 |

Table 5.10 Change in Biomass Diversity after 20 years

EVPI = 1.5

| <i>Management Policy</i> | <i>Model 1</i> | <i>Model 2</i> | <i>Model 3</i> | <i>Model 4</i> | <i>Average</i> |
|--------------------------|----------------|----------------|----------------|----------------|----------------|
| 1 | 8 | 0 | 8 | 5 | 5.3 |
| 2 | 5 | 4 | 4 | 3 | 4.0 |
| 3 | 7 | -1 | 7 | 3 | 4.0 |
| 4 | 8 | -2 | 8 | 6 | 5.0 |
| 5 | 9** | -1 | 9** | 5 | 5.5 |
| 6 | 8 | 1 | 8 | 7 | 6.0 |
| 7 | 9** | -1 | 5 | 8** | 5.3 |
| 9 | 9** | 6** | 6 | 5 | 6.5* |

Table 5.11 Percentage Change in Total Catch after 20 years.

EVPI = 0.25

| <i>Management Policy</i> | <i>Model 1</i> | <i>Model 2</i> | <i>Model 3</i> | <i>Model 4</i> | <i>Average</i> |
|--------------------------|----------------|----------------|----------------|----------------|----------------|
| 1 | -5.3 | -16.9 | -8.8 | -15.3 | -11.6 |
| 2 | -2.7** | -6.3 | -3.0** | -6.2** | -4.6* |
| 3 | -3.3 | -14.2 | -6.3 | -12.8 | -9.2 |
| 4 | -6.8 | -19.2 | -6.8 | -19.2 | -13.0 |
| 5 | -8.9 | -23.2 | -12.7 | -21.9 | -16.7 |
| 6 | -12.2 | -26.1 | -16.5 | -21.6 | -19.1 |
| 7 | -13.3 | -26.1 | -20.6 | -26.5 | -21.6 |
| 9 | -5.9 | -5.3** | -11.1 | -11.3 | -8.4 |

** = optimum policy for each model, * = best policy averaged over all models

Table 5.12 Change in Catch Distribution after 20 years
EVPI = 0.75

| <i>Management Policy</i> | <i>Model 1</i> | <i>Model 2</i> | <i>Model 3</i> | <i>Model 4</i> | <i>Average</i> |
|--------------------------|----------------|----------------|----------------|----------------|----------------|
| 1 | 11 | -4 | 12** | 5** | 6.0 |
| 2 | 7 | -4 | 6 | 1 | 2.5 |
| 3 | 12** | -1 | 12** | 5** | 7.0* |
| 4 | 7 | -6 | 7 | 3 | 2.8 |
| 5 | 10 | -6 | 11 | 4 | 4.8 |
| 6 | 5 | -7 | 6 | 3 | 1.8 |
| 7 | 6 | -6 | 0 | -5 | -1.3 |
| 9 | -3 | 2** | -3 | -1 | -1.3 |

Table 5.13 Percentage Change in Total Revenue after 20 years
EVPI = 0

| <i>Management Policy</i> | <i>Model 1</i> | <i>Model 2</i> | <i>Model 3</i> | <i>Model 4</i> | <i>Average</i> |
|--------------------------|----------------|----------------|----------------|----------------|----------------|
| 1 | -8.4 | -21.6 | -11.0 | -15.5 | -14.1 |
| 2 | 0.7** | -3.8** | 0.3** | -2.6** | -1.4* |
| 3 | -2.9 | -8.5 | -4.5 | -9.4 | -6.3 |
| 4 | -13.4 | -42.5 | -17.0 | -26.9 | -25.0 |
| 5 | -11.4 | -24.6 | -14.1 | -21.8 | -18.0 |
| 6 | -17.8 | -48.7 | -20.0 | -31.6 | -29.5 |
| 7 | -18.5 | -37.0 | -23.0 | -29.0 | -26.9 |
| 9 | -15.1 | -27.0 | -19.4 | -24.7 | -21.6 |

Table 5.14 Percentage Change in Profit per Small-scale Gear after 20 years
EVPI = 0.9

| <i>Management Policy</i> | <i>Model 1</i> | <i>Model 2</i> | <i>Model 3</i> | <i>Model 4</i> | <i>Average</i> |
|--------------------------|----------------|----------------|----------------|----------------|----------------|
| 1 | 58.3 | -20.8 | 42.5 | 15.0 | 23.8 |
| 2 | 28.3 | 5.0 | 26.7 | 10.0 | 17.5 |
| 3 | 46.7 | 15.8** | 38.3 | 10.8 | 27.9 |
| 4 | 65.8 | -119.2 | 42.5 | -20.0 | -7.7 |
| 5 | 53.3 | -30.0 | 36.7 | -11.7 | 12.1 |
| 6 | -30.8 | -157.5 | 37.5 | -40.8 | -47.9 |
| 7 | 116.7** | -14.2 | 80.8** | 42.5** | 56.5* |
| 9 | 75.8 | -10.0 | 52.5 | 15.0 | 33.3 |

** = optimum policy for each model, * = best policy averaged over all models

criteria. This reflects the benefit to be gained from discriminating model 2 from the other models.

The performance criteria, “biomass diversity”, discriminates between the immigration models, between the immigration and non-immigration models, but it does not discriminate between the two non-immigration models. The best non-adaptive policy, policy 9, is also optimal for models 1 and 2. The EVPI of 1.5 is quite high relative to the value of the best non-adaptive policy.

For all the performance criteria except “biomass diversity”, the two immigration models behave the same as model 1, the bottom-up model and no immigration. This may mean three things. It could mean that there is no need to model immigration, but to assume it will give results equivalent to the bottom-up assumption. Alternatively it could mean that there is no point in modelling both bottom-up and top-down control if immigration does occur. It could also mean that immigration only makes a difference if some form of top-down control is the true model.

Faced with these results a manager would in all likelihood conclude that there was no real value to be gained from future learning about the resource. For one criteria there is a dominant policy and for four others there are robust non-adaptive policies. The only performance criteria that produced high EVPIs were “total biomass” and “biomass diversity”. However, even if the main management concern was to increase biomass, policy 6 would be a fairly safe bet. It is

the best non-adaptive policy for total biomass and the second best policy for biomass composition.

Walters (1986) cites four basic conditions to justify a probing policy:

1. there must be uncertainty about the best policy to take,
2. system performance must be sensitive to the action taken,
3. alternative actions must be differentially informative,
4. no single one of the alternative models for the systems response must be considered very probable.

The results above do not meet the first condition, although the three other conditions are met.

An experimental approach however, is not warranted.

The Role of Immigration

The inclusion of fish immigration in the model structure produced a system that is more stable than when there is no immigration. Changes in biomass are less severe, whether they are increases or decreases. It is reasonable that these results are similar to the results of model 1 since a bottom-up assumption about tropho-dynamic control is also an assumption about stability in the system. Quantitative differences exist between bottom-up and top-down control when there is immigration, but the buffering effect of immigration masks their import.

If model 1 were correct, it would not be possible to tell whether the true model were model 1, 3 or 4. None of the performance criteria distinguishes well between these models. This points to a problem with the criteria. If it had proven useful to use an experimental approach, different criteria would have to be used to distinguish the two immigration models from one another and from model 1.

Six Performance Criteria and Five Best Non-Adaptive Policies

The six performance criteria used were chosen to reflect various aspects of the stated management objectives in San Miguel Bay and to capture some of the impacts relevant to a multispecies approach. The different criteria highlight different management issues at stake in San Miguel Bay. There is clearly a trade-off between conserving biomass and maintenance of total catch or total revenue. When 11 policies were compared in the decision tables, two of the policies dominated the rest. When these two, and one other, extreme policies were excluded from the analysis, five best non-adaptive policies resulted. Policy 2 was the best non-adaptive policy for the “total catch” and “total revenue” criteria. The other criteria all have different best non-adaptive policies. What are the management implications of this mélange of policies?

All the policies include the trawling ban except policy 2 and policy 9. The latter reduced all gillnet effort by 50% whilst policy 2 decreased the number of baby trawlers from 50 to 11 and increased the number of fish corrals from 123 to 767. Policy 9 is unlikely to be politically acceptable, because of the loss of employment in the traditional gillnet sector, but it does

produce the second best non-adaptive policy for catch in addition to the best non-adaptive policy for biomass diversity, and the second best non-adaptive policy for revenue per gear. These features make it quite an attractive policy.

Policy 2 is also an attractive policy. The SMBMC have already decided to ban all trawling, so a substantial decrease in the number of baby trawlers should not present a problem. In addition it substantially increases the number of fish corrals and thus employment in the small-scale sector. However, it scores very badly for the “total biomass” and “biomass diversity” criteria.

