EVALUATING THE EFFECT OF THE 1980 TRAWL BAN IN THE JAVA SEA, INDONESIA: AN ECOSYSTEM-BASED APPROACH

by

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ABSTRACT

Trawl fishing was banned in Indonesia in 1980 (except in the Arafura Sea) due to resource access conflicts between trawl operators and small-scale artisanal fishers. Recently, however, the new Indonesian government opened a debate on trawl ban revocation.

This research evaluates the effect of the 1980 trawl ban on the sustainability of Java Sea fisheries, using an ecosystem approach. Three approaches were used. The first was to construct an Ecopath model for the Java Sea prior to the ban, to get a snap-shot picture of the ecosystem before fishery development escalated. The second used Ecosim, a dynamic multispecies model, to project the pre-ban model forward for 50 years. Ecosystem dynamics were examined in light of four hypothetical scenarios. The third approach employed an economic impact analysis to evaluate the economic implications of a trawl ban revocation.

Results showed that the Java Sea in mid-1970s was a moderately mature and stable ecosystem, thus relatively resilient to perturbations. Various scenarios that were simulated suggested that the Java Sea was resilient enough to absorb perturbations, mainly by creating alternative stable states. However, these alternative stable states involved the loss of at least some parts of the initial food web structure. In all cases, this involved the increase of economically low-value species at the cost of the loss of economically high-value species.

Economic impact analysis predicted some short-term gains by opting for the trawl ban revocation. In the long term, however, this revocation would create a greater impact on ecosystem integrity, and further reduce the economic viability of the already highly-exploited fishery. The major conclusion is that the trawl ban has not so far been able to provide enough opportunity for most of the heavily impacted fished groups to recover. The failure to recover is partly due to the concurrent and continual increase of fishing pressure from purse seiners and small scale gears. Therefore, a revocation of the trawl ban is not recommended.

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CHAPTER 1:

INTRODUCTION

1.1 Brief overview of Indonesian marine capture fisheries

Indonesia is the world's largest archipelago, containing over 17,000 islands straddling the equator between 6°N and 11°S latitude. These islands endow Indonesia with a vast extent of marine environments, over 6.8 million km^2, comprising some 80% of the country's surface area (Kusuma-Atmadja 1991, Table 1). Records show that Indonesia maintains the richest marine biodiversity in the world, with for example, more than 2,000 fish species recorded (Froese *et al.* 1996). Viewed in such a setting, it is obvious that marine capture fisheries have through the centuries played an important role in the social, political, and economic activities of the archipelago (Bailey *et al.* 1987; Bailey 1988; Butcher 1996).

Recent statistics compiled by the National Development Information Office (1996) showed that the agricultural sector - embracing food and commodity crops, livestock, forestry and fisheries - remains the largest employer, accounting for over 46% of the nation's workforce, despite oil, gas, and now manufacturing being the leading contributors to the country's GDP. Indonesian marine capture fisheries data of the early 1980s showed that this sector employed approximately 1.2 million fishers (Directorate General of Fisheries[1] 1984), 90% of whom were artisanal operators (Bailey 1987). Bailey (1988) found that in Indonesia, the various fisheries acted as a social 'safety valve' for the landless and unemployed. This safety valve effect, the basic cause of the phenomenon now known as Malthusian overfishing[2], suggests that the number of artisanal fishers will increase with time (Pauly 1989, 1997).

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[1] The Directorate General of Fisheries or DGF is known as *Ditjen Perikanan* in Indonesia. DGF is a Jakarta-based national authority (an agency of the Ministry of Agriculture) responsible for the management and decision making processes in the Indonesian fisheries sector, which include marine capture, aquaculture, brackishwater and inland/freshwater fisheries. DGF has operational authorities in provincial and district levels known as Fisheries Service Offices or *Kantor Dinas Perikanan*, which provide periodical statistical report to DGF. Based on the reported data, the information is published annually by DGF.

[2] The Malthusian overfishing concept was hypothesized and defined based on the contention of Rev. T.R. Malthus (1798) that "production (of food [or fish in this case]) cannot *in the long run* keep up with an ever increasing demand" (Pauly 1997). Elaboration of this concept are given in Pauly (1994, 1997).

The marine fisheries resources in Indonesia are unevenly exploited. Exploitation mainly occurs in the western part of the country (Bailey 1987), simply because of proximity to the major markets and infrastructure, located on the nearby Java and Sumatra islands. As a result, conflicts over gear types and access to the resources tend to be concentrated in western Indonesia.

1.2 Problem identification and research question

Fisheries development in Indonesia (including the trawl fisheries), in particular in western Indonesia, is shaped by its historical background. Butcher (1996) comprehensively reviewed the economic history of marine fisheries in the western part of the archipelago, from 1850 to 1960s. One of key results was that various political and economic reforms enacted by the Dutch colonial government had major impact on the dynamics of fishing activities in the area, until independence in 1945, and beyond.

Prior to 1966, the marine fisheries in Indonesia were almost exclusively small scale in nature (Bailey 1987). Introduction of synthetic material for fishing nets occurred in late 1960s, enabling trawls and purse seines to develop (Martosubroto 1987). Even though both gears were introduced at about the same time, trawlers accounted for most of the increase in landings from the Malacca Strait and the Java Sea, until their ban in 1980 (McElroy 1991). With relatively low investment for high yields (Sardjono 1980), trawling was the dominant component of revenues from fisheries, mainly as a result of the capture of penaeid shrimps for export. Initially, those who invested in trawlers were members of the fishing industry. However, the profitability of trawling attracted investors from other sectors of the economy (Bailey 1997).

Penaeid shrimps typically prefer relatively shallow water with muddy bottom. Trawlers were attracted to these waters by the presence of penaeid shrimps. The same coastal areas have for generations been plied by traditional, mainly indigenous, artisanal fishers, who used simple traditional gears, such as *bagan tancap*[3] and sail boat or *jukung* (i.e, dug-out boats, made of

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[3] Stationary liftnets, built using bamboo platform on pilings in shallow waters and used at night to capture anchovies or other small schooling pelagic species which are attracted over the net with light from powerful kerosene pressure lamps (Bailey and Dwiponggo 1987, fig. 3.9).

hollowed logs). Problems arose because trawler operators and artisanal fishers generally exploited the same resource and often competed for access to the same coastal fishing grounds. Obviously, in this competition, the artisanal fishers were at a distinct disadvantage. Bailey (1997) noted that "[t]he larger trawlers, often operating at night without lights, occasionally rammed smaller boats and damaged or destroyed gill nets or other small-scale gear". This conflict was viewed by the artisanal fishers as unfair competition and it was considered as a serious threat to their livelihoods (Sardjono 1980). Initial conflicts lead to an accumulation of social tension, physical clashes and violence.

In response to this situation, the government had tried several times to restrict trawl operations through a series of regulatory measures (Table 1.1). However, a lack of effective monitoring and enforcement meant that the more aggressive trawl operators ignored the regulations, rendering them more or less ineffective (Sardjono 1980).

Table 1.1 Various decrees and year issued by the Minister of Agriculture, and primary issues addressed with the intent to restrict trawl operations (adapted from Bailey 1987b; and the original text of the decrees).

Decrees issued	Issues addressed
No. 561/1973	Called for the "rational" exploitation of fisheries resources. Specifically noted the destructive impact of trawling onto nursery grounds of commercially valuable finfish and shrimp species;
No. 40/1974	Stated that the trawler by-catch must be fully utilized rather than cast overboard;
No. 1/1975	Stipulated that the following aspect will be regulated: (1) seasons of operation, (2) type, size and number of boats in a particular area, (3) allowable mesh size, and (4) establishment of quota;
No. 607/1976	Establishment of fishing (coastal) zones, parallel to the shore in which operations of various types of boats (based on hull size and engine power) and gears were restricted;
No. 609/1976	Restricted trawlers to operations within areas for which they were specifically licensed.

Furthermore, regardless of the social problem, on the biological side, Indonesian scientists had already begun expressing their concerns about the impact of the trawl fishery on the demersal resources in the early 1970s (see e.g., Unar 1972).

Because many trawler owners were Indonesians of Chinese descent, the government became concerned that expanding violence would take on racial overtones, which would make it harder to control (Bailey 1997). In 1980, continuing social unrest and the recognized poor performance of restrictive regulations finally culminated in a government ban of trawl operations for western Indonesia (Presidential Decree No. 39/1980)[4]. Subsequently, in 1982 the ban was extended to cover all Indonesian territorial waters - except the Arafura Sea[5] (Sardjono 1980; Bailey and Dwiponggo 1987; Bailey 1987b, 1997). The main rationale given for this was to protect the poor artisanal fishers from unfair resource competition, by allowing them uncontested access to the traditional coastal fisheries resources (Sardjono 1980).

An account by Chong *et al.* (1987) who performed a comparative study of 1978 (pre-ban's) and 1984/1985 (post-ban's) costs and returns for artisanal fisheries showed that the trawl ban had a positive impact on profits at the aggregate level, particularly in the shrimp fishery. At the individual fishers' level, however, the improvement of incomes were marginal at best and not widespread.

Chong *et al.* (1987) reported that suggestions had been made by local industrialists to go back to trawling. However, the government was taking a very careful and considerate approach in addressing these calls; and hence, the issue was discussed only discreetly (Chong *et al.* 1987). In early 1996, an informal discussion, among fisheries scientists and managers in Indonesia, on trawl ban revision and revocation took place (Aziz 1996). One argument given in favor of the revocation was that after so many years without trawling, the ecosystem should have had recovered (Aziz 1996).

Since 1998, Indonesia has been undergoing massive changes in the political and economic arenas. Nearly continuous demonstrations and riots that culminated in May 1998 brought

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[4] See Appendix 1 for a translation of the decree

[5] In the Arafura Sea (eastern part of Indonesia), double-rigged shrimp trawlers owned by joint-venture and other large scale enterprises are still allowed to operate. Because there is little evidence that trawlers operating in this area are in conflict with local artisanal fishers, these trawlers were exempted from the 1982 ban; although concerns were expressed at the large bulk of non-shrimp by-catch dumped at sea by trawlers only interested in shrimp (Sujastani 1981). However, this type of trawl was later modified to include a by-catch exclusion device, and renamed 'shrimp net' (Bailey and Dwiponggo 1987; Bailey 1987).

down President Soeharto and his 'New Order', and ushered in President Habibie and his 'Reform Order'.

The Reform Order opened a public debate of the trawl ban. Moreover, the current Indonesian Minister of Agriculture, Dr. Soleh Solahuddin, announced during a working session of the Third Commission of Indonesian Legislative Assembly (*DPR RI*) on June 16, 1998 that the new government would review the trawl ban (*Kompas Online* [Jakarta], 17 June 1998). The Minister emphasized that this revision of the policy is intended to allow artisanal fishers to have access to trawling. Furthermore, the Minister indicated that this revision of the policy (already on the government's agenda) is a target for the reform of the fisheries sector in the country in the next two years.

Responding to the new government's plan of revoking the trawl ban, Taryono (1998) reviewed the historical background, implementation processes and occasional problems associated with it. Taryono's findings indicated that the trawl ban has been socially and institutionally established among artisanal fishers as a solution to a threat (i.e., trawling) to the marine ecosystem. Subsequently, he added that the advancement of fishing gear technology has been such that revoking the trawl ban now or in the future presents the artisanal fishers with an even greater threat than in the past. Thus, revoking the ban would create even more serious socio-political problems.

Underlying the emergence of the many arguments surrounding the trawl ban and its potential revocation is the fact that, while the importance of effective fisheries management is clearly understood in Indonesia, (so far) most government efforts have been directed towards resource development through expanded use of more productive fishing gear and boats, rather than through effort controls (Priyono and Sumiono 1997). Further, Priyono and Sumiono (1997) emphasized that existing management regulations in Indonesia attempt to protect both vulnerable resources and the small scale fishers' right of access to fishing grounds. These two objectives are highly conflictive by their very nature. In the long term, some very hard choices will have to be made.

Though the rhetoric would have it that the trawl ban has resulted in the recovery of the heavily exploited ecosystem, there is no data in support of this opinion. The lack of data can be attributed, at least in part, to the lingering ill feelings and mistrust of government by the artisanal fishers. Several years after the ban was imposed, an attempt was made by a government research vessel, equipped with trawl gear, to study the impact of the trawl ban on the ecosystem, in particular on the demersal resources (Bailey 1997). However, Bailey (1997) reported that the research team was threatened with attack by local fishers, and local authorities were unable to guarantee the vessel's safety. Hence, the researchers had to abandon their survey. Dwiponggo (1988) also suggested that on top of safety issues, budgetary constraints also limit the scientists ability to measure the biological impacts of the trawl ban.

Given the current climate of the government fisheries policies and capabilities (especially the lack of law enforcement capability) and the aforementioned problem identification (including the absence of studies measuring the biological impacts of the trawl ban), the author proposes the following research question: *"is it really biologically safe to revoke the trawl ban, and furthermore, would it benefit the fisheries ?"*. The nature of the research question requires analyses at the ecosystem scale, using ecosystem modelling and simulation. These analyses coupled with an economic impact analysis, extending over time and in the context of ecosystem dynamics, will address the issue of benefit to the fisheries. An in-depth explanation of approaches to be used in this study will be presented in section 1.5.

The Java Sea (Figure 1.1) is selected as a case study, because it is one of the most heavily exploited areas in Indonesia (see e.g., Martosubroto 1996); and one of the best studied regions in the country (see e.g., Martosubroto 1982; Pauly and Martosubroto 1996). The first rationale forms the basis for a sound research approach to explore ecosystem sustainability; while the second helps avoid being faced with 'no data' or 'not enough data', which is a common problem of fisheries research in developing nations.

In this study, the Java Sea is defined as a single ecosystem, and emphasis will be given to the impact of the trawl ban to that ecosystem. Although mild seasonal variability within the region does exist, in this work it will be ignored as trivial in comparison with the abrupt changes resulting from the trawl ban.

Though defining ecosystem boundaries is usually difficult, considering the Java Sea as an ecosystem is defensible. The main rationale is that the Java Sea is relatively secluded from neighboring waters by narrow straits (except in the eastern part), which limit water exchange with the open ocean. Wyrtki (1961) demonstrated that water exchange is pronounced only during the monsoons. During the east monsoon from June to August, highly saline water is pushed into the Java Sea from the Makassar Strait and the Flores Sea. During the west monsoon from December to February, low salinity water enters the Java Sea from river discharge due to heavy rain, and from the South China Sea (Figure 1.2).

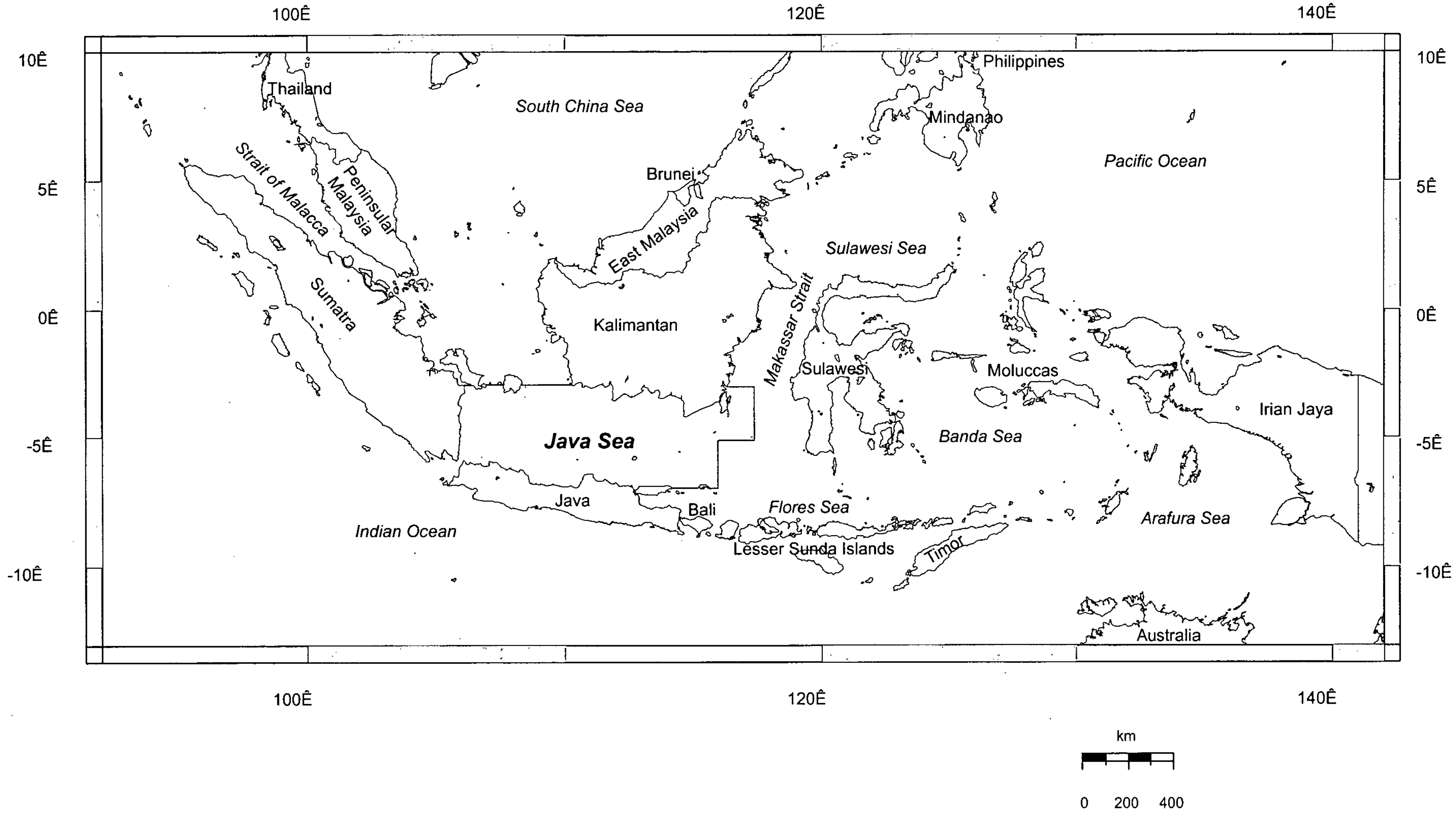

Figure 1.1 Location of the study area.