Policies 3, 6 and 7 ban trawling. Policy 3 also increases the crab gillnet effort by 100%, and performs well against the “total catch”, “total revenue” and “revenue per small-scale gear” criteria. Like policy 2 though, it does not do well when scored against the “total biomass” and “biomass diversity” criteria. Policy 6 also bans the hunting gillnet and the filter net. The only criteria for which this policy performs well are “total biomass” and “biomass diversity”. It performs badly on all four other criteria. Policy 5 which bans only the filter net in addition to trawling behaves in a similar fashion. Policy 7 reduces most other gears by 25% in addition to the trawl ban. Because of the big gear reduction, it performs well on the “revenue per small-scale gear” criterion, and fair on the “total biomass” and “biomass diversity” criteria, but does not do well on the three other criteria.

The current management policy of banning all commercial trawling and phasing out the municipal trawlers is not a best non-adaptive policy for any of the performance criteria. This indicates that other options should be considered for the San Miguel Bay fishery. Since the

trawl ban has already begun, advantage could be taken of this initiative for change and progress towards sustainable fishing.

The choice of management options is a classic decision analysis problem, and indeed this would be the most appropriate approach to determine the best management action in San Miguel Bay⁹⁴. The six criteria would address the ecological, economic and social objectives of the IPC decision tree (Luna 1995a).

Discussion

The results indicate that, for the management options considered, whether the extreme policies are included or not, there is no value in distinguishing between the different resource models in San Miguel Bay, in learning more about the resource dynamics or the role of immigration. For each performance criteria, there is a robust policy for all models. These results also suggest that multispecies considerations are unimportant to the broad goals of fisheries assessment and management. These are unexpected findings and fall within Walters' description of learning being less valuable than we would intuitively expect. This does not, however, mean that adaptive management should not be the preferred management method in San Miguel Bay. A passive adaptive management can still abundantly benefit the resource and the resource users (Walters 1986).

⁹⁴ See footnote 83 for a description of this method.

The concern about the interaction between fishing and resource dynamics is well expressed by Kitchell (1996) who asserts that “..fishing causes much of the variability observed in ecosystem processes and, therefore, much of the uncertainty in resource management practices. Reducing that uncertainty requires a better understanding of food web and ecosystem process responses to fishery effects” (1996:17). The results presented here support the first part of this statement, that fishing causes much of the variability, but they do not support the idea that a better understanding of foodwebs etc., will be particularly useful to resource management practices, at least from the fishery perspective. It was suggested that the changes in fishing pattern induced by the management policies overwhelmed the differences between the different resource models. The San Miguel Bay fishery has been seriously over-exploited for over a decade. It is feasible that the system is easily overwhelmed in this vulnerable state, and that the same policies invoked on a more resilient system would have a different effect.

There are two sets of questions which follow from this result. The first is whether there would ever be a set of conditions where it would make sense to discriminate between the different resource models in San Miguel Bay. The multispecies analyses conducted in Chapter 4 implied that this would be so⁹⁵. The second set of questions concerns the methodology used here, which is at variance with that prescribed by Walters (1986) and others.

Some Notes on the Non-Bayesian Approach Used

The method used here to draw up decision tables and assess the EVPI differs from the

⁹⁵ Note that these simulations were made without considering fish immigration.

approach described in by Walters (1986). Walters' recommends using a Bayesian approach where uncertainty is recognised explicitly by placing initial probabilities on the different resource models and examining how the uncertainty propagates through time, using Monte Carlo simulation. As noted above, this was not possible here. Clearly this is a simplification, but would a Bayesian approach have altered the outcome?

It was assumed that each of the models were equally likely, but no likelihood estimation was made to estimate the posterior probability. It could be argued that the immigration models are more likely than the non-immigration models (see Chapter 3). However, if greater probability were placed on the immigration models, the results would be much the same. The immigration models produced the same best policy for all the criteria except "biomass diversity" and "catch diversity", when the extreme policies were included. The propagation of this greater certainty through time would merely make this result more substantial.

The calculation of EVPI involved comparing the results in year 20, when the system had stabilised, with the initial values of the performance criteria. This ignores the results during the transient years. It was argued earlier that it is the long term results which are of interest when comparing long term policies. This method has a precedent. Lave and Dowlatabadi (1993), for example, constructed decision tables to investigate the uncertainty over the effects of global warming and actions that should be taken using qualitative data from expert predictions. They did not run simulations, but used point estimates. Russ and Alcala (1992) conducted a practical experiment on the before and after effects of fishing in an area where

there had been protective management. Their results showed that there was a significant decline in the abundance of species after the reserve was subjected to fishing.

However, it would be wise to examine the transient effects before making any final policy decisions. For policy 4, for example, which bans the trawl and hunting gillnet, the biomass of the sergestids declines by several fold when top-down control is assumed. For a few years, this would mean that the mini-trawler and scissor net catches would be very low. To implement this policy, an additional cost might be necessary to compensate sectors dependent on the sergestids for their main catch.

Some workers have included both experimental and non-experimental policies in their comparison of EVPI (e.g., Sainsbury 1991). No deliberate experimental policies were included here, for it seemed more reasonable to first establish whether, for the performance criterias relevant to managing the fishery, the models did indeed make different predictions. They did not.

As a result of this analysis then, there is no increase in certainty about which model is the most likely. There is also, for current management purposes, no need to know.

When Would an Experimental Approach be Worthwhile?

The performance criteria used are mass scale criteria, meaning that they clump information, so that individual change is masked by the sum change. The performance criteria reflect the current management interests in the fishery. It is suggested that a more focused management objective would produce a different set of results. If the aim, for example, was to increase the biomass of a certain species, or to increase the catch of a particular gear type, an experimental approach might prove worthwhile.

The decision tables may also be interpreted differently. There are two questions asked in the decision tables. One is whether the major force governing energy flow is bottom-up or top-down and the second is whether fish immigration plays a significant role in San Miguel Bay. Without the immigration models, consistent differences would exist between the bottom-up and top-down predictions. The EVPIs for differentiating only between the bottom-up and top-down assumptions are higher for “total biomass”, “total catch”, “catch diversity” and revenue per small-scale gear” than the EVPIs for all four models. This would re-open the question of whether it be worthwhile to use an experimental approach to distinguish between the two models. If there is immigration of fish and crustacea into San Miguel Bay, there is no consistent difference between the predictions for the top-down and bottom-up scenarios. The primary question and uncertainty over resource dynamics in San Miguel Bay may be more accurately defined around the question of immigration. If the immigration is as high as it is modelled, the issue of the nature of tropho-dynamic control becomes a non-issue.

Experimenting with Fish and Crustacea Immigration

Most of the performance criteria used above were unable to distinguish the immigration models from the non-immigration models on a consistent basis. Further simulation experiments indicate that no policy which deliberately decreases effort can discriminate between the immigration and non-immigration models. The results of the immigration models were always very similar to the results of the model 1. In order to effect significantly different predictions from all four models, fishing effort has to be increased. Furthermore, the increase in effort has to be fairly substantial. Two experimental policies were developed to probe the system and produce a feedback response that would provide information on the role of immigration, and possibly, tropho-dynamic control.

The first experimental policy is policy 10, an extreme policy taken from the list of policies above. Effort throughout the entire fishery is increased by 33% for five years. This policy produced varying degrees of increase and decrease in the biomass of the large crustaceans and the juvenile sciaenids. These relative differences can be used to discriminate between the models. However, increasing the whole fishery on a trial basis for 5 years is problematic, particularly if the long term management decision was to reduce effort in the fishery.

An alternative policy is to increase the fishing effort of the hunting gillnet by 100% for 5 years. This policy produces similar results to the above policy, but would be a less expensive

and more flexible experimental option⁹⁶.

The effect of the different model predictions on the biomass of the large crustaceans and the juvenile sciaenids are illustrated in Figure 5.1. The total fished biomass is not predicted to decrease from the 1992-1994 value for any of the models. The biomass of the large crustaceans increases for all models and the biomass of the juvenile sciaenids decreases for all models. The models are discriminated on the relative levels of biomass increase and decrease. For model 1, no immigration and bottom-up control, the biomass of the large crustaceans doubled and the biomass of the juvenile sciaenids decreased to about 10% of the biomass before the experiment. For top-down control, model 2, the biomass of the large crustaceans increased to five times the initial biomass and the juvenile sciaenids biomass decreased to a very low level. When there is immigration and bottom-up control, model 3, the biomass of the large crustaceans increased by around 50%, while the biomass of the juvenile sciaenids only decreased by about 40%. For model 4, immigration and top-down control, the biomass of large crustaceans increased three fold and the juvenile sciaenids decreased by 60% . Thus there are clear predictions for the four models after 5 years of the experimental policy. These biomass changes are summarised in Table 5.15.

The time transients for each model over the five experimental years indicate that it is likely

⁹⁶ A 100% increase in the hunting gillnet effort would entail an extra 288 boats and nets. Increasing the whole fishery by 33% would mean an extra 378 boats and nets, just for the trawl, mini-trawler, ordinary gillnet and hunting gillnet gears.

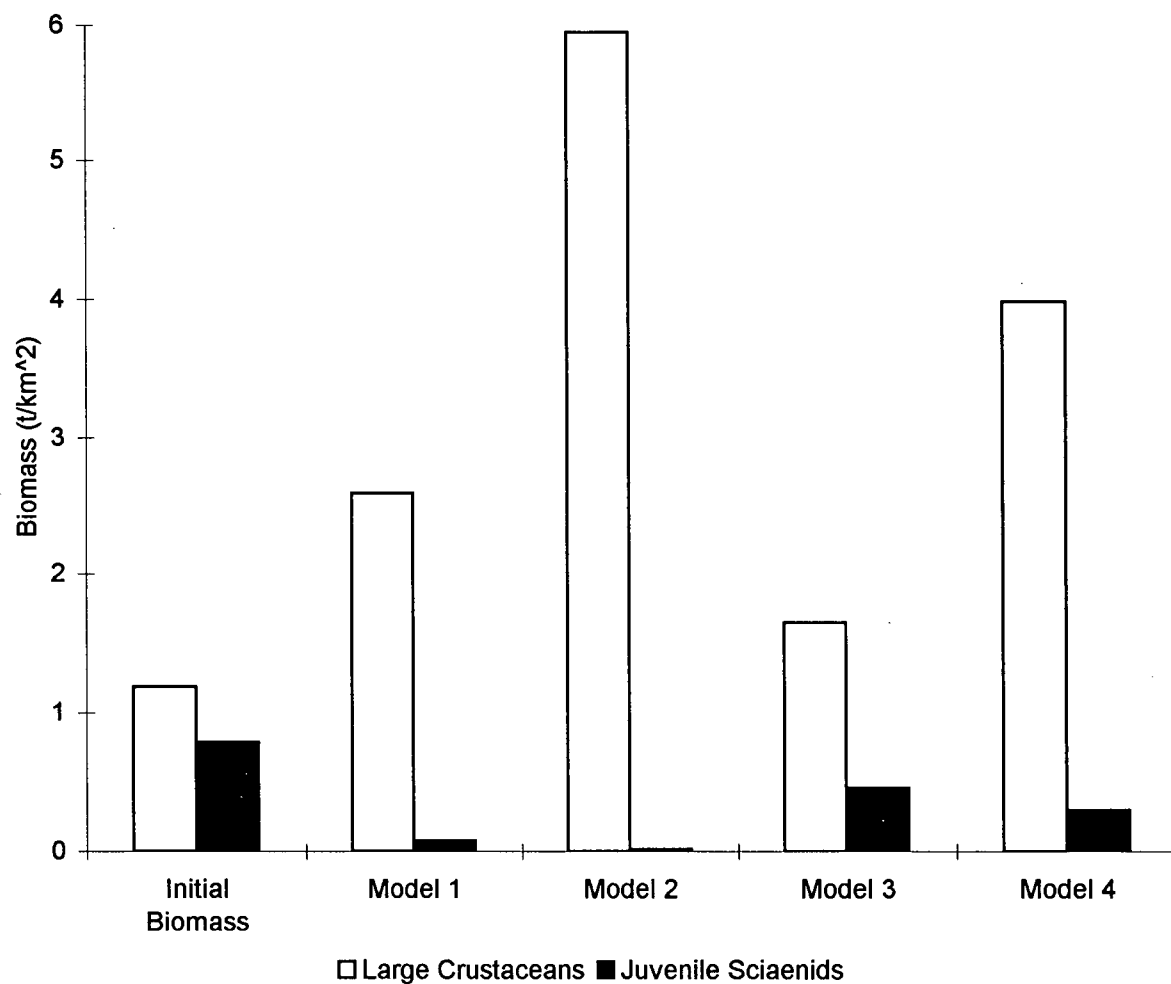


Figure 5.1. Results of an experimental policy where the hunting gillnet is increased by 100% for 5 years. Shown are the changes in the biomass of two indicator species, the large crustaceans and the juvenile sciaenids. See text for further details.

Table 5.15 Table summarising the changes in biomass of the large crustaceans and juveniles sciaenids after an experimental policy lasting 5 years.

| <i>Performance Criteria</i> | <i>Model 1</i> | <i>Model 2</i> | <i>Model 3</i> | <i>Model 4</i> |
|-----------------------------|--------------------------|----------------------------------|----------------------------|--------------------------|
| Large Crustaceans | biomass doubles | biomass increases 5 fold | biomass increases 1.5 fold | biomass increases 3 fold |
| Juvenile Sciaenids | biomass decreases by 90% | biomass decreases almost to zero | biomass decreases by 40% | biomass decreases by 60% |

that by year 3 of the experimental period, it would be possible to use feedback information to begin to discriminate between the models. Estimating learning rates and the duration of the experimental phase is critical to adaptive management (Walters and Hilborn 1976). Overly optimistic predictions can lead to an expensive experimental situation with no gain, whereas pessimistic predictions may lead to no experiment and no gain. In this experiment, the large crustaceans and the juvenile sciaenids are not the only species which can be used to indicate change. The feedback information on biomass changes of some of the other species, for example, the penaeids would be useful and may hasten the learning process. If the model predictions are correct, there would be no loss in overall yield during this experiment.

This experimental policy could discriminate between the four models. During the course of the experiment, feedback information would be used to update the model, its parameters and predictions. This is a simplified description of events. It would be necessary to conduct extensive sensitivity studies into the different assumptions of the model, for example, less extreme or more extreme top-down assumptions, different assumptions about the immigration rates. Here, four simple models of the extremes have been used for illustrative purposes. It would also be necessary to consider whether the experiments should be replicated in space or time (McAllister and Peterman 1992, Hilborn and Walters 1992, Walters 1986). This prolongs the experimental phase and thus the expense, but provides for an experiment more robust against spatial and temporal variability.

Validating the ECOSIM model

At a different level, active adaptive management could be used to validate the ECOSIM model and its parameterisation for San Miguel Bay. The form of multispecies analysis used, although comparatively undemanding of data, nonetheless requires many parameters⁹⁷. Many of these values were not empirically derived from San Miguel Bay. Although it was possible to construct ECOPATH and ECOSIM models, there is still uncertainty surrounding some of their parameter values. An active experimental approach which aimed to update these parameter estimates, including the diet composition, would help to decrease uncertainty over parameters and make the model a more useful tool for management of San Miguel Bay.

One of the extreme policies, for example, predicts an abundant fishery based on crustaceans. The ECOSIM predictions, if correct, imply that a sustainable fishery which would employ 33% more people than are currently employed⁹⁸. A fishery with greater catch, profits and employment is a tempting prediction. The policy would entail the loss of biodiversity, but a utilitarian would point to the fact that San Miguel Bay is already well overfished, perhaps irrevocably so. This would be an opportunity to improve the welfare of the fishing communities of San Miguel Bay, albeit at the expense of the resource. It is one fishery amongst many in the Philippines. Arguably a harsh experiment, to test whether this kind of increase was likely would benefit the fishers of San Miguel Bay. The results of such an experiment would also be useful for the management of other fisheries in the Philippines, and globally.

⁹⁷ The derivation of these parameters was described in chapters 3 and 4.

The Reality of Adaptive Management

From a scientific point of view, the prospect of using experimental management to resolve uncertainty is enticing. However, the challenges are very real and as a consequence, there have been few practical applications of adaptive management (Bundy, *In Press*, McAllister and Peterman 1992). The best known practical application of adaptive management is that of Sainsbury (1989, 1991) from the Northwest shelf of Australia. Others include Johnson *et al.* (1992) who used an adaptive management framework to evaluate the effect of a rehabilitation plan on the yellow perch fishery in Lake Michigan; Ward and Jacoby (1992) reported on their preliminary success using an adaptive management approach in Jervis Bay, Australia; Alcala and Russ (1990) tested the effect of protective management in the Philippines and Milliman *et al.* (1987) proposed adaptive management as a means to evaluate the success of rehabilitation of the lake trout fishery in the Laurentian Great Lakes and to make policy choices. Only the work of Alcala and Russ occurred in a developing country.

There are several, interrelated reasons why adaptive management has not been more widely adopted. The first is that although in concept, adaptive management is quite straightforward, in detail it is usually complex and involves sophisticated numerical modeling. Such modeling is not readily accessible to many. The second problem with adaptive management is that it requires cooperation from those involved in the fishery. This is not always easy to obtain. In part this is due to a lack of understanding or disbelief in the numerical game playing. At a more fundamental level problems arise when adaptive management policies involve reduction

⁹⁸ One would think that the stagnation and decline of the large-scale sector since the 1980s tends to contradict this prediction however .

in fishing effort: fishers do not want to be in the experimental area where fishing intensity is low (McAllister and Peterman 1992). Experimental design may circumvent this problem, or extensive discussion may be required. In either case, additional time and economic resources are required.