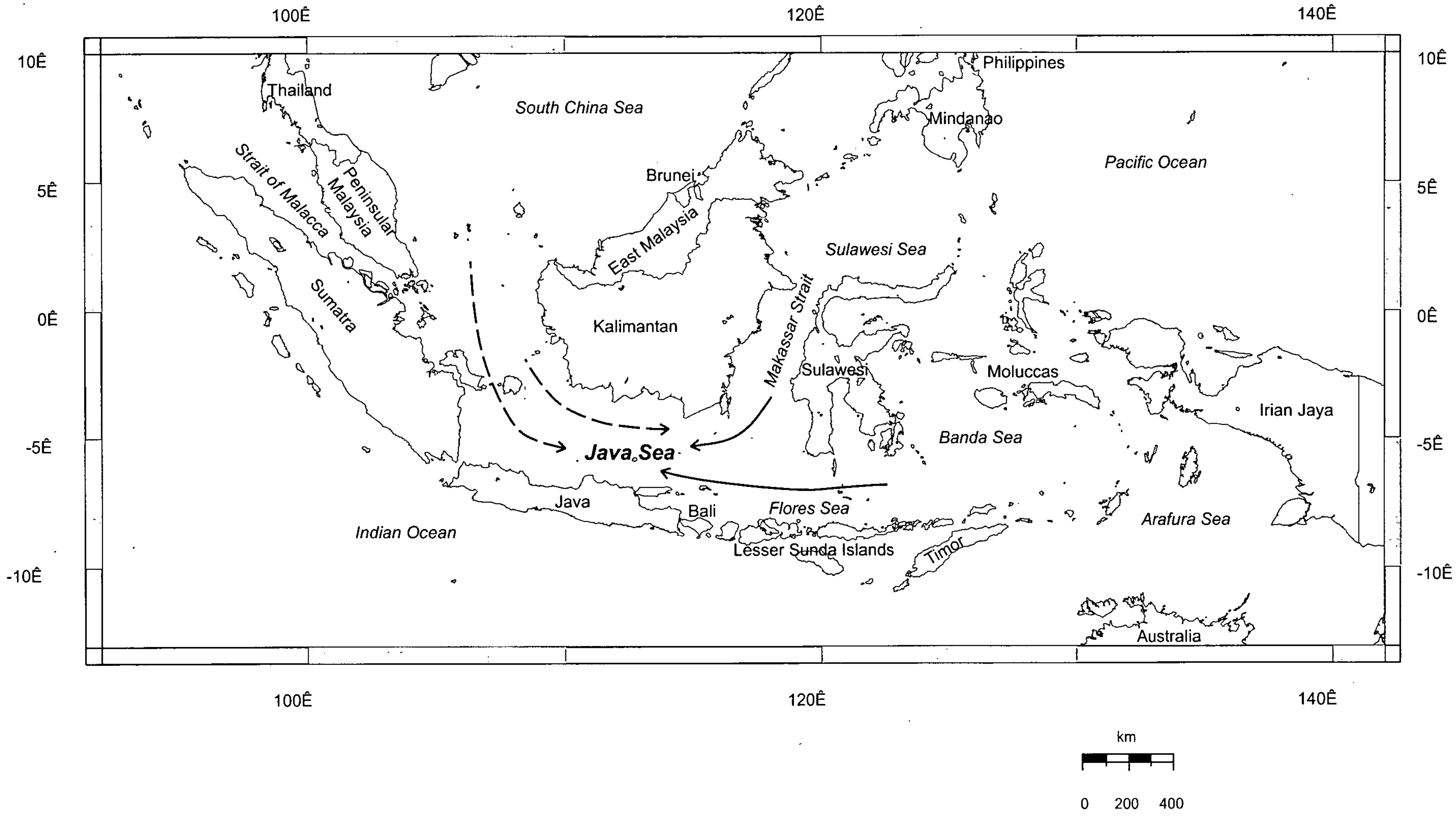

Figure 1.2 Surface current directions around the Java Sea. Solid lines represent east monsoon, dashed lines represent west monsoon.

1.3 Research objective

The main objective of this research is to evaluate the effect of the 1980 trawl ban on the Java Sea ecosystem. More specifically, this study aims to explore the behavior of a marine ecosystem subjected to drastic human perturbation (i.e., trawling), evaluate its recovery after the removal of the perturbation, and then to see whether it is ecologically defensible to re-impose a similar perturbation. The secondary objective is to perform an economic impact analysis of the Java Sea fisheries, in order to evaluate options for policy makers reviewing the trawl ban in Indonesia.

To achieve these objectives, three approaches were taken. The first approach was to construct a snap-shot model of the Java Sea ecosystem in mid-1970s, *viz.,* 1974 to 1975 (pre-ban period), using Ecopath, an equilibrium mass-balance model (Christensen and Pauly 1992a, 1992b, 1995). The reference period was selected because during that time trawling had just come to dominate the fisheries. Trawl operations in the western part of Indonesia took-off in the late 1970s, then dropped to zero by the early 1980s everywhere except in the Arafura Sea, allowing purse seining activity to increase. As Ecopath can only take a snap-shot of the ecosystem, strong differences between production and loss in the ecosystem (such as those caused by the trawl boom in the late 1970s, followed by the abrupt drop due to the 1980 trawl ban) should be avoided in the model. Hence, the selection of mid-1970s as the reference period for the Ecopath model of the Java Sea.

In the second approach, Ecosim, a dynamic multispecies simulation model (Walters *et al.* 1997) was used to simulate changes forward from the pre-ban period. This allows for an evaluation of the ecosystem's response to four simulated scenarios, i.e.:

(1) what would have happened to the ecosystem if the trawl ban had never been implemented ?

(2) what can be expected to happen to the ecosystem and the fisheries if the status-quo is maintained ?

(3) what would happen to the ecosystem and the fisheries if the trawl ban were to be revoked ?

(4) what would be the impact to the ecosystem if the living bottom structure (LBS, composed of sponges, gorgonians, sea fans, and other living organisms that are attached onto the sea floor) were eliminated through dramatic increase of trawling ?

As Ecosim cannot accommodate the qualitative changes that would result from invasion by previously absent species, and other structural changes, the simulations were presented over a relatively short period of 50 years (i.e., from the beginning of 1975 to the end of 2024).

Finally, the third approach employed an economic impact analysis (Hannesson 1993; Sumaila 1998; Pitcher *et al.*, in press) to evaluate the economic gains that would accrue under different simulated fisheries management scenarios.

By the end of this research, the author should be able to provide: (a) a basis for ecosystem-based management of the Java Sea fisheries resources, (b) a reference point for further socio-economic studies of the fisheries in the area, and (c) a stepping stone for further marine ecosystem studies in the archipelago.

1.4 Working hypotheses

The working hypotheses that the author uses for this research are as follows:

a. due to the trawl ban, there was a dramatic shift in ecosystem structure, from demersal to pelagic species;

b. the impact of trawling on living bottom structure (LBS) and several groups of demersal species that depend on it can be emulated, based on trophic relationships alone;

c. right after the ban, there was a short period of rapid growth in pelagic populations (due to the transfer of primary production from the demersal to pelagic subsystems), followed by a decline due to increasing pelagic fishing;

d. demersal populations recovered after the ban, except for those species that depend on LBS to survive.

1.5 Research methods

Walters *et al.* (1997) described an emerging consensus among fisheries scientists and managers of aquatic resources that traditional single-species approaches in fisheries management should be replaced by an ecosystem approach. This approach explicitly accounts for ecological interactions, especially those of a trophic nature, that cannot be captured within traditional single species approach. This approach fundamentally assumes that we cannot ignore the interactions among parts of an ecosystem, treating some as though they were isolated and others as non-existent. This is especially true in a tropical ecosystem context such as the Java Sea with multispecies, multigear fisheries.

Walters *et al.* (1997) identified and compared three approaches put forward so far to study trophic interactions within an ecosystem: (1) multispecies virtual population analysis (MSVPA; Sparre 1991); (2) differential equation models of biomass dynamics (Larkin and Gazey 1982); and (3) bioenergetic modelling (Stewart *et al.* 1981; Kitchell *et al.* 1994). Walters *et al.* (1997) concluded that none of these models were of general acceptability. Moreover, as those approaches require great amount of data, they are not applicable to ecosystems such as the Java Sea.

1.5.1 The Ecopath approach

A straightforward approach to study trophic interactions within an ecosystem, called 'Ecopath', was proposed by Polovina (Polovina 1984a, 1984b, 1993), for construction of equilibrium mass-balance models. Herein, the ecosystem in question is divided into several interacting compartments acting as 'functional groups', *viz.,* ecologically or taxonomically related species, single species, or size/age groups. The approach was then extended and developed by Christensen and Pauly (1992a,b; 1993), and its evolution continues today. The latest version of the Ecopath software package is 'Ecopath 4.0 with Ecosim' (Christensen 1998). However, when the Java Sea model was constructed, this version was not yet available, and hence, the ecosystem model built in this research used version 3.0 of Ecopath, the first to run under the Windows™ operating system (Christensen and Pauly 1995).

Ecopath models can be conceptually expressed (Christensen and Pauly 1992a, 1992b, 1995) as:

production by (i) - all predation on (i) - non predation losses of (i) - export of (i) = 0, for all *(i)*.

This can also be formulated as:

$$P_i - B_i \bullet M2_i - P_i \bullet (1 - EE_i) - EX_i = 0 \qquad \text{Eq. (1)}$$

where: P_i is the production of *(i)*; B_i is the biomass of *(i)*, $M2_i$ is the predation mortality of *(i)*, EE_i is the ecotrophic efficiency *(i)*, (1 - EE_i) is the 'other mortality' or non-predation losses of *(i)*, and EX_i is the export of *(i)* consisting of fishery catches and migration.

Equation (1) can be re-expressed as:

$$B_i \cdot (P/B)_i - \sum_{j=1}^{n} B_j \cdot (Q/B)_j \cdot DC_{ji} - (P/B)_i \cdot B_i \cdot (1 - EE_i) - EX_i = 0 \qquad \text{Eq. (2)}$$

or

$$B_i \cdot (P/B)_i \cdot EE_i - \sum_{j=1}^{n} B_j \cdot (Q/B)_j \cdot DC_{ji} - EX_i = 0 \qquad \text{Eq. (3)}$$

where: B_i is the biomass of *(i)* during the period in question, with the system having $i = 1, \ldots, n$ functional groups; $(P/B)_i$ is the production/biomass rate of *(i)* which is equal to total mortality rate (Z_i) under the assumption of equilibrium (Allen 1971); EE_i is the ecotrophic efficiency, i.e., the fraction of the production $(P_i = B_i \cdot (P/B)_i)$ that is consumed within the system or caught by fishing; EX_i is the export of *(i)*, which consists of catch (Y_i, with $Y_i = F_i \cdot B_i$, where F is the fishing mortality; B_j is the biomass of the consumers or predators) and can also include migration (if included, emigration is treated as a positive export, and immigration as a negative export); $(Q/B)_j$ is food consumption per unit of biomass for consumer *(j)*, and DC_{ji} is the fraction of *(i)* in the diet of *(j)* ($DC_{ji}=0$ when *(j)* does not eat *(i)*) (modified from Walters *et al.* 1997, and Christensen and Pauly 1992a).

Ecopath 3.0 and its successors include routines not only for estimation of biomasses, but also other ecosystem parameters, and for analysis of flows between the ecosystem elements (i.e., functional groups), based on various approaches in theoretical ecology as outlined by Ulanowicz (1986), Odum (1969), Ulanowicz and Norden (1990), and others. This allows users to estimate the maturity and stability of the ecosystem under study. A Monte Carlo routine (i.e., 'Ecoranger') is also included, enabling consideration, in a Bayesian context, of uncertainty in the input data (Walter 1996).

The maturity and stability of the Java Sea ecosystem in the mid-1970s were estimated using the routines for analysis of flows. The results are presented in Chapter 3, along with a brief overview of the trophic interactions in the system.

1.5.2 The Ecosim approach

Ecopath models provide only a snap-shot or static picture of an ecosystem's trophic structure. Thus, lacking a dynamic component, Ecopath cannot provide answers to "what-if" questions on various perturbations that affect the ecosystem, *viz.,* policy or ecosystem changes that would swing the balance of the trophic interactions. In 1996, the Ecosim software was developed (Walters *et al.* 1997) to use mass-balance Ecopath models in the context of simulation. Therein, the linear equations (see Eq. 1 to Eq. 3) that described trophic fluxes in mass-balance, equilibrium assessments of ecosystems were re-expressed as differential equations (see Eq. 4), defining trophic interactions as dynamic relationships varying with biomasses and harvest regimes (Walters *et al.* 1997). Perturbations are introduced into the ecosystem under study by changing the fishing mortality imposed on various functional groups. Thus, the equilibrium ecosystem model is transformed into a dynamic simulation model and the perturbation's impact can then be examined for each component in the ecosystem (Bundy 1997).

Ecosim modelling requires a mass-balance model as an input. Various experiments on adaptive management options in an ecosystem scale can then be run, without requiring the users to engage in, or have expert knowledge of, complex modelling nor the gathering of new information beyond that required by Ecopath. The main equation of Ecosim is:

$$dB_i/dt = f(B) - M_o B_i - F_i B_i - \sum_{j=1}^{n} c_{ij}(B_i, B_j) \qquad \text{Eq. (4)}$$

where:

$f(B)$ is a function of B_i if (i) is a primary producer, or:

$f(B) = g_i \sum_{j=1}^{n} c_{ji} \cdot (B_i, B_j)$ if (i) is a consumer;

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

c is food consumption rate per unit biomass;

g is gross food conversion efficiency = P/Q; and

M_0 is other (non-predation) mortality.

Moreover, Ecosim includes a representation of prey vulnerability, wherein each pool is split into a vulnerable biomass, and an invulnerable biomass. The rate of transfer between these two states (*v.f.*, vulnerability factor) determines whether the system is 'top-down' or 'bottom-up' controlled (see Walters *et al.* 1997 and section 1.5.2.2).

Ecosim modelling has two major routines[6], one performing dynamic simulations ('dynamic run routine'), and the other using the derivatives of equation (4) to simulate equilibrium conditions ('equilibrium fishing routine').

In the present study, only the 'dynamic run routine' was used. This routine allows the users to simulate various fishing patterns over the time period specified. The biomass of each functional group in the model is estimated through time in response to user-imposed changes in fishing regime[7]. The simulations are carried out by changing the relative fishing mortality of a single functional group, a specific gear type, or of all gears combined in the fishery. Fishing regime simulation is done graphically through the computer mouse, using a 'sketchpad' interface, to represent different trends in fishing mortality. In the present study, this routine was used to explore the responses of the ecosystem and trophic interactions throughout specified time, based on the four scenarios of the research objectives (see section 1.3). These runs were applied to all energy flow levels (see text at section 1.5.2.2).

The results from the Ecosim runs were dumped to an Excel spreadsheet for use in the economic impact analysis. This allowed for the forecast of economic yields from different

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[6] See Bundy (1997) and Walters *et al.* (1997) for details on these two routines.

[7] In the present study, three fishing regimes were applied for the simulations, i.e., trawl fishery, purse seine fishery and small scale fishery. See further explanation in Chapter 2 and 4.

fishing regimes, and identification of optimal returns from the fisheries (see further explanation in section 1.5.4).

1.5.2.1 Juvenile-adult split pools characteristics

As the population represented in the model consist of different age classes, juvenile/adult split functional groups or pools were introduced in both Ecopath and Ecosim approaches, to allow representation of ontogenic changes in feeding habit and other parameters.

Thus, in this study, some of the functional groups were split into two separate pools (juvenile and adult), to account for both trophic ontogeny and cannibalism. However, the limitation of Ecopath, which is presently designed to handle a maximum of 50 functional groups (Christensen and Pauly 1992a), has led to a decision of allowing only three functional groups with distinct trophic ontogeny to be split. These groups were penaeid shrimps, large pelagic predators and large demersal predators. Other groups represent all age/size classes.

Although trophic ontogeny is considered in Ecopath - by using split pools for juveniles and adults, and by having different diet composition for those three different groups - the dynamic interactions between juveniles and adults cannot be accommodated; hence, for example, the mixed trophic impact routine of Ecopath gives misleading results for split pools. On the other hand, Ecosim as a dynamic multispecies modelling tool, deals with this issue by tracking numbers of individuals in each split pool, thus allowing recruitment (i.e., flows from juvenile pools into adult pools) to be explicitly considered.

The split pools tracking is represented through delay-differential model (Deriso 1980) which takes into account not only the biomass of the biota that recruit into the adult pool, but also the absolute numbers (Walters *et al.* 1997).

Three additional parameters are needed in the Ecosim simulation, to define juvenile-adult links through the delay-differential model. They are: (1) the weight at which juveniles becomes adults ($W_{k,}$ in kg); (2) the age at which juveniles become adults (T_k, in years); and (3) the curvature parameter of the von Bertalanffy growth model (K, year^{-1}).

1.5.2.2 Top-down versus bottom-up control

In the absence of a unifying paradigm on how energy flow between trophic levels in an ecosystem is governed, two theories, *viz.,* top-down control with trophic cascades, and bottom-up, donor control, have dominated the ecological literature (Matson and Hunter 1992).

In top-down control (Carpenter *et al.* 1985; Carpenter and Kitchell 1993) predation drives the dynamics of the system. An increase in the abundance of a top predator has a 'cascading' effect down the food web. Thus, in a simple system, an increase in the abundance of animals at trophic level 4, causes an increase in the predation mortality on trophic level 3 (hence, a decrease of its biomass), and thus a reduction of predation mortality on trophic level 2 (hence, an increase of its biomass), and so on. An increase in productivity at the lower trophic levels has little effect on the structure of the system. In Ecosim simulations, top down control can be modelled by a high vulnerability factor (*v.f.*). A functional group or system with a high vulnerability factor would be one in which a large part (or all) of the prey biomass is vulnerable to predation (Walters *et al.* 1997).

Bottom-up control, in contrast, assumes that food availability limits the system (Hall *et al.* 1970; Hunter and Price 1992; Power 1992), and an increase in predator abundance does not lead to an increase in the mortality rate on the prey. Hence, energy flow is controlled by prey abundance and the mortality rate is stable over time. In Ecosim simulations, this is modelled by a low vulnerability factor (Walters *et al.* 1997).

The characteristics of energy flow dynamics in the Java Sea ecosystem are unknown, and no studies have been undertaken to examine this aspect. Therefore, in this study simulations were run assuming top-down (*v.f.*=10), bottom-up (*v.f.*=1.5), and an intermediate (*v.f.*= 4) system dynamics[8]. These *v.f.* values were obtained using the method described by Bundy (1997). Results of Ecosim simulations are presented in Chapter 4.

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[8] In 'Ecopath 4.0 with Ecosim' version, the *v.f.* scales are replaced by settings ranging from 0 to 1, with 0 representing bottom-up, donor control, 0.5 intermediate control, and 1 top-down control (Walters *et al.* 1998).

1.5.3 Economic impact analysis

For many decision and policy makers, in the developing world in particular, ecosystem analyses often do not appear meaningful unless translated into economic values. This applies to the management of fisheries as much as the management of any ecosystem subject to human impact. Hence, in this study the simulated consequences of ecosystem impact were translated into economic terms, notably for 'status-quo' (scenario 2, see section 1.3) and 'lift the ban' (scenario 3, see section 1.3) options. These terms were expressed as discounted net economic gains or losses. The impact analysis approach used was similar to the one applied to assess the impact of fishing by distant water fleets on both the ecology of the Namibian ecosystem and the economics of Namibia's fisheries (Sumaila 1998). It has also been applied to assess the impact of placing artificial reefs on both the ecology of Hongkong's ecosystem and the economics of Hongkong's fisheries (Pitcher *et al.* in press).

In this analysis, the results, notably biomass and fishing mortality, from Ecopath/Ecosim modelling were used to represent the dynamics of the ecosystem under different exploitation scenarios, and were used to derive estimates of yield. From here, prices and costs were then used to estimate net landed value, giving an indication of their economic impact.

Landing prices for the simulation period were projected using constant 1974 price (Martosubroto and Pauly 1976), converted to US$ using the 1974 exchange rate (1 US$ = 415 Rp.). The operating costs of fishing by gear type were obtained from a study conducted by Collier *et al.* (1977). In their account, it was found that average cost for trawlers, purse seiners and small scale gears relative to the gross landed value for each gear type were 16%, 24% and 29%, respectively. Based on this study, the average cost, relative to gross landed value, was used to estimate the net current landed value. To allow some sensitivity analysis, three discount rates were used in the analysis, i.e., 3%, 5%, and 10%.

Two options were modelled in this study, a 'status-quo option' and a 'lift the ban option'. The latter simulated a trawl ban revocation in the year of 2000. A normal economic and a shock economic scenarios were explored under both options. In the normal scenario, the economy was allowed to function as if the economic crisis had not occurred. In this case, an average exchange rate of Rp. 3,000 per US$ was used to assess the benefit of lift the ban option. For the shock scenario, in which the current economic crisis was modelled, an average exchange rate of Rp. 10,000 per US$ was assumed.

Thus, for each simulated option, total discounted net benefits (Hannesson 1993) were calculated for the entire fishery for 50 years, i.e., from the beginning of 1975 to the end of 2024, using:

$$\sum_{t=0}^{t=50} \left[1/(1+r)^t\right]\cdot\left[\sum\left(P_{i,t}\cdot Y_{i,t}\right)-C_t\right]$$

where:

r = discount rate;
t = time, in which $t = 0$ is the beginning of 1975 and $t = 50$ is the end of 2024;
$P_{i,t}$ = landing price (US$/tonne) of functional group i at time t;
$Y_{i,t}$ = total yield (tons) of group i (landed by all gears) at time t, under a given management scenario;
C_t = total operational cost (US$) of all gears at time t.

Finally, to compare between the 'status-quo' and lift-the-ban' options above, economic impact analysis was focused on the period of year 2000 to year 2024 (see section 4.2.6)[9].

The objective of this analysis was to estimate and compare the discounted net economic benefit for both options, at the aggregate level, for the entire fishery within the study area. From here, the results of the analyses were used to evaluate whether the revocation of the trawl ban is defensible from an economic perspective, given a proper account of food web interactions. Issues such as equity of distribution among various stakeholders, are considered to be outside the scope of this study. Results are presented in Chapter 4.

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[9] Although trawl revocation was simulated to officially start in the year 2000; trawlers that cheated were also simulated, and was simulated to start as early as 3 years prior to the official revocation.

CHAPTER 2:

REVIEW ON THE JAVA SEA FISHERIES

2.1 Physical settings of the study area

The Java Sea (Figure 2.1) is nearly rectangular in shape and covers approximately 471,000 km^2 (Martosubroto 1982). It is second in size and importance among the marine regions of the Indonesian archipelago, after the Banda Sea (Morgan and Fryer 1985). The Java Sea is surrounded by the three largest and most densely populated islands of Indonesia (i.e., Java, Sumatra, and Kalimantan islands). In total, these islands account for 85% of the Indonesian population (National Development Information Office 1992, 1994, 1996). The Java Sea falls within the jurisdiction of one municipality and 8 provinces (Figure 2.1).

Geographically, the Java Sea is located between approximately latitudes 3° S and 6° 50' S, and between longitudes 105° 50' E and 116° 30' E - 117° 30' E (Martosubroto 1982), and represents the most southern part of the Sunda Shelf. Its bottom gently slopes from the shoreline to the center and from the west to the east; the western part has an average depth of about 20 m, while the eastern part is about 60 m deep (Martosubroto 1982, 1996).

The Java Sea is a generally soft-bottomed ecosystem. Emery (1969) showed that 69% of the total bottom area comprised of thick gray mud, 17% of mud and sand, and 12% of sand. The remaining 2% of the sea bed consisted of rocks and coral, mainly in the vicinity of the Sunda Strait and along the edge of the continental shelf at the eastern boundary.

Wyrtki (1961) provided a comprehensive description of the physical oceanography of the Java Sea. A brief discussion of the extent of water exchange is given in Chapter 1 (see section 1.2). The direction of exchange is driven by the monsoon seasons. During the west monsoon, the current flows from the South China Sea with a speed of about 1.5 knots. On entering the Java Sea, the current splits towards the southwest into the Sunda Strait, and eastward along the north coast of Java island. Conversely, during the east monsoon, the currents flow to the west from the Banda Sea and Flores Sea, with a speed of about 0.5 - 1.0 knots (Emery *et al.* 1972).

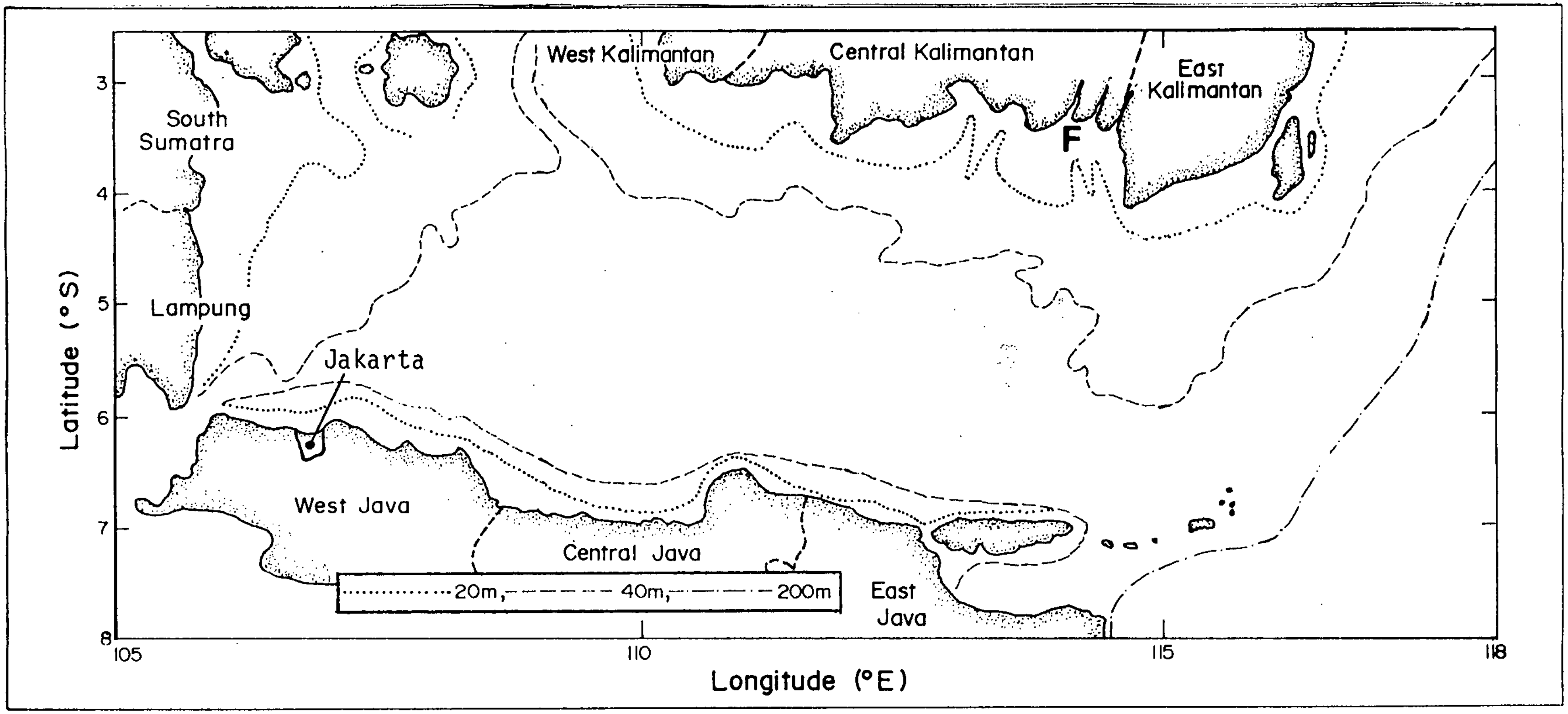

Figure 2.1 Map of the Java Sea showing its bathymetry, and the bordering eight provinces and one municipality (adapted from Martosubroto 1996).

Tide levels range between 1.5 to 3 m on the south coast of Kalimantan and the northeast coast of Java; elsewhere they are less than 1.5 m (Morgan and Fryer 1985).

Water temperature varies from 27° to 29°C, with average annual variations of sea surface temperature of less than 2°C (Dwiponggo 1979). The highest temperature occurs in September in the western part of the Java Sea, and lowest temperatures occur in the central and eastern part of the study area in the months of July and August (Losse and Dwiponggo 1977). A thermocline forms only during the months of June and July in the eastern part of the Java Sea, where the water is deep enough (Dwiponggo 1979).

Salinity ranges from 32.7 to 33.6 ‰ are found in open waters, but in coastal waters, salinity is about 29 ‰, due to local river discharge (Emery *et al.* 1972).

Apart from fisheries, shipping and oil drilling activities also occur in the Java Sea. Important oil drilling activities are located in the western areas, with their offshore moorings on the northwest coast of Java island capable of handling tankers in the 150,000 - 175,000 DWT[10] range (Morgan and Fryer 1985).

2.2 Multispecies and multigear nature of the Java Sea fisheries

Widodo (1976) recorded over 750 specimens of fish, distributed in 80 families, 146 genera and 231 species from the Java Sea and the most southern tip of the South China Sea. However, for the purpose of fisheries statistics, the Directorate General of Fisheries (DGF) only records 66 important species (Bailey 1987; Dwiponggo 1987; DGF various years). Those recorded include 45 finfish species and/or species groups, 18 species and/or species groups of both crustaceans and mollusks, and four other species groups (Table 2.1).

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[10] Dead weight tons

Table 2.1 Species categories for marine capture fisheries statistics as standardized by the DGF (adapted from Dwiponggo 1987; DGF various years).

Indonesian name	English name	Scientific name*)
Demersal fish groups:		
1. Ikan Sebelah	Indian halibuts	Psettodidae
2. Ikan Lidah	Tongue soles	Cynoglossidae, Pleuronectidae
3. Ikan Nomei	Bombay duck	*Harpadon nehereus*
4. Ikan Peperek	Ponyfishes/Slipmouths	Leiognathidae
5. Manyung	Marine catfishes	*Tachysurus* spp.
6. Beloso	Lizardfishes	*Saurida* spp.
7. Biji Nangka	Goatfishes	*Upeneus* spp.
8. Ikan Gerot-gerot	Grunters/Sweetlips	*Pomadasys* spp.
9. Ikan Merah/Bambangan	Red Snappers	Lutjanidae
10. Kerapu	Groupers	Serranidae
11. Lencam	Emperors	*Lethrinus* spp.
12. Kakap	Barramundi breams	*Lates calcarifer*
13. Kurisi	Threadfin breams	*Nemipterus* spp.
14. Swanggi	Big eyes	*Priacanthus* spp.
15. Ekor kuning/Pisang-pisang	Yellow tail/Fusiliers	*Caesio* spp.
16. Gulamah/Tigawaja	Croakers, drums	Sciaenidae
17. Cucut	Sharks	Carcharhinidae, Sphyrnidae, Orectolobidae
18. Pari	Rays	Trigonidae
19. Bawal Putih	Silver Pomfrets	*Pampus argenteus*
20. Kuro/Senangin	Threadfins	*Polynemus* spp.
21. Layur	Hairtails	*Trichiurus* spp.
Crustacean groups:		
22. Rajungan	Swimming crabs	*Portunus* spp.
23. Kepiting	Mud crabs	*Scylla serrata*
24. Udang Barong	Spiny lobsters	*Panulirus* spp.
25. Udang Windu	Tiger prawns	*Penaeus monodon, P. semisulcatus*
26. Udang Putih/Udang Jrebung	Banana prawns	*Penaeus merguensis, P. indicus*
27. Udang Dogol	Endeavour prawns	*Metapenaeus* spp.
28. Jenis-jenis udang lain seperti rebon udang pasir	All shrimps other than listed above	Mysids, sergestids, and penaeid post larvae
29. Binatang berkulit keras lainnya	All crustaceans other than those listed above	-
Mollusk groups:		
30. Tiram	Cupped oysters	*Crassostrea* spp.

Table 2.1 Species categories for marine capture fisheries statistics as standardized by the DGF (adapted from Dwiponggo 1987; DGF various years).

Indonesian name	English name	Scientific name*)
31. Simping	Scallops	*Amusium* spp.
32. Remis	Clams	*Meretrix* spp.
33. Kerang darah	Blood cockles	*Anadara* spp.
34. Cumi-cumi	Common squids	*Loligo* spp.
35. Sotong	Cuttlefishes	*Sepia* spp.
36. Gurita	Octopuses	*Octopus* spp.
37. Binatang lunak lainnya	All mollusks other than those listed above	-
Pelagic fish groups:		
38. Bawal Hitam	Black Pomfrets	*Formio niger*
39. Alu-alu	Barracudas	*Sphyraena* spp.
40. Ikan Layang	Scads	*Decapterus* spp.
41. Selar	Trevallies/Yellow stripped trevallies	*Selar* spp., *Selaroides* spp.
42. Kuwe	Jacks, Trevallies	*Caranx* spp.
43. Tetengkek	Hardtail scads	*Megalaspis cordyla*
44. Daun Bambu/Talang-talang	Queenfishes	*Chorinemus* spp.
45. Sunglir	Rainbow runner	*Elagatis bipinnulatus*
46. Ikan terbang	Flyingfishes	*Cypselurus* spp.
47. Belanak	Mullets	*Mugil* spp.
48. Julung-julung	Garfish, Halfbeak	*Tylosurus* spp., *Hemirhamphus* spp.
49. Teri	Anchovies	*Stolephorus* spp.
50. Japuh	Sardines	*Dussumieria* spp.
51. Tembang	Fringescale sardinella	*Sardinella fimbriata*
52. Lemuru	Indian oil sardinella	*Sardinella longiceps*
53. Golok-golok/Parang-parang	Wolf-herrings	*Chirocentrus* spp.
54. Terubuk	Tolishads (Chinese herrings)	*Clupea toli*
55. Kembung	Indo-Pacific mackerels	*Rastrelliger* spp.
56. Tenggiri papan	Indo-Pacific Spanish mackerels	*Scomberomorus guttatus*
57. Tenggiri	Narrow-barred Spanish mackerels	*Scomberomorus commersoni*
58. Tongkol	Eastern Little Tuna	*Euthynnus* spp.
59. Tuna	Tunas, Big tunas, Broadbill/Swordfishes, Indo-Pacific marlin, Indo-Pacific sailfishes	*Thunnus* spp., *Parathunnus obesus, Xiphia* spp., *Makaira* spp., *Istiophorus orientalis*
60. Cakalang	Skipjack tunas	*Katsuwonus pelamis*

Table 2.1 Species categories for marine capture fisheries statistics as standardized by the DGF (adapted from Dwiponggo 1987; DGF various years).

Indonesian name	English name	Scientific name*)
Other marine biota:		
61. Ikan-ikan lain	Other fish, i.e., all fishes other than those listed above (includes both demersal and pelagic groups)	-
62. Rumput laut	Seaweeds	*Euchema* spp., *Gracilaria* spp.
63. Penyu	Marine turtles	Not identified
64. Teripang	Sea cucumbers	Not identified
65. Ubur-ubur	Jellyfish	Not identified
66. Hewan laut lainnya	All marine biota other than those listed above	-

Note: *) The scientific name of biota comprised of more than one species in one category are listed at the genus or family level.

Reflecting the diverse nature of the ecosystem, a wide array of fishing gear types have always been in use. DGF listed 28 categories of the most important gears used in the country, ranging from the simplest hand line to the most sophisticated purse seiners and trawlers (Bailey 1987; DGF various years). Most of these gears are used today in the Java Sea (Table 2.2), with the exception of *Huhate* (Pole and Line), *Rawai Tuna* (Tuna Long Line) and *Pukat Udang* (shrimp trawlers, after the ban).