Associated with the above are the economic and social costs of adaptive management.

Adaptive management costs are high. However, in addition to the scientific costs there is also the very real issue of the costs to displaced fishers and their families. These social and economic costs can be extremely high, and far outweigh the scientific costs of the experiment. They are included in the "expected value" as, for example, the value of lost catches, or unemployment costs. However, the calculation of such may not be readily apparent to the fisher or manager.

Essentially, it is better that the main conclusion from this analysis is that experimental management it is not required in San Miguel Bay for more effective management. The challenges facing the implementation of adaptive management in fisheries of developing countries are likely to be greater than have just been described. The funds to meet the costs of such experiments may simply not be available. Small-scale fisheries of developing countries involve many more fishers than are commonly found in the fisheries of developed countries. There are therefore many more people to both compensate and to organise and cooperate with. Furthermore, many fishers do not only rely on fishing for their livelihood but they rely on it for their existence. In the Philippines, for example, there is no Unemployment Insurance or other such welfare safety nets. The risks of experimentation are thus great. In addition, the

widespread nature of landing sites challenges the ability to monitor effort or landings effectively. Even with widespread agreement by fishers to participate in adaptive management schemes, these agreements are not always maintained for long (C. Walters pers. comm.). Voiced cooperation does not always meet the real needs of controlled adaptive management.

Adaptive Management in San Miguel Bay

The spirit of co-management in San Miguel Bay, the deputisation of citizens as wardens and the general awareness of sustainability issues will hopefully be sustained. This is pertinent because the conclusion that experimental management is not immediately useful in San Miguel Bay, does not mean that adaptive management is not useful. Although adaptive management has commonly been equated with active or experimental adaptive management (J. Sainsbury, pers. comm.), it is, at its core, a management system which responds to the feedback. Feedback that results from management policies, active or passive. A passive policy also produces information that is pertinent and important to resource management. A passive policy will still perform, or not, according to model predictions. The key to adaptive management is the feedback response. The difference between passive adaptive management and “normal” management is that in the former, the system is actively monitored and the response used to adapt theories and policy formation. Adaptive management should not be forsaken.

The results showed that the current management policy of banning and phasing out trawling is not the optimal management strategy for San Miguel Bay. However, the implementation of this policy should be monitored. If there is a strong determination by the fishers of San Miguel Bay and the SMBMC to ban trawling completely, this is an excellent opportunity to monitor the response of the system to this change in fishing pattern. To date there has been no annual catch and effort statistics, so a monitoring program would have to be organised. Monitoring could be done by volunteers from the fisher's organisations or by members of the Barangay task force component of the SMBMC. Since there is strong cooperative spirit in San Miguel Bay, it may be possible to take advantage of this, the proposed management changes and to substantially further knowledge and understanding of the resource.

Chapter 6

Summary and Concluding Comments

“The biological interactions between species (competition, predator-prey), including the stability of the community structure and the operational interactions between different groups of vessels targeting a particular species, need to be assessed and taken into account.....” Gulland and Garcia 1984:155

The assessment and management of multispecies, multigear gear fisheries presents challenges in temperate fisheries, where data are relatively good and the number of species and fishing gears relatively small. The situation is vastly more challenging for tropical multispecies, multigear fisheries where data are relatively poor and the number of species and fishing gears relatively large. Furthermore, there has been relatively little research into tropical fisheries compared to the research body that has focused on temperate fisheries. Most of the research carried out for either type of fishery has been performed on a single species basis. The overall objective of this thesis was to use a ecological theory to inform fisheries assessment and management. The following aims directed this research: to study species interactions in a multispecies fishery; to examine the biological and ecological impacts of a multigear fishery on a multispecies resource; to determine the effects of ecosystem considerations on fisheries assessment and management; to develop a systematic and integrated approach to the assessment of multispecies, multigear fisheries using relatively simple methods with wide applicability; to develop tractable sustainable management strategies for multispecies, multigear fisheries, and to focus on developing countries, where fisheries are frequently multispecies, multigear and data sparse.

These aims were addressed using a case study from the Philippines, San Miguel Bay, a fishery typical of many multispecies, multigear fisheries in the tropics. In addition to the aims above, it was noted in Chapter 1 that the research presented in this thesis could be interpreted simply as a rigorous assessment of a fishery. For indeed, at its core, that is what this work is. But it is also meant to be more than that. It demonstrates the types of analyses that can be conducted for a fishery for which few of the methods commonly used in fisheries assessment are applicable. It demonstrates, that multispecies methods such as ECOPATH and ECOSIM can be usefully applied, even if empirical data do not exist for every required parameter. Most fundamentally, it demonstrates how, when the ecological interactions that occur between species and the interactions that occur between the fishery and the ecosystem are examined, a much fuller and greater understanding of the fishery is gained. Such an understanding is vital to sustainable management. The scope for analysis, using these this suite of methods, is vast.

Multispecies assessments can take several forms and at their simplest, they consist of a descriptive interpretation of species changes with time. In Chapter 2, a study of the changes in species composition, from trawl survey data dating back to 1947, demonstrated that the species changes that have occurred in San Miguel Bay are comparable to the changes that occurred in the Gulf of Thailand (Pauly 1979a). These types of changes, plus the decrease in total biomass as species incompletely replace one another, are indications that the fishery is suffering from ecosystem overfishing (Pauly 1994). Generalists such as the crab, *Portunus pelagicus* and the hairtail, *Trichiurus haumela* have increased, while small prey fish such as the leiognathids, and rays and medium predators have decreased. Most of these changes had

occurred by the time of the 1979-1982 study, although the biomass of *P. pelagicus* and *T. haumela* have since increased.

The fishery was also assessed using analytical and descriptive methods. The aim of Chapter 2 was to gain a thorough understanding of the fishery before embarking on the multispecies analyses and assessment in Chapters 3, 4 and 5. The overwhelming message that resulted from the analyses is that the fishery is overexploited. The total annual catch decreased by around 3,000 t since 1979-1982. The number of large-scale trawlers decreased by 50% while effort by the small-scale sector had both increased and diversified. However, the catch and CPUE of most gears decreased. Exceptions include the crab gillnet, which targets *P. pelagicus* and the shark gillnet. The small-scale sector now takes over 80% of the total catch in comparison to 64% in 1979-1982. The gillnets are the dominant sector in the fishery, landing 42% of the total catch. There are a number of different types of gillnet in the fishery, and between them, they catch a large percentage of the species present in the Bay.

Nine families and “trash” fish account for 87% of the total catch (Table 2.17). Using a variety of methods such as length-based assessment of mortality, length-based yield-per-recruit analysis, longitudinal comparison of abundance from trawl survey data, catch data and CPUE data, the status of these families was assessed. Of the nine families, five are overexploited on all counts (sciaenids, sergestids, leiognathids, engraulids and mugilids). The penaeids, trichiurids and carangids are stable, meaning that their catch and CPUE have not decreased since 1979-1982. However, the current rate of exploitation of the trichiurids is

greater than the optimal rate (Figure 2.4). The crab, *P. pelagicus* is abundant. Thus the only growth area in the fishery appears to be for crabs.

In Chapter 3, the first fully mass-balanced and parameterised ECOPATH model (Christensen and Pauly 1992a) of San Miguel Bay was developed⁹⁸. ECOPATH is a means to collate data and to enable a greater understanding of an ecosystem. Using a range of maturity indices, it was determined that the San Miguel Bay ecosystem is relatively mature, and thus likely to be resistant to stress. It was compared to the results of ECOPATH models of 41 other fisheries studied by Christensen (1995, 1994) and Christensen and Pauly (1993c). Of the total flow in the ecosystem, 56% originates in the detritus, a sign of a mature system. The fishery operates at an average trophic level of 4, although most species occur at around trophic level 3. The occurrence of such a large number of species at this level indicates that there is likely to be strong competition between species for resources. The results of the trophic impact routine indicate that both bottom-up, donor control (Hall *et al.* 1970, Hunter and Price 1992) and top-down control with trophic cascades (Carpenter and Kitchell 1993, Carpenter *et al.* 1985) occur in the fishery. Since the top predator biomass has been severely reduced, it is not surprising that donor control is observed.