For management purposes, DGF categorized marine fisheries in Indonesia into three subsectors: small scale fisheries, medium scale fisheries, and large scale fisheries (Bailey and Dwiponggo 1987; Priyono and Sumiono 1997). The categorization of the fisheries is mainly based on the level of investment required, in turn reflected by the vessel type. Additionally, medium and large scale fisheries are also defined by legal criteria.

Table 2.2 Gear types considered important and widely used in Indonesia (adapted from DGF; various years).

Indonesian name	English name
1. Pukat pantai	Beach seines
2. Dogol	Danish seines (demersal)
3. Payang	Lampara seines (pelagic)
4. Pukat cincin	Purse seines
5. Pukat udang	Trawls
6. Jaring insang hanyut	Drift gill nets
7. Jaring insang lingkar	Encircling gill nets
8. Jaring klitik	Shrimp gill net
9. Jaring insang tetap	Surface fixed gill nets
10. Muroami	Drive-in reef nets
11. Bagan perahu/rakit	Drift lift nets (with boat)
12. Bagan tancap (termasuk Kelong)	Stationary lift nets (incl. Kelong)
13. Jaring angkat lainnya	Other lift nets
14. Rawai tetap	Stationary long lines
15. Rawai tuna	Tuna long line
16. Rawai hanyut lainnya	Other drift long lines
17. Huhate	Pole and line
18. Pancing tonda	Troll lines
19. Pancing lainnya	Other lines
20. Trammel nets	Trammel nets
21. Alat pengumpul kerang	Mollusks collectors
22. Alat pengumpul rumput laut	Seaweeds collectors
23. Serok	Scoop nets
24. Bubu	Portable traps
25. Jermal	Stow net traps
26. Sero	Guiding barrier traps
27. Perangkap lainnya	Other traps
28. Jenis alat lainnya	Other gears *

* Cast nets, harpoon, etc.

Small scale fisheries are defined as consisting of all fishing units which use sail-powered boats, or outboard engine-powered boats (Priyono and Sumiono 1997). As well, fishers who operate fixed or mobile gears without any boat are assigned to small scale fisheries, regardless of the size of their gears. There are twelve types of gear that are employed in the small scale fisheries, representing 81% of all gear units (Bailey and Dwiponggo 1987). This subsector includes more gear variety than the medium and large scale subsectors combined, including gill nets, seine nets, cast nets, drive-in reef nets, traps, trammel nets, shrimp (monofilament) gill nets, hand lines, and shellfish and mollusks collectors. Vessels in this subsector are divided into five categories (DGF 1975):

(1) Dug-out boats or *jukung* - made of hollowed-out logs;
(2) Small non-powered plank-built boats; up to 7 m long;
(3) Medium non-powered plank-built boats; from 7 to 10 m long;
(4) Large non-powered plank-built boats; more than 10 m long; and
(5) Boats powered by outboard engines (with long propeller shaft, gasoline or diesel, 2 to 15 HP [Horsepower]).

The second category is medium scale fisheries (Priyono and Sumiono 1997), which includes privately-owned (as opposed to company-owned) boats, of generally less than 5 gross tons (GT) and up to 30 GT, that use inboard engines. Medium scale fishing boats tend to have access to few or none of the shore-based amenities such as ice plants, cold storage facilities or workshops (Priyono and Sumiono 1997). Individual operators own one or, at the most, a few fishing units. The most common gears used by medium scale fisheries are skipjack pole and line, purse seines, gill nets, otter trawls (until the 1980 ban), and trammel nets.

At the upper end of the category are the large scale fisheries (Priyono and Sumiono 1997). These operate in isolation from the small and medium scale fisheries, due not only to the offshore nature of the fishing operation, but also because of their export orientation. The large scale fisheries are operated by a combination of private domestic (Indonesian) companies, joint-venture corporations, and state-owned enterprises (Bailey and Dwiponggo 1987). However, the large scale fisheries mainly operate in the eastern part of the country, and do not operate at all in the Java Sea. Hence, this subsector is outside the scope of this study. A detail review on small scale, medium scale and large scale fisheries are given in Bailey and Dwiponggo (1987). A detail nomenclature, including common names of fishing gears and vessels in Indonesian is given in Ayodhyoa and Diniah (1989).

For the purpose of dynamic multispecies modelling in this study (Chapter 4), all fishing gears were grouped into three categories, namely trawlers, purse seiners and small scale or artisanal gears. The following is a brief overview of the characteristics of each category, as applied in the Ecosim simulation presented below.

The Trawl Fishery

Typical Indonesian trawlers were first used in 1966 in area surrounding the Rokan estuary in the Malacca Strait (Tiews 1973). The little coastal town of Bagan Siapi-api, in Riau province, Sumatra, gave the trawlers their present name of "*Pukat Bagan Siapi-api*" (Butcher 1996). Although this trawl fishery mainly targeted shrimp, the non-selective nature of the gear meant it caught almost everything. Hence, in Indonesia, it is also well known as "*Pukat Harimau*" or "Tiger Trawl".

The fishery was a medium scale fishery. Vessels had wooden hulls, displaced 5 to 35 GT, and were powered by inboard engines generating 66 to 200 HP. This type of vessel usually had a crew of five, and employed a single gulf-type (polyethylene nets) otter trawl of 12 to 22.5 m headrope length, with cod-end mesh size of 20 to 25 mm (Priyono and Sumiono 1997; Bailey 1997).

The Purse Seine Fishery

Purse seiners first appeared in the 1970s at Batang harbor, in the northern coast of Central Java. The trawl ban significantly increased the numbers of purse seiners that operate in the region. By 1982, this fishery covered the whole Java Sea, the southern part of South China Sea, and the mouth of Makassar Strait (Nurhakim *et al.* 1987). In the early years, purse seiners mostly caught anchovies (*Stolephorus* spp.), but in latter years they were replaced by scads (*Decapterus* spp.) and Indian Mackerel (*Rastrelliger* spp.).

Nurhakim *et al.* (1987) and Potier and Sadhotomo (1995) indicated that based on geographical condition, human behavior factors and availability of large fish resources, the purse seine fleet in the Java Sea can be categorized into three groups: the mini purse seiners, the medium purse seiners and the large purse seiners.

Mini purse seiners use either semi dug-out or plank-built wooden vessels, that have a length of 10 to 18 meters. They have mostly outboard engines and are equipped with seines nets up to a maximum length of 300 meters. The fleet exploits the coastal waters of the north coast of Java island and the south-east coast of Kalimantan island (i.e., about 30 miles away from the shores). Fishing trips usually last from one to three days. Most of the fleet is based in east Java province and around Laut island in South Kalimantan (Nurhakim *et al.* 1987; Potier and Sadhotomo 1995).

The medium purse seiners, unlike the other two fleets, entered the fishery around 1987 in the north coast of central Java, and operate only from Pekalongan harbor. The vessels have the traditional hulls of the north coast of central Java, and a length of 15 to 20 meters. The usual fishing grounds for this fleet are located in the Java Sea, over depths of 30 to 45 meters. Trip lengths range from 8 to 15 days (Nurhakim *et al.* 1987; Potier and Sadhotomo 1995).

The last category of purse seiners are *"Cungking"* or *"Malav"* boats of 20 - 35 m, with flat bottoms and shallow draught. They are generally equipped with inboard engines (of at least 160 HP), and seines with length of about 400 meters or more and a mesh size of 2.5 cm. This large purse seine fleet exploits the whole of the Java Sea, especially areas of deeper than 50 meters, the southern part of the South China Sea, and the Makassar Strait. Fishing trips usually last 25 days, but the vessels, which carry a crew of 30 - 40, are capable of trips of up to 40 days (Nurhakim *et al.* 1987; Potier and Sadhotomo 1995).

Most purse seine vessels have not been modernized to any great extent. However, beginning in mid-1980s, a few new vessels operated by masters trained in State Fisheries Technical Schools, have been equipped with modern equipment such as echo sounders and single side band (SSB) radio (Nurhakim *et al.* 1987).

Judging from the types of the vessel used by these purse seiners, the mini purse seiners can be categorized as a small scale fishery, while the last two categories are medium scale fisheries. Unfortunately, the Indonesian fishery statistics do not differentiate whether a purse seine gear is of mini, medium or large purse seine types. All purse seine gears are lumped together as one category (see Table 2.2). Hence, in the Ecosim simulation (Chapter 4), the "purse seiners" category also lumps all of the above three purse seiner categories.

Small Scale or Artisanal Gears

The importance of this gear category is given by the fact that in 1981, 95% of Indonesian fishing vessels were of 10 GT or less, and 98% of marine fish landings came from these crafts operating in coastal waters (Gunawan 1986). For the purpose of Ecosim modelling (Chapter 4), all remaining gear types are aggregated into this category. This category thus includes a variety of gears (see Table 2.2), from the simple hand lines (*pancing*), to bamboo traps (*bubu*), to the crew intensive drive-in coral reef nets (*muroami*). Detailed descriptions of these various gears may be found in Bailey and Dwiponggo (1987).

2.3 Fisheries statistics system

The fisheries statistics system in Indonesia was initiated in 1972 (Martosubroto 1982). However, the statistics were simple at first, and often did not consider such factors as the location of the catches (Martosubroto 1989). With the assistance of a statistician assigned by FAO from 1973 to 1977, Dr. Tadashi Yamamoto, improvements were made (Yamamoto 1980). In 1976 the national fisheries statistics system (incorporating such new items as periodic data on landings, fishers, gears, and vessels) presently used was officially instituted (Dudley and Harris 1987; Martosubroto 1989). However, since the FAO project started in 1973, statistical data that were collected in 1974 and 1975 (used in this study) were collected by the new methods designed by Dr. Yamamoto (Ms. Alfida, DGF, *pers. comm.*, 7 November 1996). Currently, DGF is planning to revise and improve the system over the next two to three years (Ms. Alfida, DGF, *pers. comm.*, 7 November 1996).

A series of manuals describing the Indonesian fisheries statistics system was first published by DGF (in Indonesian) in four volumes in 1975, then revised in 1980 and 1990 (DGF 1975, 1980, 1990a, 1990b). The English version of the system was comprehensively described in Yamamoto (1980), then reviewed and summarized by Dudley and Harris (1984, 1987).

Given the vastness of the marine waters of Indonesia, for the purpose of fisheries statistics, Indonesian marine waters were divided into eleven major statistical areas (Figure 2.2), based on landing areas (DGF 1975, Table 1; Martosubroto 1989).

One problem is that because some provinces are also facing marine areas other than the Java Sea (Figure 2.1 and 2.2), report of landings from these areas are lumped with landings from the Java Sea. For example: (a) landings reported from West Kalimantan province will include landings from the South China Sea, (b) landings reported from South Kalimantan province will include landings from the Makassar Strait, and (c) landings reported from South Sumatra province will include landings from the Bangka Strait and perhaps also the lower part of South China Sea. For this study, to account for the over-aggregation of statistical data, adjustments were made following Martosubroto (1996), based on the relative coastline length within each provincial unit that faces the Java Sea.

The assumptions involved here are as follows:

(1) The fisheries statistics (i.e. landings, numbers of boats and fishers) for the part of the coast of South Sumatra province facing the Java Sea account for 35% of the total landing of that province;

(2) The fisheries statistics (i.e. landings, numbers of boats and fishers) for the part of the coast of West Kalimantan province facing the Java Sea account for 5% of the total landing of that province;

(3) The fisheries statistics (i.e. landings, numbers of boats and fishers) for the part of the coast of South Kalimantan province facing the Java Sea account for 75% of the total landing of that province.

Obviously, any drastic changes in the level of effort within each provincial unit could invalidate these assumptions. However, such drastic changes did not occur during the relevant mid-1970s period (Martosubroto 1996).

Another problematic issue in the Indonesian fisheries statistical system is the 'other fish' category (Table 2.1). This category - also often called 'trash fish' - lumps all landings of fish that are either unimportant economically, or important species that are below marketable size. It is not possible to identify the specific groups to which fish from this category belong. In order to incorporate landings from this category into the Ecopath model, the landings were divided into two categories, small demersals and miscellaneous pelagics, based on the proportion of the known average landings for these two groups from the Java Sea during mid-1970s. The ratio of landings between small demersals and miscellaneous pelagics in this period was 0.32.

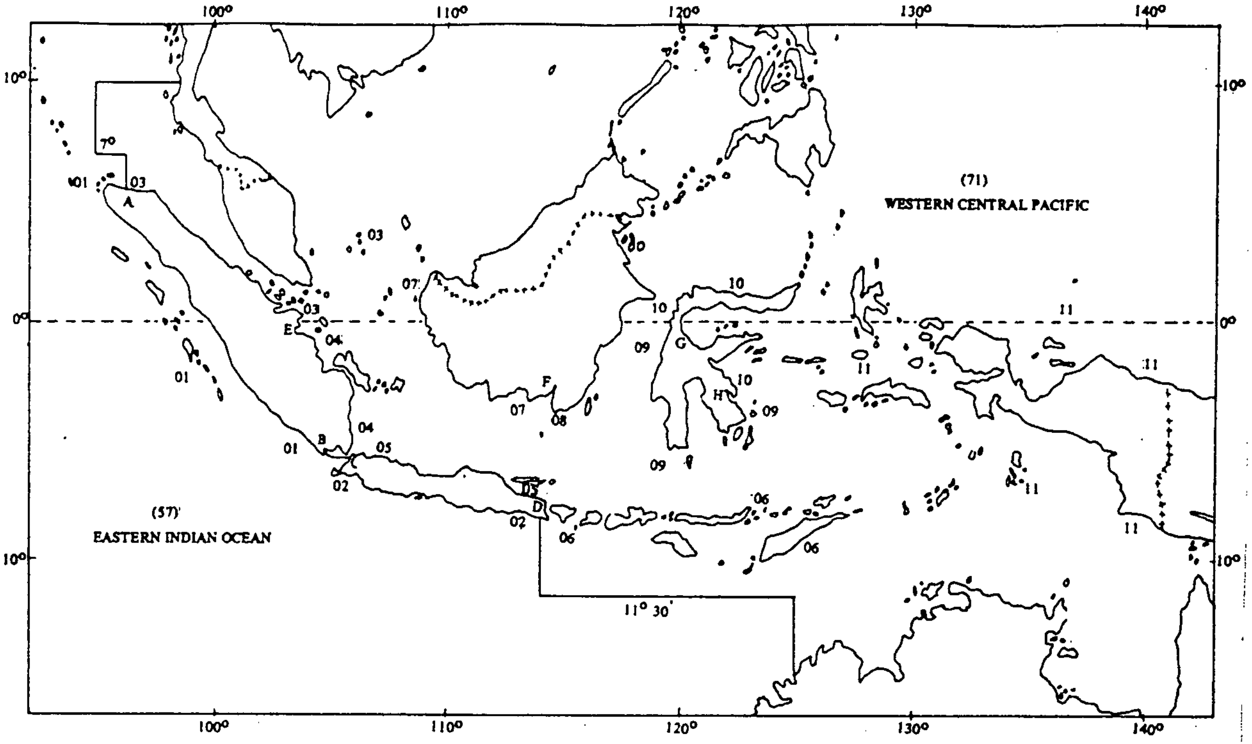


Figure 2.2 Major fisheries statistical areas in Indonesia. Note that western Sumatra coast (area 01) is included in area 57 (Eastern Indian Ocean). Other areas are included in area 71 (Western Central Pacific), *viz*., the coast of the Strait of Malacca (area 03), eastern Sumatra coast (area 04), northern Java coast (area 05), the coasts of Lesser Sunda Islands and Bali (area 06), southern and western coasts of Kalimantan (area 07), eastern coast of Kalimantan (area 08), southern coast of Sulawesi (area 09), northern coast of Sulawesi (area 10), and the coasts of Moluccas and Irian Jaya (area 11). Point 'A' separates area 57 from area 71, and is located at the boundary line between Aceh Besar district and Pidie district. Point 'B' separates area 57 from area 71, and is located at the boundary line between South Lampung district and North Lampung district. Point 'C' separates area 57 from area 71, and is located at the boundary line between Pandeglang district and Serang district. Point 'D' separates area 57 from area 71, and is located at the boundary line between Jember district and Banyuwangi district. Point 'E' separates area 03 from area 04, and is located at the boundary line between Riau province and Jambi province. Point 'F' separates area 07 from area 08, and is located at the boundary line between Central Kalimantan province and East Kalimantan province. Point 'G' separates area 09 from area 10, and is located at the boundary line between Southeast Sulawesi province and North Sulawesi province. Point 'H' separates area 09 from area 10, and is located at the boundary line between South Sulawesi province and Central Sulawesi province (Source: DGF 1975).

2.4 Statistics of landings, gears, vessels and fishers

The development of fisheries in the Java Sea started in early 1970s, corresponding with the introduction of trawlers and purse seiners into the region (Martosubroto 1987). This period also marked the launch of the first five-year national development plan - also known as REPELITA I (1969 to 1974) - which emphasized various development programs for the fisheries sector, both marine capture and aquaculture fisheries (Republik Indonesia 1968; Zachman 1973).

For the Java Sea fisheries, however, a remarkable subsequent rapid growth, in terms of landing (Figure 2.3) and landed value occurred. An account by Martosubroto (1982) indicated that in nine years, from 1970 to 1979, the total catch from the study area more than doubled, from 140,000 t to more than 332,000 t. Fisheries statistical reports from the DGF showed that total catch from the Java Sea was 538,221, 674,971 and 885,032 t in 1985, 1990, and 1994, respectively. Compared to its initial stage of development in 1970, the marine fisheries landings from the Java Sea increased by more than 500% in 24 years; the majority of this was landed by small scale or artisanal fishing gears (Figure 2.4).