This was also the first ECOPATH model in which the fishery was directly modelled as “predators” of the ecosystem. This process enables the study of the impact of particular fishing gears on the ecosystem, not just on the single species that they target. The small-scale sector collectively fishes at a lower trophic level than the large-scale sector, although the

⁹⁸ See also Chapter 3, note 2

trophic level at which individual small-scale gears fish is variable. The fine meshed gears and the mini-trawler fish at the lowest trophic levels, and the ordinary gillnet and hunting gillnet fish at the highest trophic levels (Table 3.10). The results of the trophic impact routine indicated that the small-scale fishery has a stronger and wider impact on the eco-groups than the large-scale sector. It was also evident from the results that complex interactions occur between eco-groups when there are changes in fishing pattern. An increase in fishing effort by a particular gear, for example, the mini-trawler, does not necessarily cause a decrease in all eco-groups that are caught by this gear. This complexity is caused by the combined effects of fishing mortality, predation mortality and competitive mortality and make prediction a difficult task. These results demonstrate the relevance and importance of studying fisheries from both a multispecies and multigear perspective.

In Chapter 4, it was finally possible to ask “what if” questions, such as “what happens if the large-scale sector is completely banned?”, the question at the forefront of much of current management effort in San Miguel Bay. These questions were addressed using a new dynamic multispecies model, ECOSIM (Walters *et al.* 1997). ECOSIM uses the output from mass-balance assessments as input parameters. In a simple way, the static mass-balance approach is thus transformed into a fully dynamic and flexible multispecies fisheries tool. It was used here to examine species interactions and to make predictions about how the ecosystem responds to perturbations, such as changes in fishing mortality on one species or changes in fishing patterns of fishing gears. It is an ideal tool for examining multispecies, multigear interactions.

The ECOSIM analysis graphically confirms the overfished state of San Miguel Bay. Many eco-groups are well below the 30-50% of the projected unexploited biomass. The extent of over fishing of individual pools was dependent on the control assumption. For bottom-up control, most pools were optimally fished and the crustacean pools were sub-optimally fished. For top-down control, virtually all pools were overfished, except the crustacean pools, which were optimally fished. When the effects of fishing by the multigear fishery were included in the equilibrium yield curve analysis (Table 4.2), as in reality, there was some consistency between the bottom-up and top-down results. Some eco-groups are overfished, some are not, but they are generally the same groups for either assumption. It is suggested that this is because, for most pools in this model, the effects of fishing mortality are stronger than the difference between bottom-up and top-down control hypotheses.

A cautious approach to management would be to assume that top-down control is the dominant force controlling energy flow in San Miguel Bay, although there was no basis, from these results, to make such an assumption. Bottom-up control presents a more optimistic scenario. Alternatively, a non cautious approach would increase fishing effort across the fishery. For either bottom-up or top-down control, a productive fishery, based on the crustacean groups resulted if total fishing effort was increased.

The ECOSIM results amply demonstrate that, if predator/prey interactions are strong in a multispecies ecosystem (top-down control), a biomass change in one part of the ecosystem can have significant effect on other parts of the ecosystem. This was evident, when the equilibrium yield curves were examined. A decrease in the biomass of the sciaenids, for

example, caused increases in the biomass of its prey, and decreases in other eco-groups. This was a straightforward interaction. However, many of the other interactions are complex. Walters *et al.* (1997) likened these interactions to a web of effects that make it difficult to make simple and clear predictions. For the bottom-up assumption, which describes a stable and resilient ecosystem, there was less change in the biomass of other eco-groups. It presents a simpler picture.

The combined effects of fishing mortality, predation and competition make prediction difficult, as noted for the ECOPATH results. With ECOSIM, however, the fishing pattern can be directly manipulated and time simulations run to determine the impact on the ecosystem of different fishing strategies. Four conclusions were drawn from these simulations. The first was that the biomass of pools can be affected by a change in fishing effort of a particular gear, even if they are not caught by that gear. This is a consequence of trophic interactions among species. The second conclusion was that the results are very sensitive to the assumptions made about tropho-dynamic control, that is, top-down or bottom-up. In some cases, bottom-up and top-down control predicted opposite responses of pools to a changes in effort (Figure 4.11). More generally, top-down control had a greater and a wider impact on individual pools than bottom-up control. The third was that the impact a fishing gear has on the ecosystem depends not only on the size of its catch, but also on the composition of its catch. The fourth conclusion was that, if fishery biomass is to be increased, it will entail a greater reduction in fishing effort than just banning trawlers.

The basic message resulting from the multispecies, multigear analyses in the preceding chapters is that a better understanding of ecosystem dynamics and the interaction of the fishery with the ecosystem is critical to improve management. This was examined in Chapter 5. Two major uncertainties were identified. The first was the nature of tropho-dynamic control, as discussed in Chapter 4. The second was the role of immigration of fish in the fishery. Immigration was discussed in Chapter 3, where it was argued that for the stationary mass-balance model of San Miguel Bay, immigration was not an issue. However, for the dynamic model, the role of immigration in the dynamics of the fishery requires further investigation. Adaptive management was used to determine whether these uncertainties were as important to management decisions as the results in preceding chapters indicated.

In essence, the question was whether there was any value in learning more about these uncertainties by using an experimental approach to management. This was tested by running a series of simulations for each hypothesis, under a range of management policies using ECOSIM. Policies included the current management policy of banning all trawling, policies which involved greater effort reduction and policies that involved increasing effort to exploit the crustaceans. The results for each policy and each model were compared in decisions tables for six criteria (Tables 5.3 - 5.14). The surprising result from this analysis, for all the criteria, was that there was a clear best policy, regardless of the resource model. That is, there were clear best non-adaptive policies for each criteria. This means that the differences between the four models of the resource have less impact on ecosystem dynamics than the fishing mortality caused by the best non-adaptive policies.

It was concluded that, for current management purposes, experimental adaptive management was not required in San Miguel Bay. However, it was recommended that a passive adaptive management approach should be pursued. The current management plan in San Miguel Bay includes a ban on all trawling. This is an excellent opportunity to observe the effect on a multispecies fishery of a large perturbation. The effect of this ban on catch levels in the fishery should be carefully monitored. The data gathered from the monitoring exercise can be fed back into the ECOSIM models to determine their utility, to update parameters, to increase knowledge of the resource and hence improve future management.

Implications for the Assessment and Management of Multispecies,

Multigear Fisheries

A hasty conclusion that might be drawn from this research is that it is not worthwhile going to the trouble of doing multispecies analyses. The results suggest that the uncertainty over the nature of tropho-dynamic control, whether there are strong or weak species interactions and whether there is immigration do not effect the long term management options for the fishery. However, these results are specific to the San Miguel Bay fishery and to the criteria used to assess the management policies. They may not be applicable to other fisheries.

Comparative studies should be conducted in order to examine this result further. Christensen (1996) concluded, on the basis of a comparative study of ecosystems from coastal, freshwater and upwelling systems, that multispecies models may give better long term advice

than single species models. Several situations were discussed in Chapter 5 where it could be worthwhile to discriminate between resource hypotheses.

Specifically, the analysis in Chapter 5 was confounded by the immigration question. If the estimation of Expected Value of Perfect Information (EVPI) had included only the uncertainty about tropho-dynamic control, a different conclusion would have been drawn. It is clear from the results in Chapter 4 that species interactions can have profound impact on the effects of fishing on the ecosystem. In the absence of uncertainty over immigration, these impacts would have been considered germane to current management objectives.

This research was primarily concerned with the importance of multispecies and multigear interactions in multispecies, multigear, fisheries, and their implications for management. One of the neat things that ECOSIM modelling enables is the study of the effect of individual, or groups of fishing gears on the ecosystem. In San Miguel Bay for example, the gillnets are a traditional and widespread gear. The ordinary gillnet is particularly prevalent. The analysis in Chapter 4 indicates that the impact of the ordinary gillnet and top-down control on the total fishery biomass is equivalent to that of the large-scale trawlers (Table 4.4). Its impact on individual pools is greater than the large-scale sector when top-down control is assumed (Figure 4.11). Yet management in San Miguel Bay is centred on banning trawling. It has not considered the impact of the massive gillnet sector. There are many reasons for this, including socio-economic concerns, but an additional and very pertinent reason is that there was not previously the means to examine the impact of one sector on the resource. In other words, ECOPATH and ECOSIM modelling enables the fishery to be examined, sector by

sector, species by species. This is a revolutionary development for fisheries where previously it was only possible to make the most basic of single species assessments.

However, no clear optimal management policy resulted from this analysis. This reflects the complexity of the issues confronting the fishery. It also reflects the fact that there is no "rule of thumb" whereby a suite of management strategies can be designed to cover all situations. For different fisheries there are different management objectives (Smith 1983). A multispecies analysis such as described here enables a thorough and complete investigation into the consequences of different management measures. In San Miguel Bay, for example, two groups, the sciaenids and the medium predators have strong repressive effects on other groups in the model, for top-down control. The same repressive effects were seen when the biomass of the sciaenids and medium predators was altered by changing the fishing pattern of the gears which target them. If a management goal was to increase the biomass of the prey of the sciaenids and the medium predators, then effort on the latter two species should be increased. By gaining an understanding of the community dynamics in the ecosystem, an understanding is gained of the impact of fishing on the ecosystem.

In the future, there may be more specific management goals in San Miguel Bay, for example, a particular ecosystem configuration may be desired (Murawski 1991). Such goals may already exist in other fisheries. Multispecies modelling, such as used here, would enable insight into how these goals could be achieved. In developing fisheries, for example, including multispecies, multigear analysis in the adaptive management approach suggested

by Hilborn and Sibert (1988), may prove invaluable to the successful management of these increasingly rare fisheries.

Adaptive management is an important component of the methods discussed. It is an attractive approach because it is specifically designed to increase knowledge by using feedback information from the impact of management policies on the resource. In experimental management, these policies are deliberate probing policies designed to evoke a specific response from the resource (Walters 1986). Since in many tropical multispecies fisheries, there are limited fisheries data, this could be an appropriate way to learn more about the resource.

It would be exciting to build a series of ECOSIM models of tropical multispecies fisheries and to determine the importance of species interactions. This might be possible in the Philippines for example, where there is a series of ongoing multi-disciplinary studies which are part of a larger Fisheries Sector Program (ICLARM 1995). If a series of ECOSIM models were developed for other fisheries in the Philippines, for example, the Lingayen Gulf (see Silvestre *et al.* 1989), the Lagonoy Gulf and Manila Bay, an experimental situation could be conceived whereby these fisheries would represent a series of replicates for experimentation. If well organised, such an experiment could furnish considerable knowledge of species interactions and fishery/ecosystem interactions.

The analyses in each chapter have been described with a clear orientation to the assessment and management of fisheries. However, at the same time, there are great scientific questions

being raised. The relative roles of bottom-up and top-down control have been debated and discussed in the literature for decades. The ECOPATH analysis gave some support for both forms of control and the ECOSIM analysis examined the impact of both assumptions. There is support for both and in all likelihood, both will be present, as suggested by Matson and Hunter (1992), for example. This question cannot be resolved here, but certainly, modelling with ECOSIM introduces great scope for experimental research into the general ecological question of tropho-dynamic control. Experimental management could also be used to test the predictions of ECOSIM, as discussed in Chapter 5.

One of the rationales for this research was to address the deficiency of multispecies fisheries research in tropical (and temperate) fisheries. Another was to develop a systematic, integrated approach to the assessment and management of multispecies, multigear fisheries, particularly using relatively simple methods with wide applicability. That is, methods that can be easily applied in multispecies fisheries in developing countries. Most of the software used for this research was produced by ICLARM, specifically for tropical fisheries, including length-based methods (see Pauly *et. al.* 1995). ECOSIM will also soon be available from ICLARM, as a routine of version 4.0 of ECOPATH⁹⁹.

With this suite of methods, it is possible to conduct multispecies analyses for many fisheries. The assumptions and limitations of these methods were discussed in the relevant chapters. Ideally, these results would be examined in more depth, for example, to examine the impact of model assumptions. However, in order to meet the objectives of this thesis, this was not

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feasible. Several methods were used for parameter estimation, where possible, in order to corroborate results. In addition, ECOPATH has now been complemented with a Monte Carlo routine whereby parameter uncertainty can be quantified (Christensen and Pauly 1995). The

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Appendix 1List of Species Present in San Miguel Bay (from Trawl and Landings Surveys)

OSTEICHTHYES

APOGONIDAE (Cardinal Fishes)

Apogon quadrifasciatus *Apogonichthys poecilopterus*
Apogon sp.

ACANTHURIDAE

Acanthurus xanthoptherus *Naso lituratus*

ARIIDAE (Catfish)

Arius leiotocephalus *Arius manilensis*
Arius spsetirostris *Arius sps.*

ATHERINIDAE (Siversides)

Atherina forskali

BELONIDAE (Needlefish)

Tylosorus strongylorus *Tylosorus giganticus*

BOTHIDAE (flounders)

Pseudorhombus cinnamoneus

CAESIONIDAE

Caesio caerulaureus *Caesio sps*
Caesio erthyrogaster

CARANGIDAE (Jacks)

| | |
|-----------------------------------|--|
| <i>Alectis ciliaris</i> | <i>Caranx malabaricus</i> |
| <i>Alepes djeddaba</i> | <i>Caranx melampygus</i> |
| <i>Alepes kalla</i> | <i>Caranx sexfasciatus</i> |
| <i>Alepes vari</i> | <i>Caranx tille</i> |
| <i>Atule mate</i> | <i>Caranx sp.</i> |
| <i>Carangoides ferdau</i> | <i>Decapterus macrosoma</i> |
| <i>Carangoides fulvoguttatus</i> | <i>Gnathodon speciosus</i> |
| <i>Carangoides leiotocephalus</i> | <i>Megalaspis cordyla (torpedo scad)</i> |
| <i>Carangoides sps.</i> | <i>Scomberoides lysan</i> |
| <i>Caranx armatus</i> | <i>Selar boops</i> |
| <i>Caranx ignobilis</i> | <i>Selaroides leptolepis</i> |
| <i>Caranx leptolepis</i> | |

CENTROPOMIDAE

Lates calcarifer *Psammoperca waigiensis*

CHAETODONTIDAE (reef fish)

Parachaetodon ocellatus (reef fish) *Chaetodon* sps.

CHANIDAE (Milkfishes)

Chanos chanos

CHICHLIDAE

Oreochromis mossambicus

CHIROCENTRIDAE (Wolf-herrings)

Chirocentrus dorab

CLUPEIDAE

Pellona ditchela *Sardinella fimbriata*
Sardinella perforata *Dussumieria acuta*

CONGRIDAE

Conger cinereus

CORYPHAENIDAE

Coryphaena hippurus

CYNOGLOSSIDAE (Sole)

Cynoglossus bilineatus *Cynoglossus gracilis*
Cynoglossus sp.

DOROSOMATIDAE

Anodontostoma selangkat

ECHENEIDAE (Shark suckers)

Echeneis naucrates

ENGRAULIDE (Anchovies)

Stolephorus commersonii *Stolephorus* sp.
Stolephorus indicus *Thryssa setirostris*

EPHIPPIDAE (Sickle Fish)

Drepane punctata *Platax orbicularis*

EXOCOETIDAE (Halfbeaks)

Hemiramphus georgii *Cypselurus poecilopterus*

FISTULARIIDAE

Fistularia petimba *Fistularia* sps.

FORMIONIDAE

Formio niger

GERREIDAE

Gerres filamentosus *Pentaprion longimanus*
Gerres oyena

GOBIIDAE

Glossogobius giurus *Istogobius spence*
Gobii sp. *Oxyurichthys microlepis*

HAEMULIDAE

Pomadasys hasta *Pomadasys* sp.
Pomadasys maculatus *Plectorynchus chaetodonoides*

LABRIDAE

Cheilinus trilobatus *Choerodon schoenleini*

LACTARIIDAE

Lactarius lactarius

LEIOGNATHIDAE (Pony Fishes)

Leiognathus bindus *Gazza minuta*
Leiognathus elongatus *Secutor insidiator*
Leiognathus equulus *Secutor ruconius*
Leiognathus splendens *Leiognathus* sp.

LETHRINIDAE (Emperors)

Lethrinus lentjan *Lethrinus ornatus*
Lethrinus miniatus *Lethrinus variegatus*
Lethrinus nebulosus

LUTJANIDAE (Snappers)

Lutjanus argentimaculatus *Lutjanus russelli*
Lutjanus bohar *Lutjanus sebae*
Lutjanus carponotatus *Lutjanus spilurus*
Lutjanus decussatus *Lutjanus* sp.
Lutjanus fulviflamma *Symphorus nematophorus*
Lutjanus rivulatus

MEGALOPIDAE (tarpons)

Megalops cyprinoides

MUGILIDAE (Mulletts)

Mugil cephalis *Liza subvirdis*
Mugil sp.

MULLIDAE (Goatfishes)

Mulliodichthys flavolineatus *Upeneus sulphureus*
Parupeneus sp. *Upeneus vitatus*

MURAENIDAE (Moray Eels)

Lycodontis javanicus (*Gymnothorax*
sp.)

MURAENESOCIDAE (Eels)

Muraenesox cinerius

NEMIPTERIDAE

Nemipterus japonicus *Scolopsis dubiosus*
Nemipterus metopias

OPHICHTIDAE

Ophictus sp.

PLATYCEPHALIDAE (Spiny flatheads)

Platycephalus indicus *Cymbacephalus nematophthalmus*

PLOTOSIDAE (Eeltail catfish)

Plotosus anguillaris

POLYNEMIDAE (Threadfins)

Polynemus microstoma *Eleutheronema tetradactylum*
Polynemus nigripinis

PRIACANTHIDAE (Big eyes)

Priacanthus sp.

PSETTODIDAE (Primitive Flatfishes)

Psettodes erumei

RACHYCENTRIDAE

Rachycentron canadum *Rachycentron* sp.