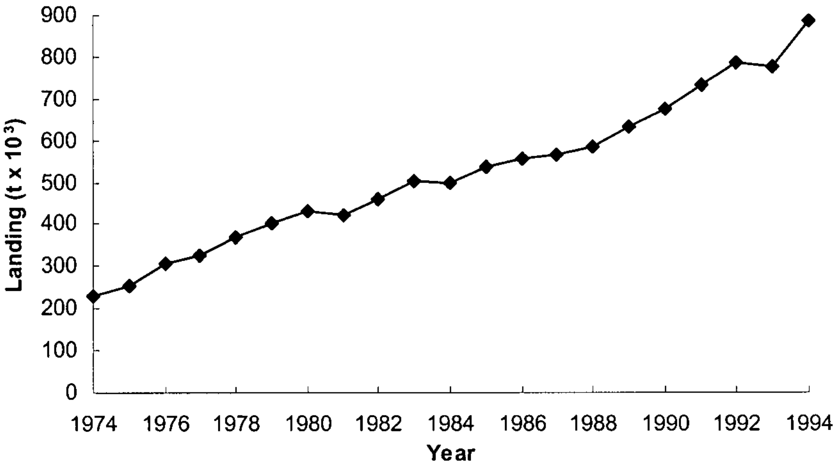


Figure 2.3 Trend of total marine fisheries landings from the Java Sea, from 1974 to 1994 (data from DGF, various years)

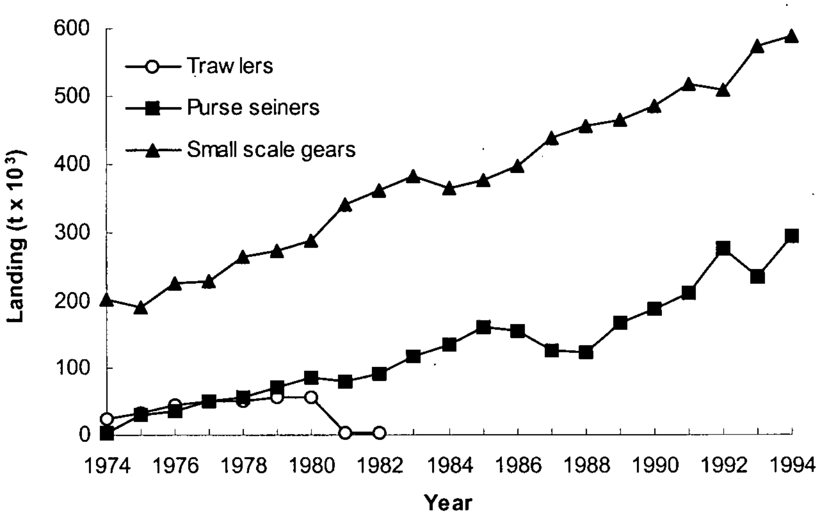


Figure 2.4 Trend of annual landings per gear type from the Java Sea, 1974 - 1994 (data from DGF, various years)

When small scale or artisanal fishing gears were segregated into different gear groupings - e.g., seines (excluding purse seines), gill nets, trammel nets, lift nets, hooks and lines, traps, and other gears - landing data showed that gill nets have dominated landings (Figure 2.5) and have increased more than 500% since the mid-1970s.

Shrimps were the major target for demersal fisheries, due to their high value (Martosubroto 1987). However, during the mid-1970s, the bulk of the catch from the Java Sea consisted of pelagic species. These species were mainly caught by purse seines, Lampara seines, and drifting gill nets (Dwiponggo 1987). This trend continues to the present (Figure 2.6). Dwiponggo (1987) indicated that Round scads (*Decapterus* spp.) are the single most important pelagic species, followed by sardines (*Sardinella fimbriata*), anchovies (*Stolephorus* spp.) and Indo-Pacific mackerels (*Rastrelliger* spp.).

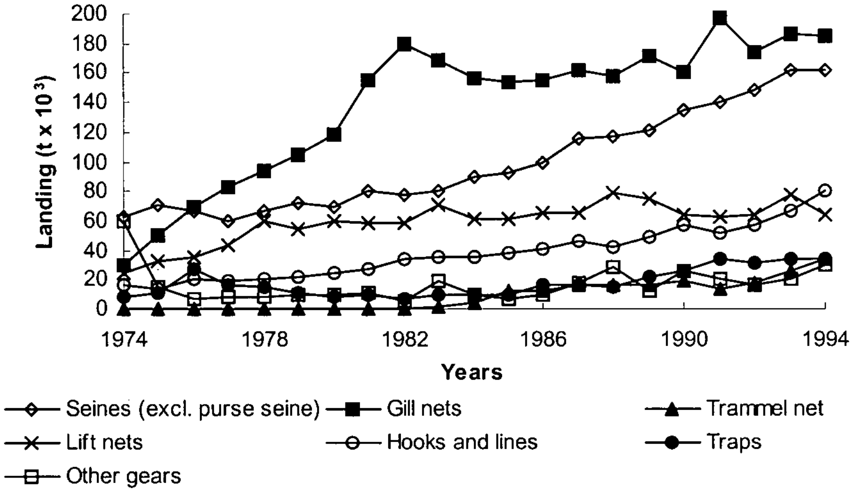


Figure 2.5 Annual landings per small scale gear type from the Java Sea, 1974 - 1994 (data from DGF, various years)

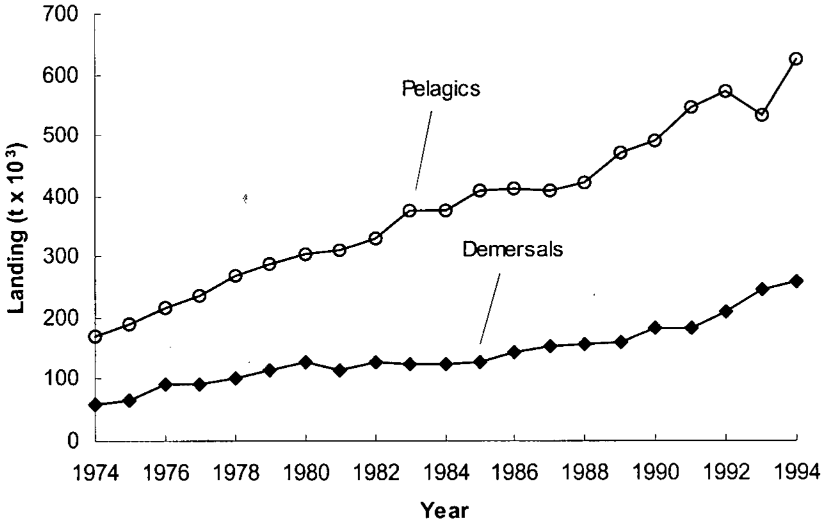


Figure 2.6 Demersal and pelagic fish landings from the Java Sea, 1974 - 1994 (data from DGF, various years)

Until they were banned in 1980, trawlers accounted for virtually all of the shrimp landings (Martosubroto 1982). After 1980, marine capture shrimp landings from the Java Sea originated from trammel nets, shrimp gill nets, bottom gill nets, Danish seines and stationary tidal traps (Chong *et al.* 1987), all of these are small scale in nature (DGF statistics, various years). After the ban, loan programs were extended in order to intensify brackishwater pond culture of penaeid shrimps (Bailey 1987).

The principal target of the marine capture shrimp fishery are penaeid shrimps, with Banana prawns (*Penaeus merguensis* and *P. indicus*) dominating, followed by Endeavour prawns (*Metapenaeus* spp.) and Tiger prawns (*P. monodon* and *P. semisulcatus*). Landing statistics (Figure 2.7) showed that the ban on trawling negatively affected the landing of Banana prawns in the year after it was promulgated.

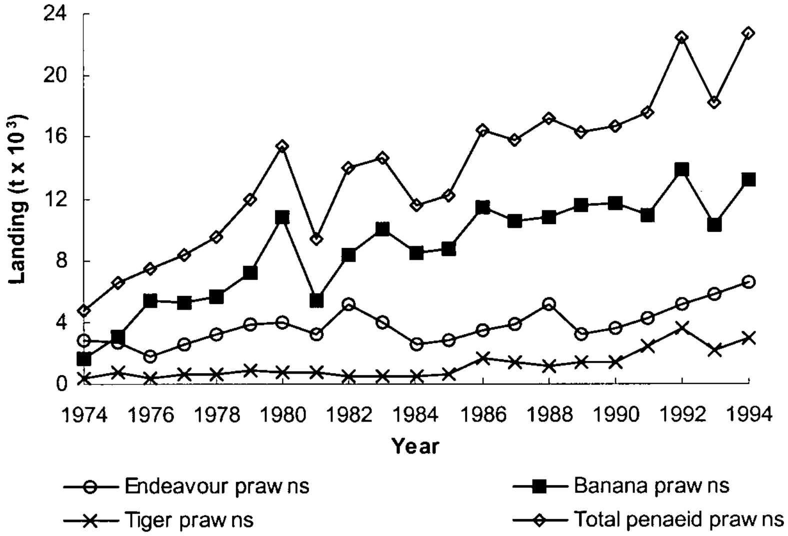
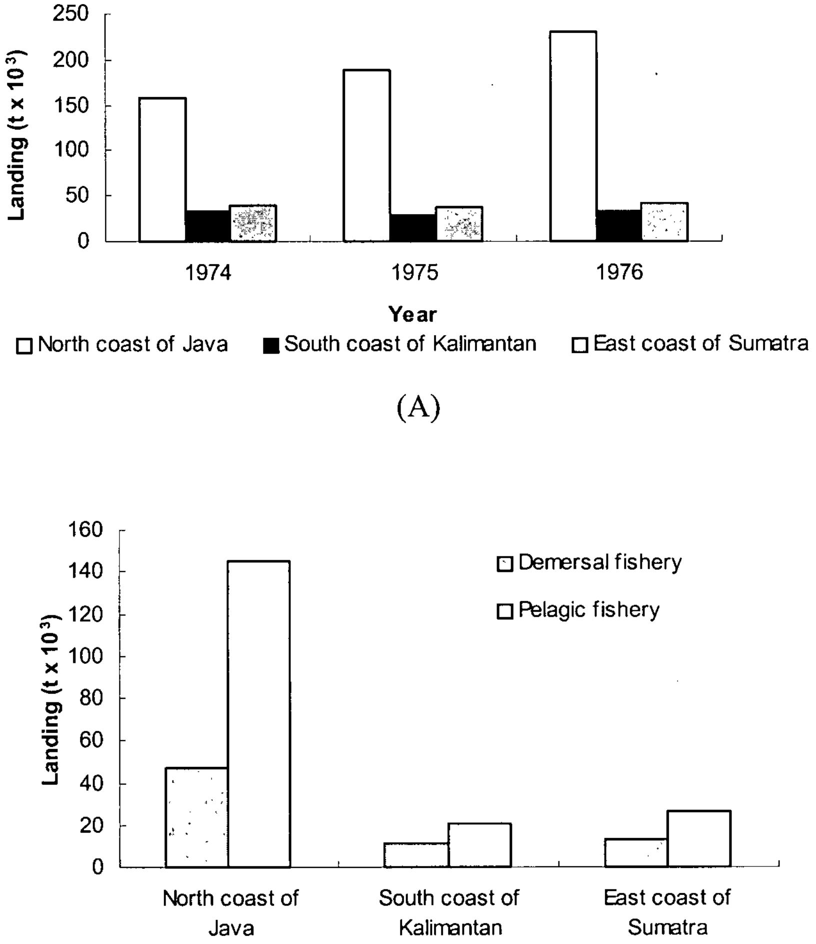


Figure 2.7 Penaeid prawns landings from the Java Sea, 1974 - 1994 (data from DGF, various years)

Reflecting the population imbalance between Java, Sumatra and Kalimantan islands (National Development Information Office, various years; Pauly 1989, Figure 2), the distribution of the bulk of catch that was landed from the Java Sea onto these three islands was very skewed (Figure 2.8). Moreover, there is a bias even within the Java Sea. Potier and Sadhotomo (1995) calculated that more than 68% of fishing gears in the Java Sea operated along the northern coast of Java, where 65% of the catch from the sea were landed.



(A)

(B)

Figure 2.8 Marine fisheries landing for the Java Sea in mid-1970s (data from DGF, various years): (A) by sub-area, and (B) by major type.

Figure 2.9 (left Y-axis) shows that the gears that operated in the study area were dominated by small scale gears. When records of trawls and purse seines are compared (Figure 2.9, right Y-axis), it is obvious that the trawlers' ban in 1980 had dramatically increased the number of purse seiners that operate in the study area. In fact, Bailey (1987) and Chong *et al.* (1987) indicated that a special credit program was established to encourage trawler owners to refit their vessels for use with other gears, most notably purse seines.

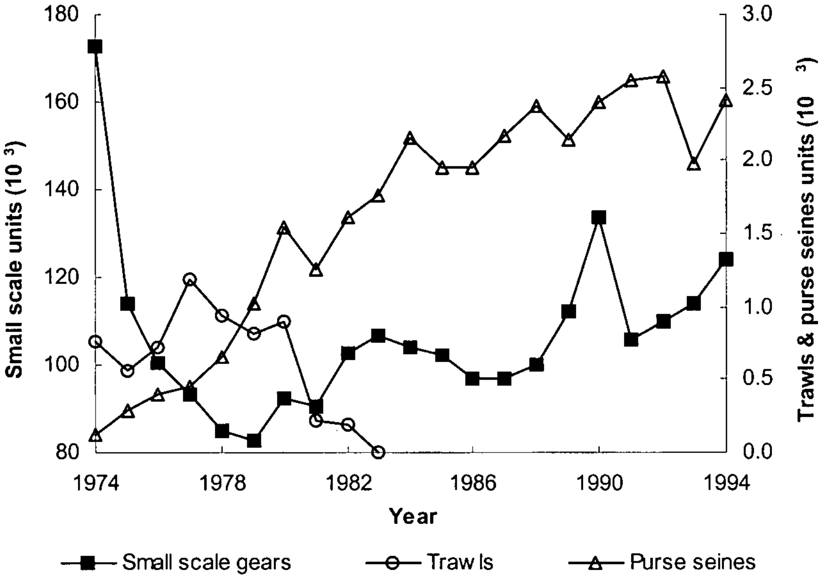


Figure 2.9 Number of gear units operating in the Java Sea, 1974 - 1994, by type. Small scale gears are plotted on the left Y-axis, while trawls and purse seines are plotted on the right Y-axis (data from DGF, various years).

In accordance with the landing records for gill nets (Figure 2.5), the number of gill nets in the Java Sea throughout 20 years of period also shows the greatest abundance among small scale gears (Figure 2.10). The DGF reports landings from four types of gill nets, i.e., from drift, fixed, encircling and shrimp gill nets. By far, the most numerous is the drift gill nets (both in monofilament and multifilament nylon), the use of which is widespread throughout Indonesia (Bailey and Dwiponggo 1987).

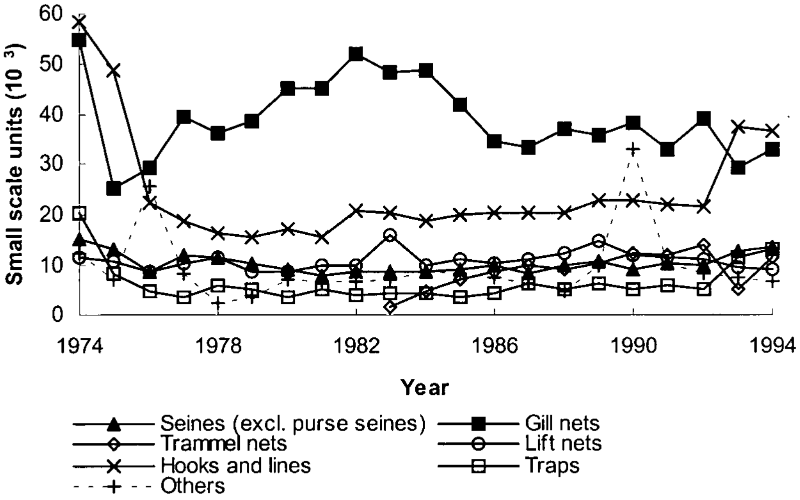


Figure 2.10 Small scale gears by type operating in the Java Sea from 1974 to 1994, illustrating the dominance of gill nets (data from DGF, various years).

Vessel statistics from the Java Sea from 1974 to 1994 showed a substantial increasing trend for outboard engine-powered vessels and a substantial decreasing trend for non-motorized vessels/crafts (Figure 2.11). This is mainly due to the introduction of government loan programs to assist small scale non-motorized fishers to adopt motorized way of fishing (Bailey 1987).

The loan programs emphasized the expanding adoption of inboard engine-powered vessels as well, especially for vessels in the range of 5 to 7 GT (Bailey 1987). However, high initial investment cost of inboard engines (Yamamoto 1978) shifted the loan concentration to outboard engines, which could be easily fitted into existing boats. An increase in inboard engine-powered vessels also took place, albeit the rate is slower and it is mostly dominated by vessels of less than 5 GT (Figure 2.12).

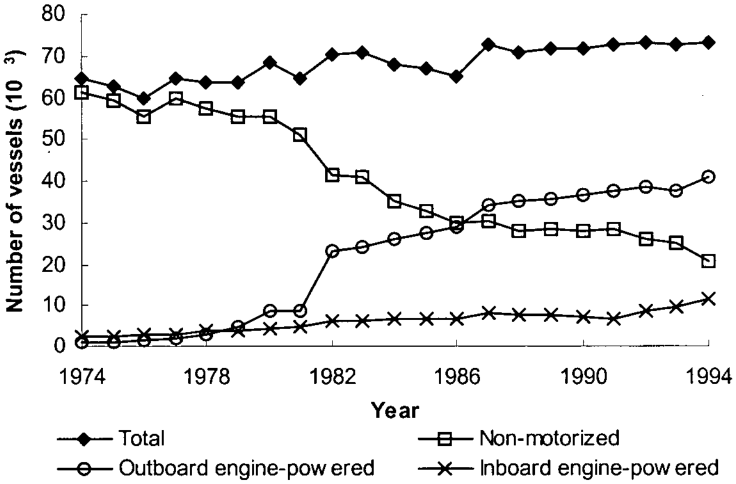


Figure 2.11 Number of vessels in the Java Sea, 1974 - 1994, by method of propulsion (data from DGF, various years).

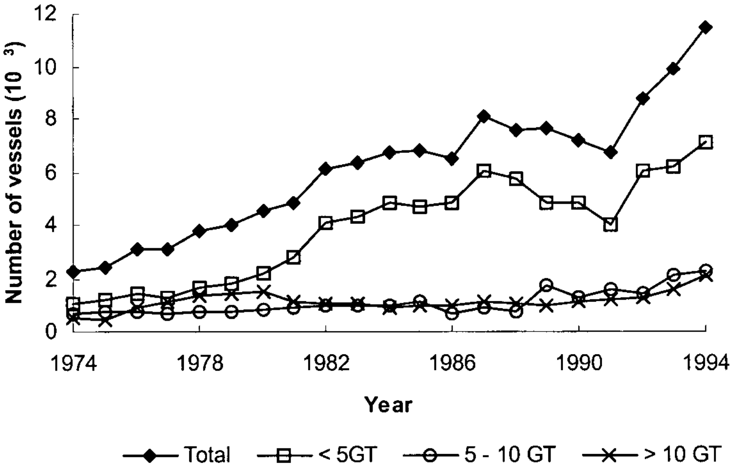


Figure 2.12 Inboard engine-powered vessels operating in the Java Sea, 1974 - 1994, by size class (data from DGF, various years).

The number of fishers operating in the Java Sea has also increased steadily over the period of 1976 to 1994 (Figure 2.13). During the same period, the average proportion of the Java Sea fishers was approximately 29% of total fishers throughout the archipelago (Figure 2.14).

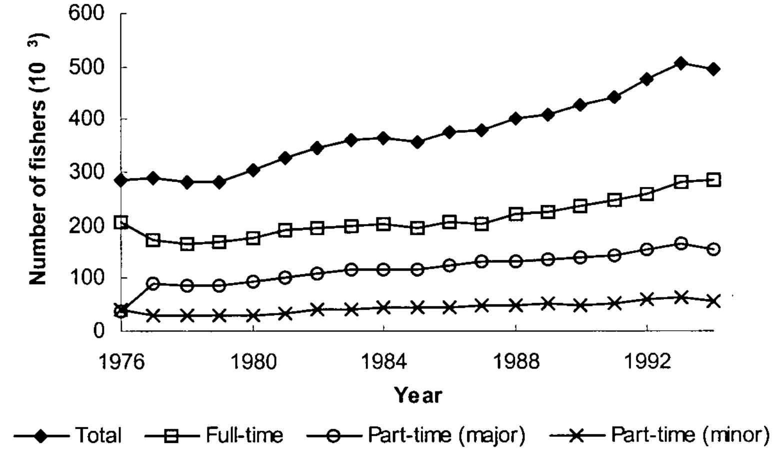


Figure 2.13 Number of fishers operating in the Java Sea, 1974 - 1994, by level of dependence on fishing (data from DGF, various years).

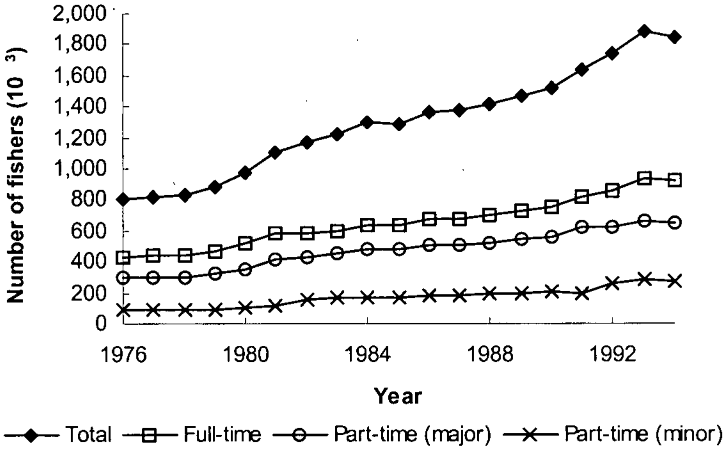


Figure 2.14 Number of fishers operating throughout the Indonesian archipelago, 1974 - 1994, by level of dependence on fishing (data from DGF, various years).

In 1994, there were almost 2 million fishers throughout the Indonesian archipelago (Figure 2.14). Data from the DGF (1996) showed that there were 477,000 fishers in the Java Sea alone in 1994. Of these fishers, 276,000 were full-time, 146,000 were part-time (major) and 55,000 were part-time (minor) fishers. This is almost double the number of fishers operating in 1976 (274,000 fishers), of whom there were 200,000 full-time, 34,000 part-time (major) and 40,000 part-time (minor) fishers (DGF 1978).

These increases, especially in the part-time categories, coupled with an increasing number of outboard engine-powered vessels, fit the causal patterns of Malthusian overfishing (Pauly 1997), as will be shown later. In Indonesia, easy entry into the small scale fisheries sector has been made possible by the low capital requirements to do the fishing with simple gears, or even gleaning without gear (Bailey 1988; Pauly 1997). Poverty, lack of alternative employment opportunities, population growth and rural disenfranchisement have increased the number of fishers. The increase arises from at least two mechanisms: internal recruitment and the immigration of destitute new entrants, driven off from their traditional occupations (Pauly 1989, 1997).

CHAPTER 3:

A SNAPSHOT MODEL OF THE JAVA SEA ECOSYSTEM IN THE MID-1970S

3.1 Constraints on data use

Data and information that are used to build this ecosystem model encompassed both published and hard-to-access unpublished reports, obtained from various sources. A more specific account of data sources for each group is given in section 3.2.

As much as possible, data from within the actual study area (*viz.,* the Java Sea) were used to build the Ecopath model. Primary sources here were the trawl survey data of the R/V *Mutiara IV*, conducted in the Java Sea and the southern tip of the South China Sea from November 1974 to July 1976 (Pauly and Martosubroto 1996; Torres *et al.* 1996). As well, various oceanographic surveys conducted by LON-LIPI[11] from 1977 to 1979, along the northern coast of Java were valuable. Data from the pre-ban period, when available, were used, especially from the period of 1974 to 1976, as this model focused on mid-1970s period.

In the event where such data or information were not available for certain functional groups, data or information from neighboring ecosystems were used. Geographically, these systems were selected in the following order of preference: South-East Asian region, tropical Indo-Pacific region, and the temperate Indo-Pacific region. In situations where no studies were available for the aforementioned, studies or models from more distal regions were consulted, but as a guide only.

Landing data from the Java Sea were obtained from the Directorate General of Fisheries (DGF) database of 1974 to 1994. Only data from the period of 1974 to 1976 were used to build the Ecopath model.

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[11] LON or *Lembaga Oseanologi Nasional* (National Council for Oceanology) is a national research institute focusing on marine resources, under the jurisdiction of LIPI or *Lembaga Ilmu Pengetahuan Indonesia* (Science Council of Indonesia). This name was later changed to 'P3O', which is an abbreviation of *Pusat Penelitian dan Pengembangan Oseanologi* (Research and Development Centre for Oceanology, RDCO).

Unless otherwise specified, all information on diet composition for each individual fish species was obtained from various studies on stomach content analysis compiled in the FishBase database (FishBase 1998). Information on diet composition for non-fish groups were derived from various models documented in Pauly and Christensen (1993) and other literature.

3.2 Defining the model components

3.2.1 Data sources

As mentioned, data were acquired from a wide variety of sources. Thus, it was necessary for this study to follow the following protocol. The initial step was cross-checking compilations of species records from the study area. Sources included: Dwiponggo *et al.* (1986), Martosubroto and Pauly (1976), Losse and Dwiponggo (1977), Schuster and Djajadiredja (1952), the FishBase[12] database (FishBase 1998), DGF landing statistics (Table 2.1), and the R/V *Mutiara IV* trawl survey data set for the period of November 1974 to July 1976 as encoded in NAN-SIS[13] database (Torres *et al.* 1996). The cross-checking revealed 214 species of fish alone recorded from the Java Sea.

Non-fish species in the Java Sea were identified from both direct and indirect sources. Direct sources included: (1) periodical oceanographic surveys conducted in the study area in 1970s to 1980s by the then LON-LIPI, (2) DGF fisheries statistics (Dwiponggo 1987), (3) research articles of the *Snellius II* Expedition (which covers mainly the eastern part of Indonesia, but also the eastern tip of the Java Sea), and (4) other literature about the study area. Indirect sources were comprised of: (1) cross-checking with literature from neighboring areas, and (2) comparing Ecopath models of neighboring ecosystems. Most notable among the latter were: the Gulf of Thailand model (Christensen 1998), the South China Sea models (Pauly and Christensen 1993), the Brunei coast model (Silvestre *et al.* 1993), the Bolinao reef model

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[12] FishBase is a scientific database (distributed on CD-ROM) on fish from around the world. It incorporates key data on various things of all important fishes, such as information on the biology, ecology, population dynamics, taxonomy, and even common names in different languages (Froese and Pauly 1998; see also www.fishbase.org).

[13] NAN-SIS is a micro-computer information system for logging, editing and analysis of scientific trawl surveys data, i.e., trawl-catch and length-frequency data (Strømme 1992).

of the Philippines (Pauly *et al.* 1993; Aliño *et al.* 1993), and to a much lesser extent, the Caribbean coral reef model (Opitz 1996).

3.2.2 Data analysis and model parameterization

Where several data sets were available, input parameters for the functional groups were calculated as arithmetic means, without weighting. The aggregation process for this model was performed based on similarities in habitat, body size, growth and mortality rates and diet composition, after the method proscribed by Christensen and Pauly (1996). Such information (notably for fish) was mainly obtained from the FishBase database (FishBase 1998).

Fish were allocated into functional groups first by habitat preference, then by body size. Size ranges were specified such that fish with an average or maximum body length of less than 40 cm, were considered to be 'small'. Fish with an average or maximum body length between 40 to 60 cm were considered as 'medium'; those that were greater than 60 cm were considered to be 'large'. For some species groups the allocation process was quite straightforward because of their significant contribution to landings (e.g., clupeoids, *Decapterus* spp., etc.) and/or in the trawl surveys (e.g., leiognathids, etc.). These formed functional groups of their own.

Some groups other than fish, however, presented problems. For instance, the marine mammal species proved difficult to handle. Initially, based on a literature review three functional groups were identified for marine mammals in the Java Sea: dugongs, resident dolphins, and transient whales (White 1983; Klinowska 1991; Jefferson *et al.* 1993; Tas'an and Leatherwood 1984; Rice 1989). Thus, during the process of data analysis and model parameterization, the following modifications were found necessary to balance the model:

(1) As an endangered species that receives very little attention in Indonesia, the population of dugong in the Java Sea is probably very small, though actual counts of this species in the study area do not exist. However, an account of the dugong's distribution, based on unstructured interviews with local people throughout its suspected distribution range, was conducted from late 1960s to late 1970s (Nishiwaki *et al.* 1979). Assuming that this data provides a relative index of abundance, an estimation of dugong biomass in

the Java Sea was obtained which is below the lowest value that can presently be accommodated in an Ecopath model, i.e., 0.0001 $t \cdot km^{-2}$. Hence, this group was not included in the model.

(2) Dolphins in the Java Sea are considered to be residents (see e.g., Tas'an and Leatherwood 1984). Whales, however, are transient in the system (Klinowska 1991; Jefferson *et al.* 1993; Rice 1989), migrating from the Indian Ocean, through the Java Sea and then to the Pacific Ocean during the winter to breed, and vice versa during the summer, mainly to feed.

Thus, dolphins and whales were initially separated into two functional groups. While balancing the model, however, unrealistic values for the transient whales occurred (e.g., respiration > assimilation). The problem was resolved by aggregating these two groups into a single marine mammal group.

Other taxa that were identified but not accommodated in the model were marine turtles and marine birds, even though their existence in the Java Sea is a matter of record.

Marine turtles in the Java Sea (White 1983) include the Hawksbill Turtles (*Eretmochelys imbricata*), and to a lesser extent, the Green Turtles (*Chelonia mydas*). No population estimates are presently available for either species in the study area, but both are considered to be endangered (Groombridge 1982). Hence, their population was assumed too small to incorporate into the model.

A similar rationale was followed for marine birds, based on the following evidence. White (1983) noted that there are 131 species of marine birds recorded in Indonesia. The majority of them are shorebirds and vagrant waders, which occur in coastal mudflats, marshes and mangroves. Additionally, oceanic seabirds usually form colonies on smaller islands away from the presence of humans. True marine birds are rare in the Java Sea. Consequently, this group was not included in the model.

To perform efficiently, and be both ecologically and biologically realistic, an Ecopath model should have at least a dozen functional groups (Christensen and Pauly 1996). As an upper limit, the Ecopath software presently can accommodate up to fifty functional groups (Christensen and Pauly 1992). The model constructed here is intermediate between these two extremes.

As outlined in Chapter 1, the Java Sea has a multitude of species. In order to have a tractable mass-balance model it was essential to aggregate species into several functional groups. Here, twenty-eight functional groups resulted from the aggregation process (Table 3.1). These 28 groups are comprised of one detritus group, two primary producer groups, eleven invertebrate groups, thirteen fish groups and one marine mammals group.

Table 3.1 Species composition for the 28 functional groups of the Java Sea model [a]

No.	Functional Group	Species included
1.	Benthic producers	Marine algae (epilithic algae, endolithic algae, reef turf algae, benthic fleshy algae, macroalgae, and benthic algae) and spermatophytes (sea grass)
2.	Phytoplankton	Dominated by diatoms (*Bacteriastrum* sp., *Biddulphia* sp., *Bacillaria* sp., *Chaetoceros* sp., *Coscinodiscus* sp., *Ditylum* sp., *Eucampia* sp., *Hemiaulus* sp., *Hemidiscus* sp., *Lauderia* sp., *Leptocylindricus* sp., *Nitzschia* sp., *Rhizosolenia* sp., *Skeletonema* sp., *Streptotheca* sp., *Thalassiothrix* sp./*Thalassionema* sp.) and dinoflagellates (*Ceratium* sp., *Dinophysis* sp., and *Peridinium* sp.)
3.	Small herbivorous zooplankton	Copepodes, ostracodes, bivalve larvae, cirripedia larvae, cladocerans, echinoderm larvae, larvacea, other mollusk larvae, and larvacea
4.	Large herbivorous zooplankton	Mysids, sergestids, euphausiids, amphipodes, luciferidae, and other decapod larvae
5.	Carnivorous zooplankton	Chaetognaths (*Sagitta* spp.), annelids and ichthyoplankton
6.	Jellyfish	All medusoid form of cnidarians (hydrozoa and scyphozoa only); however, scyphozoan medusae dominates
7.	Benthic infauna	Includes all burrowing benthos of the size ≤ 1.0 mm, *viz*., polychaetes, mollusks, echinoderms, crustaceans, sipunculans, and benthic stage larvae of other larger organisms
8.	Structure-associated fish (SAF)	This group is designed to include all biota (notably fish) that ecologically depend (directly and/or indirectly) on living bottom structure (LBS, see components of group #10) to complete their life cycle. For this model, it includes fish of the families Balistidae, demersal Carangidae, Lethrinidae, Lutjanidae, Pentapodidae, Platacidae, Polynemidae, Sparidae, Sphyraenidae, Scaridae, Acanthuridae, and Tetraodontidae
9.	Macrozoobenthos	This group includes all larger (size > 1.0 mm) mollusks and echinoderms, such as conch, oyster, scallops, clams, cockles, mussels, sea urchins, sea cucumbers, and sea stars
10.	Living bottom structure (LBS)	This functional group comprised of all biogenic organisms that live on and/or are attached to the sea floor, such as sponges (notably giant cup sponges, viz. *Poterion nautilus*, *P. neptuni* and *P. amphitritae*), gorgonians (sea fans and sea whips), soft corals, sea pens, sea anemones, etc.
11.	Juvenile penaeid shrimps	This group includes all juvenile shrimps, not just penaeid shrimps. However, the parameterization of this group is based on the juveniles of *Penaeus* spp. and *Metapenaeus* spp.
12.	Large pelagic predators (juvenile)	Juveniles of the families Chirocentridae, large Scombridae, and Trichiuridae

Table 3.1 Species composition for the 28 functional groups of the Java Sea model [a]

No.	Functional Group	Species included
13.	Adult penaeid shrimps	Includes all adult shrimps, but data refer only to *Penaeus* spp. and *Metapenaeus* spp.
14.	Miscellaneous pelagics	Belonidae, Carangidae, Harpodontidae, Hemirhamphidae, Lactaridae, Mobulidae, Sciaenidae (*Kathala axillaris*), and Scombridae (*Auxis* sp.)
15.	Leiognathids	*Gazza minuta*, *Leiognathus* spp., and *Secutor* spp.
16.	Crabs and lobsters	Portunidae, Palinuridae, Scyllaridae, etc.
17.	Cephalopods	Includes squids (*Loligo* spp.), cuttlefish (*Sepia* spp.) and octopus (*Octopus* spp.)
18.	*Decapterus* spp.	*Decapterus macrosoma, D. maruadsi* and *D. russelli*
19.	*Rastrelliger* spp.	*Rastrelliger brachysoma* and *R. kanagurta*
20.	Clupeoids	Clupeidae and Engraulidae
21.	Small demersals	Apogonidae, Ariidae, Ariommatidae, Caesionidae, Cynoglossidae, Dactylopteridae, Drepanidae, Ephippidae, Gerridae, Haemulidae, Holocentridae, Kurtidae, Menidae, Mullidae, Nemipteridae, Priacanthidae, Serranidae, Siganidae, Sillaginidae, Stromateidae, Synodontidae, Teraponidae and Sciaenidae
22.	Medium demersals	Haemulidae, Lutjanidae (*Pristipomoides typus*), Muraenesocidae (*Congresox talabon*), Nomeidae, Psettodidae, Rachycentridae, Scatophagidae, Serranidae, and Mugilidae
23.	Large demersal predators (juvenile)	Juveniles of the families Ariidae (*Arius thalassinus*), Carcharhinidae, Muraenesocidae, Serranidae (*Epinephelus lanceolatus*), Sphyrnidae, Stegostomidae, Pristidae, Rhinidae, and Rhinobatidae
24.	Demersal rays	Dasyatidae and Myliobatidae
25.	Large pelagic predators (adult)	Chirocentridae, large Scombridae, and Trichiuridae
26.	Large demersal predators (adult)	Ariidae (*Arius thalassinus*), Carcharhinidae, Muraenesocidae, Serranidae (*Epinephelus lanceolatus*), Sphyrnidae, Stegostomidae, Pristidae, Rhinidae, and Rhinobatidae
27.	Marine mammals	Comprised of resident dolphins *viz.*, Bottlenose dolphins (*Tursiops* spp), Spinner dolphins (*Stenella longirostris*), Irrawady dolphins (*Orcaella brevirostris*), Finless porpoises (*Neophocaena phocaenoides*), and Indo-Pacific Hump-backed dolphins (*Sousa chinensis*); and transient whales, *viz.*, Sperm whales (*Physeter catodon*), Bryde's whales (*Balaenoptera edenii*), and Minke whales (*B. acutorostrata*)
28.	Detritus	Comprised of particulate and dissolved organic matters

[a] Fish groups are only listed as families. Complete species list can be found in Froese *et al.* (1996) and FishBase (Froese and Pauly 1998).