SCARIDAE

Scarus ghobban

SCATOPHAGIDAE (Scats)

Scatophagus argus

SCIAENIDAE (Croakers)

*Dendrophysa russelli**Pennahia macrophthalmus**Otolithes ruber**Pennahia* sp

SCOMBRIDAE

*Euthynnus affinis**Scomberomorus commerson**Rastrelliger brachysoma**Thunnus albacares**Rastrelliger kanagurta*

SERRANIDAE (Sea basses)

*Ephinephelus areolatus**Anyperodon leucogammicus**Ephinephelus fuscogattus**Cephalopholis pachycentron**Ephinephelus macrospilus**Cromileptis altiveles**Ephinephelus merra**Plectropomus maculatus**Ephinephelus sexfaciatus**Epinephalus* sp.

SIGANIDAE (Spinefoots)

*Siganus canaliculatus**Siganus javus**Siganus guttatus**Siganus virgatus*

SILLAGINIDAE (Sillagos)

Sillago sihama

SOLEIDAE

Solea sp.

SPARIDAE (Porgies)

Mylio berda = *Acanthopagrus**berda*

SPHYRAENIDAE (Barracudas)

*Sphyraena barracuda**Sphyraena jello**Sphyraena obtusata*

SYNODONTIDAE (Lizard Fishes)

*Saurida tumbil**Trachinocephalops myops*

TETRADONTIDAE (Puffers)

Lagocephalus sps

THERAPONIDAE (Zebra fishes)

*Therapon jarbua**Therapon quadrilineatus**Therapon puta*

TRIACANTHIDAE

Tripodichthys blochi

TRICHURIDAE (Hairtails or Scabbard Fish)

Trichiurus haumela

TRIDONTIDAE

Sphoerodon lunaris

CHONDRICHTHYES

CARCHARHINIDAE (Requiem Sharks)

Carcharinus melanopterus

DASYATIDAE (Stingrays)

*Dasyatis kuhlii**Dasyatis urnak*=*Himanturna Urnak**Dasyatis narinari*

MOBIULIDAE (Devil Rays)

Mobula diabolus

MYLIOBATIDAE (Eagle Rays)

Aetobatus narinari

RHINOBATIDAE (Guitarfishes)

Rhynchobatus djiddensis

RHINOPTERIDAE (Cownose Rays)

Rhinoptera sp.

SPHYRINIDAE (Hammerhead Sharks)

Sphyrna zygaena

INVERTEBRATES

BIVALVIA

Bivalves

OCTOPODA

Octopus sp.

PENAEIDAE

Penaeid sp.

SERGESTIDAE

Sergestid sp.

FAMILY?

Pandaka pygmea

PORTUNIDAE (True crabs)

*Portunus pelagicus**Scylla serrata*

STOMATOPODS (Mantis Shrimps)

Squilla sp.

TEUTHOIDEA (Squids)

Loligo sp.

Sepia sp.

MISCELLANEOUS

Assorted fish

Mixed of species

Trash fish

Fish Meal

Appendix 2

Restructuring the Ecopath Model to include adults and juveniles

In order to adapt the model to include the juveniles of these groups, the following parameters had to be re-calculated for both the adults and the juveniles:

P/B, Q/B, EE, export/harvests, biomass and diet. Since there was no explicit data available for the juveniles distinct from the adults, some assumptions had to be made on the basis of the available data.

Estimation of the proportion of juveniles in the catch

The proportion of juveniles in the catch was estimated from the length-frequency data in the landings survey and catch data (see Chapter 2). Fish below L_m (length at first maturity) in the survey were considered juveniles, those above, adults. Values of L_m were taken from the literature in FishBase (1995). Ecopath is a biomass model, so length was converted to weight (using the length weight relationships estimated in Cinco *et. al* 1995b) and the proportion of the juveniles in the catch expressed in terms of weight. Once the proportion of juveniles in the catch for each gear was calculated, the proportion of juveniles in the total catch was estimated from the annual catch per gear figures (Chapter 2).

The Sciaenids

The croaker, *O.ruber*, comprises 65% of the total abundance of sciaenids and was again used as the representative sciaenid species (due to lack of data for the other sciaenids, *Dendrophysa* spp and *Pennahia* spp). The catch of *O.ruber*, is composed almost entirely of juveniles (Figure 2.21). The modal length caught by many gears was less than either the calculated L_m of 14.3 cm (see Chapter 2) or the L_m of 22.1cm for this species (Almatar 1993). Most gears appear to take 100% juveniles, although the various gillnets are more selective. The resultant proportion of juveniles in the total sciaenid catch was 86%.

The Medium Predators

This pool is composed of 62% *Trichiurus* spp., 11% Synodontidae, 8% Psettodidae, 7% Sphryaenidae and 12% others. Using a L_m of 32.5cm for the *Trichiurus* spp, which is the mean of values given in Nakamura and Parin (1993) and Banerji and Krishan (1973), the proportion of juveniles in the total catch was calculated as 11%. The gears catching juveniles were the filter net and the mini-trawler, both small meshed gears. Unfortunately, the landings length-frequency data for the other medium predator groups was scant. However, the catch of the Syphryaenidae and "others" (*Arius* spp, *Formio* spp and *Muraenosox* spp) was composed entirely of juveniles. In order to estimate one juvenile catch figure for the medium predators, it was assumed that all groups other than *Trichiurus* spp. were caught as juveniles. For each gear type, the relative catches of *Trichiurus* spp. and the other groups were weighted by their relative abundance in the catch to produce the proportion of juvenile medium predators for

each gear. The proportion of juveniles in the total medium predator pool was estimated as 44%.

The Large Predators

There is no length frequency data for the large predators from the landings survey. An estimate of the proportion of juveniles in the catch was made using the *O. ruber* data as a guide, since the large predators catch is also likely to be dominated by juveniles. Only gillnets and crab gear were considered likely to catch adult large predators. The gillnets were given the same adult to juvenile proportion as *O. ruber*. The crab gear was initially given a 50% proportion of juveniles, but this was later increased to 70% when balancing the Ecopath model. The final proportion of juveniles in the large predators catch was estimated to be 78% juveniles, 22% adults.

Estimation of the proportion of juveniles in the biomass

The proportion of juveniles in the biomass of the three groups was estimated from the length-frequency data in the trawl survey. As above, fish below L_m were classified as juveniles and those above as adults and the proportion expressed in terms of weight. The relative proportions are given in Table A2.1. Note that the data for the large predators is estimated on the basis of the *O. ruber* data, as described above.

Recalculation of P/B and Q/B values for adults and juveniles

The values of P/B and Q/B originally estimated for the sciaenids were used for the juvenile sciaenids since the data from which they were estimated were largely juvenile data. Similarly, the P/B for medium predators, which was estimated from the mainly adult *Trichiurus* sps data, was used for the adult medium predators. The original Q/B for the medium predators was also used for the adults. There were insufficient data to estimate new parameters for the adult sciaenids or the juvenile medium predators. Accordingly, the adult medium predators P/B and Q/B values were used for the adult sciaenids. For the juvenile medium predators, values intermediate between adult medium predators and juvenile sciaenids were used - see Table A 2.1.

The P/B and Q/B values of the large predators were not estimated for the earlier Ecopath model because of insufficient data and instead, literature values were used. These values were assumed to be for the adults. However, the Q/B was raised to allow for fishing mortality (see Chapter 3). However, since most fishing mortality in San Miguel Bay is inflicted on the juveniles, the lower Q/B value was used for the adults, and the higher value for the juveniles. The Juvenile P/B of the medium predators was also used for the juvenile large predators.

Recalculation of EE values for adults and juveniles

As before the default value of 0.95 was used for all groups other than the adult large predators where 0.5 was used.

Table A 2.1 New adult and juvenile parameters for Ecopath model

| <i>ECOSIM Group</i> | <i>Adult/ Juvenile</i> | <i>Biomass Proportion</i> | <i>Catch Proportion n</i> | <i>P/B (year⁻¹)</i> | <i>Q/B (year⁻¹)</i> | <i>EE</i> |
|---------------------|----------------------------|-------------------------------|-----------------------------------|------------------------------------|------------------------------------|-----------|
| Sciaenids | Juveniles | 97 | 86 | 4.39 | 16.0 | 0.95 |
| | Adults | 3 | 14 | 2.20 | 7.6 | 0.95 |
| Medium Predators | Juveniles | 52 | 44 | 3.30 | 11.8 | 0.95 |
| | Adults | 48 | 56 | 2.20 | 7.6 | 0.95 |
| Large Predators | Juveniles | 97 | 78 | 3.30 | 11.9 | 0.95 |
| | Adults | 3 | 22 | 2.00 | 7.3 | 0.50 |

Adult and juvenile diet composition

The diets described in Chapter 3 were split into adult and juvenile components. The juvenile diets reflect a diet based on small organisms, and the adult diets place a greater emphasis on organisms at trophic levels 3 and above.