3.3 Model inputs and balancing the model

Unless otherwise stated, the unassimilated food fractions were set to default value of 0.2 (Winberg 1956) for all groups, assuming that 80% of the consumption is physiologically useful while the 20% unassimilated part (consisting of urine and faeces) is directed to the detritus. Winberg's 20% estimate was found to be generally accurate for predatory fishes (Brett and Groves 1979).

3.3.1 Primary producers

In this model, there were two functional groups of primary producers: phytoplankton and benthic producers.

3.3.1.1 Phytoplankton

LON-LIPI conducted an oceanographic monitoring survey in the northern coast of Java, aboard the research vessel KM Samudera in December 18 - 31, 1979 (LON-LIPI 1980). Phytoplankton was one of the components that was monitored. For the resulting species list, see Table 3.1.

The average phytoplankton biomass that was calculated from the monitoring data was very low (0.218 t·km^{-2})[14] compared to the Gulf of Thailand 10-50m sub-model (18.25 t·km^{-2}), the Brunei coast model (12.825 t·km^{-2}), and even the Bolinao coral reef ecosystem (0.3 t·km^{-2}, Aliño *et al.*. 1993). Since there was no justification to select a value from any other model, the biomass input parameter for phytoplankton functional group was left empty. An EE value of 0.95 was entered, which assumes nearly total exploitation of the phytoplankton biomass, and the Ecopath software was allowed to calculate the required biomass value.

Values for the P/B ratio of phytoplankton in the Java Sea were not available. Hence, an estimate was obtained by averaging P/B ratios from the Gulf of Thailand 10-50 m sub-model, which was 200.0 year^{-1} (Pauly and Christensen 1993); and from the Brunei coast

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[14] The survey (LON LIPI 1980) used a vertical towing method. Pauly *et al.* (1996), who collected zooplankton samples in the study area using a similar method indicated that this method is not effective. Thus, the low biomass. Instead, oblique towing should have been used.

model, which was 71.2 $year^{-1}$ (Silvestre *et al.* 1993), giving an averaged P/B ratio of 135.0 $year^{-1}$.

3.3.1.2 Benthic producers

As explained in Table 3.1, benthic producers functional group consists of marine algae and spermatophytes. Studies on marine algal (i.e., seaweeds) and spermatophyte (i.e., seagrass) species in the study area are rare, and focused on taxonomic issues. Therefore, the input parameters for this functional group was extracted from the South China Sea model, following the same rationale as Pauly and Christensen (1993). Thus, a biomass of 153.0 $t{\cdot}km^{-2}$ was adopted. The P/B value for benthic producers was derived similarly, i.e., from the average of the P/B values of seaweeds and seagrass in the South China Sea model (Pauly and Christensen 1993). The resulting value was 11.885 $year^{-1}$.

Landings of benthic producers in the Java Sea ecosystem consists mainly of seaweeds, including *Euchema* spp. and *Gracilaria* spp. However, for the period of 1974 to 1976, the landings of seaweeds (DGF 1976, 1977, 1978) from the study area were too low to be inputted in the Ecopath software (i.e., 0.0002 $t.km^{-2}$). Hence, the landing value was here considered to be nil.

3.3.2 Detritus group

The only input parameter needed for this functional group is the biomass. Detritus biomass may be estimated empirically for any system, given simple data such as the amount of primary production and the euphotic depth (Pauly *et al.* 1993). Data on the euphotic depth was not available, however, so the detritus biomass from the Brunei coast model (Silvestre *et al.* 1993), *viz.,* 120.0 $t{\cdot}km^{-2}$ was used here. In this model, the fate of detritus generated by all functional groups was to be sent to a single detritus group.

3.3.3 Invertebrate groups

There are eleven functional groups under this category, detailed as follows:

3.3.3.1 Zooplankton and jellyfish

For the purpose of this model, the zooplankton population was segregated into three functional groups based on the body size and their diet composition. The three groups were small herbivorous zooplankton, large herbivorous zooplankton and carnivorous zooplankton (Table 3.1). Though very different from other planktonic groups, jellyfish are included here, as the values for this group were obtained from the same sources and in the same manner as for the other zooplankton groups.

A total zooplankton biomass value was obtained from a survey conducted by the R/V *Mutiara IV* from November 1974 to July 1976 in the southern tip of South China Sea (Pauly *et al.* 1996). The average biomass obtained was 0.123 g·m^{-3}, or 3.40 t·km^{-2} (Table 3.2).

The total biomass estimate was then allocated to the zooplankton groups, using the frequency of occurrence (Table 3.3) obtained from the monitoring survey in the study area (LON-LIPI 1980). The final model inputs are listed in Table 3.3.

Table 3.2 Observations on zooplankton biomass by the R/V *Mutiara IV* survey in area 5, South China Sea, November 1974 to July 1976 (modified from Pauly *et al.* 1996).

Station	**Depth (m)**	**Water filtered (m^3)**	**Biomass (g·m^{-3})**	**Biomass (t·km^{-2})**
38	30	179	0.27	8.1
44	40	327	0.14	5.6
50	38	251	0.10	3.8
56	26	187	0.10	2.6
62	22	200	0.08	1.8
68	16	90	0.15	2.4
75	30	46	0.17	5.1
82	20	60	0.15	3.0
88	32	103	0.06	1.9
94	46	72	0.05	2.3
100	20	64	0.13	2.6
104	20	50	0.08	1.6
		Total biomass =	1.48 g·m^{-3}	40.8 t·km^{-2}
		Ave. biomass =	**0.123 g·m^{-3}**	**3.40 t·km^{-2}**

Table 3.3 Final biomass values for the three zooplankton groups and the jellyfish group. Allocation was based on frequency of occurrence data (LON-LIPI 1980) and biomass data from Pauly *et al.* (1996).

No.	Functional groups	Frequency of occurrence (%)	Biomass distribution (t·km^{-2})
1.	Small herbivorous zooplankton	71.37	2.43
2.	Large herbivorous zooplankton	16.62	0.56
3.	Carnivorous zooplankton	9.12	0.31
4.	Jellyfish	2.89	0.10
	TOTAL =	**100**	**3.40**

In the absence of studies on P/B and Q/B of zooplankton and jellyfish in the study area, both P/B and Q/B values for all zooplankton and jellyfish functional groups were obtained from other ecosystem models and other studies.

P/B ratio for small herbivorous zooplankton of 60.225 year^{-1} was derived from studies on warm water small herbivorous copepods in the Coral Sea (Greze 1978).

Q/B ratio for small herbivorous zooplankton was taken over from an ecosystem model of the Monterey Bay (Olivieri *et al.* 1993), notably from the Q/Bs of mesozooplankton (i.e., 140 year^{-1}) and microzooplankton (i.e., 300 year^{-1}) groups, which resulted an averaged Q/B ratio of 220 year^{-1}. Wood (1987) indicated that mesoplankton is comprised of plankton with the size of 0.2 - 20 mm, and mostly consists of copepods; while microplankton is comprised of plankton with the size of 0.02 - 0.2 mm, and mostly consists of phytoplankton, protozoans and the smallest metazoans. The small herbivorous zooplankton group of the Java Sea consists of similar organisms.

The P/B ratio of large herbivorous zooplankton was derived from the P/B ratio of macrozooplankton in the Monterey Bay model (Olivieri *et al.* 1993), i.e., 25 year^{-1} and from studies on warm water large copepods in the Gulf of Guinea (Malovitskaya 1971), i.e., 18.25 year^{-1}, which resulted an averaged P/B of 21.625 year^{-1}. As indicated by Wood (1987), macroplankton is comprised of plankton with the size of 2 - 20 cm, and notably euphausiids.

The large herbivorous zooplankton of the Java Sea consist of similar organisms (Table 3.1). During the balancing process of the model it was necessary to slightly reduce the P/B ratio, to 20 year^{-1}.

The Q/B value for large herbivorous zooplankton was adopted directly from Olivieri *et al.*. (1993), in particular from the macrozooplankton group (i.e., 70 year^{-1}).

For the carnivorous zooplankton group, the P/B ratio from was derived from P/B values estimated in studies on chaetognaths and carnivorous copepods in the Coral Sea (Greze 1978) and the equatorial Pacific (Shushkina and Kisliakov 1975), resulting an averaged P/B ratio of 42.58 year^{-1}.

The Q/B ratio of carnivorous zooplankton was adopted from a consumption study on *Sagitta nagae* in Suruga Bay, Central Japan (Nogasawa and Marumo 1972). This study found that *S. nagae* consume 37% of its own body weight each day in prey, resulting a Q/B ratio of 135.05 year^{-1}.

The P/B and Q/B ratios of jellyfish were both derived from the Gulf of Thailand 10-50m sub-model (Pauly and Christensen 1993), and were 5.011 year^{-1} and 25.050 year^{-1}, respectively.

Of all zooplankton groups in the Java Sea, only large herbivorous zooplankton that is caught commercially, i.e., mysids and sergestids. The fisheries landing statistics of mysids and sergestids from the study area, averaged from 1974 to 1976 landing data yielded catch estimate of 0.008 $\text{t}\cdot\text{km}^{-2}\cdot\text{year}^{-1}$ (DGF 1976, 1977, 1978). As for other zooplankton groups (i.e., small herbivorous and carnivorous zooplanktons), landings were here considered to be nil.

Catches of jellyfish were derived from fisheries landing statistics in the area, averaged from 1974 to 1976 landing data, i.e., 0.003 $\text{t}\cdot\text{km}^{-2}\cdot\text{year}^{-1}$ (DGF 1976, 1977, 1978).

Diet compositions for all zooplankton functional groups were derived from the Brunei coast (Silvestre *et al.* 1993) and the Gulf of Thailand (Christensen and Pauly 1993) models. Some adjustments were made when balancing the model. Information on diet composition of jellyfish was obtained from a general invertebrate zoology text book (Barnes 1982). Again, some modification was necessary during the balancing process.

Following Christensen and Pauly (1992a), the unassimilated food fraction parameter was set to 0.4, both for small and large herbivorous zooplankton, on the basis that herbivorous animals leave a high proportion of their food unassimilated.

3.3.3.2 Benthic infauna

Benthic infauna in the Java Sea was studied exhaustively by LON-LIPI, both pre-ban and post-ban along the northern coast of Java, and post-ban along the southern coast of Kalimantan; especially at near shore areas (LON-LIPI 1977a, 1977b, 1978a, 1978b, 1980, 1981a, 1981b, 1981c, 1981d, 1982a, 1982b, 1983a, 1983b; Kastoro *et al.* 1990). A study was also conducted during the post-ban period (as part of *Snellius II* Expedition) that focused on areas in the middle of the Java Sea and around the islands of Madura and Bali (de Wilde *et al.* 1990). For this model, only pre-ban data were used, notably from surveys along the northern coast of Java on December 19-31, 1979 (LON-LIPI 1980), and from surveys in Jakarta Bay on January 26-29, 1977 (LON-LIPI 1977a), August 10-13, 1977 (LON-LIPI 1977b), November 26-29, 1977 (LON-LIPI 1978a), and January 17-20, 1978 (LON-LIPI 1978b). No surveys were conducted along the southern coast of Kalimantan during the pre-ban period.

All surveys used similar sampling gear, sorting and analysis methods.. There were, however, some minor differences such as the size of the Van Veen grab used. As well, an interview with Ms. Widiarsih Kastoro of LON-LIPI, lead scientist for the surveys, revealed that the oven used in the 1970s was an old model and had different specification. Thus, unable to obtain the ash-free dry weight (AFDW). Ms. Kastoro suggested some unit conversions as outlined by Lie (1968) in order to convert non-ash free dry weight ($g \cdot m^{-2}$) to ash-free dry weight ($g \cdot m^{-2}$ AFDW). The corrected values were then converted to wet weight ($t \cdot km^{-2}$), for biomass input parameters.

The calculated biomass for benthic infauna group from LON-LIPI's surveys was high (34.48 t·km^{-2}). The value is biased upward, since all sampling was conducted in inshore and near shore areas. These areas consist of primarily muddy habitat, capable of supporting high populations of benthic infauna, but not reflective of the Java Sea as a whole. Therefore, an average value derived from both the LON-LIPI's value (i.e., 34.48 t·km^{-2}) and a value obtained from an observation on zoobenthos in the southern tip of South China Sea (Pauly *et al.* 1996) were used. The latter survey used similar type of sampling gear, sorting and analytical methods. The mean value thus obtained for the benthic infauna functional group was 18.94 t·km^{-2} (Table 3.4).

Table 3.4 Average biomass for benthic infauna, based on LON-LIPI surveys in 1977, 1978 and 1979 in the Java Sea inshores (LON-LIPI 1977a, 1977b, 1978a, 1978b, 1980), and R/V *Mutiara IV* survey in South China Sea southern tip and the Java Sea offshore, from November 1974 to July 1976 (Pauly *et al.* 1996).

No.	**Taxa**	**LON-LIPI surveys (Java Sea inshores) [t·km-2]**	**R/V Mutiara IV survey (SCS southern tip /Java Sea offshore) [t·km-2]**	**Ave. biomass [t·km^{-2}]**
1.	Polychaetes	17.74	2.91	10.33
2.	Mollusca	4.36	0	2.18
3.	Echinodermata	4.82	0	2.41
4.	Crustacea	7.56	0.47	4.02
	TOTAL =	**34.48**	**3.38**	**18.94**

Studies identifying P/B ratios of benthic infauna in the Java Sea were not available. Hence, a derived value from comparative studies of P/B, based on organisms from the orders Polychaeta, Lamellibranchia and Echinodermata (Greze and Kinne 1978, Table 3.4) was used (i.e., 6.57 year^{-1}). The study consisted of a meta-analysis on populations from tropical and temperate regions, and provided lower and upper limits of P/B ratios. For this study, the upper limit of values were selected, assuming they were more representative of a tropical environment.

The Q/B input parameter for benthic infauna was initially obtained from energy demand studies of benthic infauna conducted in the Java Sea (de Wilde *et al.* 1989). This study indicated that the actual carbon demand (from detritus) of the benthic system of the Java Sea was 38 $gC \cdot m^{-2} \cdot year^{-1}$. Using a conversion factor also used by Opitz (1996) for detritus, a conversion (of carbon to wet weight detritus) of 10%, yielded a Q/B ratio of 20.063 $year^{-1}$ for benthic infauna with a biomass of 18.94 $t \cdot km^{-2}$. Estimates of the Q/B ratio for benthic infauna species exist for the Java Sea (see de Wilde *et al.* 1989), but yielded unrealistic GE (gross efficiency) values during the model balancing. Hence, a Q/B ratio was taken from the benthos group of the Gulf of Thailand 10-50m sub-model (Pauly and Christensen 1993). The value thus obtained was 27.40 $year^{-1}$. There are no fisheries catches for this group.

No data on diet composition of benthic infauna were available for the study area. Therefore, the diet composition information from the Gulf of Thailand 10-50m sub-model (Pauly and Christensen 1993) was adopted. Some modifications during balancing were required.

3.3.3.3 Macrozoobenthos

Macrozoobenthos functional group in this model comprised of all epibenthic organisms with body sizes > 1.0 mm (see Table 3.1 for species composition). Studies on the macrozoobenthos in the Java Sea related to parameters needed for Ecopath were not found. Most studies in the area (or nearby areas) focused on either taxonomic aspects, general ecological aspects (i.e., distribution and numerical abundance) or simply studies on ecological indices (i.e., similarity indices, dominance indices, etc.). As a consequence, input values, except for the biomass, diet composition and harvest, were adopted from the Brunei coast model (Silvestre *et al.* 1993). Biomass was calculated by the Ecopath software, with EE pre-set at 0.75. The P/B ratio of heterotrophic benthos in the Brunei coast model was 3.00 $year^{-1}$. Similarly, the Q/B ratio for macrozoobenthos group adopted was 12.50 $year^{-1}$. Information on diet composition was adopted from the Brunei coast model and a coastal ecosystem model in southwestern Gulf of Mexico (Arreguín-Sánchez *et al.* 1993). Some modifications were made during balancing.

The landing data for macrozoobenthos were derived from landing statistics of cupped oysters (*Crassostrea* spp.), scallops (*Amusium* spp.), clams (*Meretrix* spp.), blood cockles (*Anadara* spp.), and sea cucumber (Holothuroidea). All landings were from the period of 1974 to 1976, from eight provinces and one municipality that caught these animals in the Java Sea (DGF, 1976, 1977, 1978). The fisheries catches of macrozoobenthos group is 0.001 $t \cdot km^{-2} \cdot year^{-1}$.

3.3.3.4 Living bottom structures (LBS)

This functional group was created to evaluate the impact of trawling on ecosystem structure and behavior (see simulation results in Chapter 4). LBS are large and sessile, tend to be slow growing and slow to re-colonize. All these properties increase their vulnerability to trawling (Nybakken 1982; Jones 1992; Sainsbury *et al.* 1997).

Studies on LBS in the Java Sea are rare. One of the few literature that examined this group in the study area is Harting (1870) and indicated two species of giant goblet-shaped sponges (i.e., *Poterion neptuni* and *P. amphitritae*). Most recent literature on sponges based on studies from the 1980s and 1990s, only covered species of much smaller size (Van Soest 1989; Amir 1991, 1992; Amir and Budiyanto 1996).