The Sciaenids

The sciaenid diet was based on stomach studies from San Miguel Bay (Palomares *et. al* 1995b). Since the study appears to be based on juvenile fish, it was used as the basis for the

juvenile diet. However, the consumption of fish (engraulids and sciaenids) by the juveniles sciaenids is not likely was removed from the diet. Instead this fraction was ascribed to zooplankton, Table A2.2 The adult diet is contrived on the basis of literature reports (In FishBase (1995)) which indicate that sciaenids eat mainly crustacea and fish. The adults diet then shares the consumption of the crustacean groups Sergestidae, Penaeidae and Large Crustacea, with the juveniles, although in different quantities. In addition, they eat Engraulids and Pelagics and juvenile sciaenids.

Table A 2.2 Diet Composition of the Sciaenids in the Original Model and Split into Adult and Juveniles Groups.

| <i>Prey</i> | <i>Original sciaenids</i> | <i>Juvenile sciaenids</i> | <i>Adult sciaenids</i> |
|---------------------------|-------------------------------|-------------------------------|----------------------------|
| Zooplankton | 0.1 | 0.15 | 0 |
| Meiobenthos | 0.05 | 0.05 | 0 |
| Macrobenthos | 0.3 | 0.3 | 0 |
| Sergestids | 0.25 | 0.25 | 0.1 |
| Penaeids | 0.15 | 0.15 | 0.3 |
| Large Crustaceans. | 0.1 | 0.1 | 0.3 |
| Juvenile Sciaenids | – | 0 | 0.1 |
| Juvenile Medium Predators | – | 0 | 0 |
| Juvenile Large Predators | – | 0 | 0 |
| Demersal Feeders | 0 | 0 | 0 |
| Leiognathids | 0 | 0 | 0 |
| Engraulids | 0.03 | 0 | 0.1 |
| Pelagics | 0 | 0 | 0.1 |
| Sciaenids | 0.02 | 0 | 0 |

The Medium Predators

The original medium predators diet was based on a composite study of information from the literature. This contained little or no indication as to whether it pertained to adults or to juveniles. Information on the diet of *Trichiurus* spp. (Nakamura and Parin 1993) indicated that

juveniles eat plankton and small fish, whereas the adult diet includes anchovies, pelagics, sciaenids, *Trichiurus* spp., squid and crustacea. This suggests that the adults are fairly indiscriminate in their predatory activity. The juvenile diet was estimated using the theory that they will feed at the lower end of the food web. Giving credence to the report that they eat small fish, engraulids were included in their diet in Table A 2.3 below. The adult diet was based on the original diet and diet information for Psettodidae (Devadoss *et. al* 1977) which comprised 8% of the trawl biomass. Excluded from the diet were zooplankton, macrobenthos and Sergestidae: included were the large crustacea and the juveniles of the sciaenids, medium predators and large predators.

Table A 2.3 Diet Composition of the Sciaenids in the Original Model and Split into Adult and Juveniles Groups

| <i>Prey</i> | <i>Original Medium Predators</i> | <i>Juvenile Medium Predators</i> | <i>Adult Medium Predators</i> |
|---------------------------|--|--|---------------------------------------|
| Zooplankton | 0.09 | 0.40 | 0 |
| Meiobenthos | 0 | 0 | 0 |
| Macrobenthos | 0.14 | 0.25 | 0 |
| Sergestids | 0.15 | 0.20 | 0 |
| Penaeids | 0.09 | 0.10 | 0.09 |
| Large Crustaceans. | 0 | 0 | 0.04 |
| Juvenile Sciaenids | — | 0 | 0.08 |
| Juvenile Medium Predators | — | 0 | 0.05 |
| Juvenile Large Predators | — | 0 | 0.01 |
| Demersal Feeders | 0.10 | 0 | 0.24 |
| Leiognathids | 0.18 | 0 | 0.22 |
| Engraulids | 0.10 | 0.05 | 0.10 |
| Pelagics | 0.10 | 0 | 0.10 |
| Sciaenids | 0.025 | 0 | 0.02 |
| Medium Predators | 0.025 | 0 | 0.05 |

The Large Predators

The large predators diet was adapted from diet information from Ecopath models for the Gulf of Thailand and Brunei (V. Christensen *pers. comm.*, Silvestre *et al.* 1993). Once more there is no discrimination between adults and juveniles. As for the medium predators, a diet focused on the lower levels of the food chain was estimated for the juvenile large predators. The adult diet is based on the original, with a greater emphasis on a fish based diet, Table A 2.4. It includes all fish groups lower at lower trophic levels in the ecosystem.

Table A 2.4 Diet Composition of the Large Predators in the Original Model and Split into Adult and Juveniles Groups.

| <i>Prey</i> | <i>Original Large Predators</i> | <i>Juvenile Large Predators</i> | <i>Adult Large Predators</i> |
|---------------------------|---|---|--------------------------------------|
| Zooplankton | 0 | 0.5 | 0 |
| Meiobenthos | 0 | 0 | 0 |
| Macrobenthos | 0.04 | 0.2 | 0 |
| Sergestids | 0 | 0.2 | 0 |
| Penaeids | 0 | 0 | 0 |
| Large Crustaceans. | 0 | 0.1 | 0 |
| Juvenile Sciaenids | – | 0 | 0.15 |
| Juvenile Medium Predators | – | 0 | 0.06 |
| Juvenile Large Predators | – | 0 | 0.05 |
| Demersal Feeders | 0.13 | 0 | 0.13 |
| Leiognathids | 0.14 | 0 | 0.14 |
| Engraulids | 0.16 | 0 | 0.16 |
| Pelagics | 0.20 | 0 | 0.20 |
| Sciaenids | 0.20 | 0 | 0.05 |
| Medium Predators | 0.13 | 0 | 0.06 |

Estimation of the additional juveniles parameters: T , W_k and K

Three additional parameters, the age (T) and weight (W_k) at which juveniles mature into adults, and the von Bertalanffy growth curve efficient (K) are required for the delay-difference

Ecosim. “T” was taken from the literature, although no value was found for the large predators (see Table Appendix 2.5 below). In the absence of other information, a higher value of $T=3$ was assumed, because of the viviparous nature of shark reproduction.

The weight at maturity was calculated from the length at maturity and the length weight relationships. *Saurida tumbil* data were used to represent the medium predators since the length weight relationship of *Trichiurus* sps has a ‘b’ quite different from the assumed cubic relationship between length and weight. Obtaining representative data for the large predators was again problematic however. The mean length of maturity for *Carcharinus melanopterus* is 96.4 cm ((Campagno 1984, Lyle 1987) The only length weight relationship found in the literature is from Lyle (1987) who estimated a ‘b’ of 3.65. Using this value, the estimated W_k for the large predators is unrealistically high (57.5 Kg). Unfortunately there are no maturity data for *Lates calcarifer* with which to estimate W_k In the absence of other data, a ‘b’ of 3 was assumed, producing a lower W_k of 2.95 Kg.

Values of the von Bertalanffy growth parameter, ‘K’, were taken from Chapter 2 for the sciaenids and *Trichiurus* spp. A value of $K=0.2$ (rounded) was used for the large predators (Reynolds and Moore 1973).

Although the parameters for the large predators are based largely on assumption, they do follow the type of pattern one would expect of this group, that is heavier weight at maturity, older “T” and a slower growth rate. Until better data is available, these “rough” measures will provide a good estimate of the likely response of this group to perturbations to the ecosystem.

Table A. 2.5. Additional Input Parameters for the Delay-Difference Ecosim

| | W_k | T | <i>von Bertalanffy 'K'</i> |
|------------------|-------|--------------------------------|----------------------------------|
| Sciaenids | 0.102 | 0.75 (Almatar 1993) | 0.4 |
| Medium Predators | 0.276 | 2 (Nakamura and Parin 1993) | 0.43 |
| Large Predators | 2.95 | 3 | 0.2 Reynolds and Moore (1973) |

Appendix 3

List of Acronyms

| | |
|--------|---|
| BFAR | The Bureau of Fisheries and Aquatic Resources |
| CPUE | Catch per unit effort |
| ELEFAN | Electronic Length Frequency Analysis |
| FAO | Food and Agriculture Organisation (United Nations) |
| FCI | Finn Cycling Index |
| FiSAT | FAO-ICLARM Stock Assessment Tools |
| FSP | The Fisheries Sector Programme of the Philippines (sponsored by the Department of Agriculture and the Asian Development Bank) |
| ICLARM | International Centre for Living and Aquatic Resource Management |
| LS | Large-scale fishery |
| REA | Resource and Ecological Assessment |
| SS | Small-scale fishery |
| VGBF | von Bertalanffy Growth Function |