The biomass for this group was derived from a survey in the mid-1970s (Pauly *et al.* 1996). All sponges in the survey (tentatively called *Poterion nautilus*, at that time) were counted and weighed (average 40 kg/sponge). Further, their distribution from the survey data was mapped by Losse and Dwiponggo (1977). Digitizing the sponges distribution map (Losse and Dwiponggo 1977) and applying the average weight (Pauly *et al.* 1996) led to a calculated sponges biomass value of 0.775 $t \cdot km^{-2}$. However, since this group represents all LBS, and not just sponges, the biomass was left empty. The Ecopath software was allowed to calculate the biomass, with the EE pre-set to 0.95. The resulting biomass was much higher (19.88 $t \cdot km^{-2}$). This, was then rounded up to 20 $t \cdot km^{-2}$, and EE was, instead, left for Ecopath to estimate.

The P/B and Q/B ratios for LBS group were initially set to values used by Opitz (1996) for the sponges of her Caribbean coral reef model (1.70 $year^{-1}$ and 4.015 $year^{-1}$, respectively). Some modifications to these values were made during the model balancing process in order to ensure a low recovery rate to trawl impacts. The P/B and Q/B ratios were then set to 0.10 $year^{-1}$ and 0.50 $year^{-1}$, respectively.

Allocating a 'catch' to LBS was important for impact simulations (Chapter 4). In trawl fisheries, LBS are a major component of by-catch and discards (e.g., Sainsbury 1991; Sainsbury *et al.* 1997). However, no such data are routinely collected by the DGF.

Nonetheless, fish (i.e., 0.01 t·km^{-2}·year^{-1}) that depend ecologically on LBS (see section 3.3.4.3 below, i.e., structure-associated fish or SAF) do occur in landing data for the period of 1974 to 1976 in the Java Sea (DGF 1976, 1977, 1978). Assuming an LBS by-catch produced by trawling activities in the Java Sea that is proportional to the landings of dependent species, the catch of LBS was set at ten times the landing of SAF. This resulting an LBS landing of 0.1 t·km^{-2}·year^{-1}.

Diet composition was adopted from those for sponges, soft corals, gorgonacea (sea fans and sea whips), and sea anemones of the Caribbean coral reef model (Opitz 1996). Some modifications were made during the balancing process.

3.3.3.5 Juvenile and adult penaeid shrimps

Due to strong trophic changes of diet composition during their ontogeny, the penaeid shrimps group was divided into two, i.e., juveniles and adults pools. For this model, the juvenile group consisted of epibenthic post larvae and juvenile post larvae, roughly between 2 to 4 months old. The adult group consists of both sub-adult and adult shrimps.

These functional groups implicitly include all taxa of shrimps. However, because of both the limited information available for other taxa and the abundance of penaeid shrimps in the landings of the shrimp fishery (Chapter 2), input parameters are based on information for penaeid shrimps only.

No biomass estimates were available for any shrimp taxa, thus this was left to the Ecopath software to calculate, with EE pre-set at 0.95.

The P/B ratio for the adult penaeid shrimps group was obtained by an average of two sources. Assuming that the annual P/B ratio is equal to the annual instantaneous total mortality rates (Z), under equilibrium conditions and assuming the von Bertalanffy growth function (Allen 1971), the P/B ratio for adult penaeid shrimps was then derived as follows:

(1). from catch curve analysis of (adult) penaeid shrimp in the Java Sea (Chullasorn and Martosubroto 1986), giving $Z = 5.98$ year^{-1} ($M = 3.16$ year^{-1});

(2). from an estimation that incorporate a pre-ban demersal fishing mortality[15] ($F_{demersal}$) of 2.44 year^{-1} and an averaged natural mortality (M) of 2.17 year^{-1} (Table 3.5) for adult penaeid shrimps, giving Z = 4.61 year^{-1}.

Table 3.5 Calculation of natural mortality (M) for adult penaeid shrimps based on the empirical formula of Pauly (1980), using the K[a] and TL_{∞}[b] data of various *Penaeus* spp. and *Metapenaeus* spp. used in the Bolinao reef ecosystem model (Pauly *et al.* 1993).

No.	Species	TL_{∞} (cm)	K (year^{-1})	M (year^{-1})
1.	*Penaeus duorarum* (m)[c]	17.6	1.45	2.69
2.	*P. duorarum*	17.6	1.20	2.37
3.	*P. kerathurus* (f)[d]	21.0	0.80	1.73
4.	*P. kerathurus* (m)[c]	18.0	0.90	1.95
5.	*P. setiferus* (f)[d]	22.5	1.25	2.28
6.	*P. setiferus* (m)[c]	19.2	1.55	2.74
7.	*P. monodon*	35.0	0.35	0.87
8.	*Metapenaeus brevicornis*	13.3	0.93	2.17
9	*M. brevicornis*	14.2	0.90	2.09
10	*M. affinis*	17.5	1.20	2.38
11	*M. kutchensis*	14.0	1.15	2.46
12	*M. kutchensis*	14.0	1.20	2.53
13	*M. kutchensis*	13.5	1.05	2.34
14	*M. kutchensis*	13.8	1.10	2.40
15	*M. monoceros*	18.0	0.67	1.61
			Averaged M (year^{-1}) =	**2.17**

[a] curvature parameter of the von Bertalanffy growth model; [b] asymptotic length, i.e., the mean length of the shrimp of a given stock would reach if they were to grow forever; [c] male; [d] female.

The final P/B ratio for the adult penaeid shrimp functional group was the rounded median of the two Z values obtained (5.98 year^{-1} and 4.61 year^{-1}, respectively), resulting a Z or P/B ratio of 5.00 year^{-1}.

The Q/B ratio for the adult penaeid shrimps group was derived from the daily food consumption of *Penaeus monodon* and *Metapenaeus monoceros*, the primary species used for the Bolinao coral reef ecosystem model (Pauly *et al.* 1993), i.e., 4.48% BWD (percent of

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[15] Based on various catch curve analysis of demersal resources provided in Dwiponggo *et al.* (1986), a median value of pre-ban demersal fishing mortality was derived ($F_{demersal}$ = 2.44 year^{-1}). Pelagic fishing mortality was estimated to be half of the $F_{demersal}$ (D. Pauly, UBC Fisheries Centre, *pers. comm.* 1997). Thus, pre-ban $F_{pelagic}$ was 1.22 year^{-1}.

body weight per day) and 11.38% BWD, respectively. Hence, their Q/B ratios were 16.352 year^{-1} and 41.537 year^{-1}, respectively. Therefore, the averaged Q/B ratio for adult penaeid shrimps was 28.945 year^{-1}.

Studies on the P/B ratio for juvenile penaeid shrimps in the study area could not be found. Therefore, the P/B ratio for juvenile penaeid shrimps was derived from other studies, notably from Dall *et al.* (1990). The value calculated (P/B = 13.0 year^{-1}) assumes a low F, supported by landing statistics, and therefore M is the only component of Z.

The Q/B ratio for the juvenile group was adopted directly from the Q/B ratio of large herbivorous zooplankton, i.e., 70.00 year^{-1}. Regardless of the lower P/B ratio, it is assumed that juvenile shrimps consumed about the same amount of food per unit biomass as the large herbivorous zooplankton.

Landing data for adult penaeid shrimps were obtained from the landing statistics of all shrimps in the study area from 1974 to 1976 (DGF 1976, 1977, 1978). These landings totaled to 0.014 t·km^{-2}·year^{-1}. As for the juvenile group, no actual landing data exist, even though anecdotal information indicated that juvenile shrimps were caught by artisanal fishers as seed supply for traditional brackish water shrimp farming. Consequently, an assumption that the landing of this juvenile shrimps was equivalent to that of large herbivorous zooplankton was made. This led to 0.008 t·km^{-2}·year^{-1}.

Information on diet composition for both groups were derived from different sources as a guide (Chong and Sasekumar 1981; Wassenberg and Hill 1987; Su and Liao 1981). Some modifications were required during balancing.

3.3.3.6 Crabs and lobsters

Studies on crabs and lobsters in the study area related to information needed for Ecopath input parameters were not available. Hence, input parameters, except for the biomass, were derived from other ecosystem models and studies.

The R/V *Mutiara IV* trawl survey in the Java Sea from November 1974 to July 1976 (Torres *et al.* 1996) provided an estimation of biomass for lobster (0.0029 t.km^{-2}) but not for crabs. Doubling this value, assuming that the crab biomass would be at least similar to that of the lobsters was not sufficient to balance the model. Therefore, the Ecopath software was allowed to estimate the biomass, with EE pre-set at 0.95.

The P/B and Q/B ratios, were adopted from the large crustacean group of the Brunei coast model (Silvestre *et al.* 1993). Values were 4.00 year^{-1} and 21.90 year^{-1}, respectively.

The landing parameter for crabs and lobsters, was derived from the combined yearly average landing statistics of crabs (mostly *Portunus pelagicus* and *Scylla serrata*) and lobsters (mostly *Thenus* spp.) of the Java Sea from the period of 1974 to 1976 (DGF 1976, 1977, 1978).The average value thus obtained was 0.001 t·km^{-2}·year^{-1}.

Diet composition was adopted from the Brunei coast model (Silvestre *et al.* 1993) and to a much lesser extent, the coastal ecosystem model of the southwestern Gulf of Mexico (Arreguín-Sánchez *et al.* 1993). Some modification was necessary during the balancing process.

3.3.3.7 Cephalopods

Unfortunately, there are very little studies about cephalopods in the Java Sea, especially with regards to parameters required for Ecopath modelling. Hence, all information needed for the model was adopted from other models except for the biomass and the landing.

Although a biomass estimate of cephalopods from R/V *Mutiara IV* survey (Torres *et al.* 1996) was available (0.0863 t·km^{-2}), it was too low to generate the required production. Therefore, the Ecopath software was allowed to calculated the biomass, with EE pre-set at 0.95.

The P/B ratio (3.10 year^{-1}) for this functional group was derived from the cephalopods group of the Gulf of Thailand 10 - 50m sub-model (Pauly and Christensen 1993). The Q/B ratio (20.318 year^{-1}) for the cephalopod functional group was derived from the averaged Q/B ratios of *Loligo duvauceli*, i.e., 34.31 year^{-1}, and *Sepoiteuthis lessoniana*, i.e., 19.345 year^{-1} (Longhurst and Pauly 1987), and the Q/B ratio of *Octopus vulgaris*, i.e., 7.3 year^{-1} (Pauly *et al.* 1993).

The landing (0.005 t·km^{-2}·year^{-1}) of cephalopods was derived from combined landing statistics of squids, cuttlefish and octopus from the Java Sea, during the year of 1974 to 1976 (DGF 1976, 1977, 1978).

The diet composition was derived from a comprehensive study on the role of cephalopods in the marine ecosystem (Amaratunga 1983). Some modifications were necessary during the balancing process.

3.3.4 Fish groups

The fish groups were partitioned into three main groups, i.e., the demersals, the pelagics and the structure-associated fish (SAF). For species composition of these groups, see Table 3.1.

3.3.4.1 Demersal groups

Biomass input parameters (Table 3.6) for all demersal groups (except for the demersal rays and juvenile large demersal predators) were obtained from the R/V *Mutiara IV* trawl survey in the study area from November 1974 to July 1976 (Torres *et al.* 1996).

The Ecopath software was allowed to calculate the biomass for the juvenile large demersal predators and the demersal rays functional groups. The EE values were pre-set as 0.5 and 0.6 respectively: the juvenile large demersal predators were assumed to be exposed to relatively little exploitation because they are top predators in the system. Hence, the relatively low value of 0.50 set for its EE. As for demersal rays, it was assumed that this functional group was subject to only moderate predation. Inferring from the landing statistics, fishing pressure for demersal rays was also assumed to be moderate. Hence, the EE was set at 0.60.

Table 3.6 Input parameters for all demersal fish functional groups in the mid-1970s model of the Java Sea (see sources in the text)

No.	Functional Groups	Biomass ($t \cdot km^{-2}$)	P/B ($year^{-1}$)	Q/B ($year^{-1}$)	Catch ($t \cdot km^{-2} \cdot year^{-1}$)	EE	GE
1.	Leiognathids	0.193	?	18.37	0.019	?	0.2
2.	Small demersals	1.687	?	12.84	0.066	?	0.2
3.	Medium demersals	0.046	?	9.14	0.015	?	0.2
4.	Demersal rays	?	1.3	8.2	0.006	0.6	?
5.	Juvenile large demersal predators	?	?	12.84	0.066	0.5	0.3
6.	Adult large demersal predators	0.13	?	6.13	0.02	?	0.15

? = To be estimated by Ecopath

The P/B and Q/B values for the six groups are also given in Table 3.6. Initially, the P/B ratios were calculated in the same manner as those for penaeid shrimps (see section 3.3.3.5). In this case the applied F ($F_{demersal}$, 2.44 $year^{-1}$) was the median pre-ban value, derived from catch curve analysis (Dwiponggo *et al.* 1986), and M was calculated using an empirical formula (Pauly 1980) and other biological parameters from FishBase 98. The results were unrealistic, however, and in the end the Ecopath software was allowed to calculate the P/B ratio for all but the demersal rays. For this group, the P/B value finally (1.30 $year^{-1}$) was adopted from the large zoobenthos feeders of the Gulf of Thailand 10-50m sub-model (Pauly and Christensen 1993). For all other groups the estimate of P/B was obtained by setting the GE to a prefixed value, according to criterion provided in Christensen and Pauly (1992a). The GE were as follows: 0.15 for adult large demersal predators, 0.20 for each of leiognathids, small demersals and medium demersals, and 0.30 for juvenile large demersal predators.

All Q/B ratios (except for demersal rays) were calculated using the empirical relationship of Pauly *et al.* (1990), as corrected in Christensen and Pauly (1992). Necessary input parameters were obtained from the FishBase database (FishBase 1998). The Q/B of demersal rays was adopted from the Q/B of large zoobenthos feeders of the Gulf of Thailand 10-50 m sub-model (Pauly and Christensen 1993).

The landings of demersal functional groups (Table 3.6) were derived from landing statistics of eight provinces and one municipality that catch fish from the Java Sea, for the period of

1974 to 1976 (DGF 1976, 1977, 1978). As the statistics do not discern between the juvenile or adult stages, all large demersal predatory fish that were caught were assumed to be in adult stage. This is due to the assumption that fishers were looking for fish that have commercial value, *viz.,* larger size, thus, adult size. In that case, for the purpose of this study, the landing of juvenile large demersal predators was estimated to be equal to the landing of small demersal fish. All diet compositions were extracted from FishBase 98. Some modifications were required during model balancing.

3.3.4.2 Pelagic groups

Pelagic fish were not caught effectively during the R/V *Mutiara IV* trawl survey. Thus, their biomass values (Table 3.7) were allowed to be calculated by the Ecopath software. For most groups, EE values were pre-set at 0.95, except for adult and juvenile large pelagic predators, whose EEs were arbitrarily set to 0.5, due to their position as top predators in the food web, where they are not heavily preyed upon.

Table 3.7 Input parameters for all pelagic fish functional groups in the mid-1970s model of the Java Sea (see sources in the text)

No.	Functional Groups	Biomass ($t \cdot km^{-2}$)	P/B ($year^{-1}$)	Q/B ($year^{-1}$)	Catch ($t \cdot km^{-2} \cdot year^{-1}$)	EE	GE
1.	Juvenile large pelagic predators	?	3.35	?	0.116	0.5	0.3
2.	Miscellaneous pelagics	?	?	10.87	0.117	0.95	0.2
3.	*Decapterus* spp.	?	?	14.05	0.071	0.95	0.2
4.	*Rastrelliger* spp.	?	?	14.16	0.035	0.95	0.3
5.	Clupeoids	?	3.35	17.62	0.116	0.95	?
6.	Adult large pelagic predators	?	1.2	8.65	0.051	0.5	?

? = To be estimated by Ecopath

The P/B ratios (Table 3.7), were initially treated as were those of the demersal fish groups. For reasons similar to those of the demersal groups, they were found to be unrealistic, however. Therefore, except for the large pelagic predator and clupeoids groups, the Ecopath software was allowed to calculate their values, necessitating the specification of the GE for each group. In this case, GEs were set to 0.2 for each of miscellaneous pelagics and *Decapterus* spp., and 0.3 for *Rastrelliger* spp. The P/B value for the adult large pelagic predators was adopted from the large pelagics of the Gulf of Thailand 10-50 m sub-model (Christensen and Pauly 1993). While the P/B ratio for the juvenile pool was set equal to the P/B ratio of the clupeoids group. The P/B for the clupeoids was estimated using the empirical formula of Pauly (1980) and biological parameters compiled in the FishBase database (FishBase 1998).

The Q/B ratios (Table 3.7), except for the juvenile large pelagic predators, were calculated using an empirical formula (Pauly 1980) and data from FishBase 98. For the juvenile large pelagic predators, the GE was set to a value of 0.3, and the Ecopath software was allowed to calculate the missing value.

The landings (Table 3.7) were derived from the landing statistics of eight provinces and one municipality that border the Java Sea, for the period of 1974 to 1976 (DGF 1976, 1977, 1978). Similar to what was done in the demersal groups, all fish that fall under the category of large pelagic predators were assumed to be in adult stage.

Diet composition was obtained from FishBase database (FishBase 1998). Some modifications were necessary during balancing.

3.3.4.3 Structure-associated fish (SAF)

Constructing members of SAF group was performed by comparing the following sources: (1) an account by Sainsbury *et al.* (1997), (2) a list of the so-called 'trash fish' that were caught during the R/V *Mutiara IV* survey in the study area (Losse and Dwiponggo 1977), and finally, (3) information on various food and feeding habits of LBS-dependent fish in FishBase 98. Members of this functional group are listed in Table 3.1.

Based on the aforementioned fish families, biomass input for SAF group was obtained from the swept area analysis of the R/V *Mutiara IV* survey in the study area (Torres *et al.* 1996), which was 0.377 $t{\cdot}km^{-2}$. The P/B ratio was left empty, and the Ecopath software was allowed to calculate the P/B value, with the EE pre-set arbitrarily to 0.75. The Q/B ratio (9.28 $year^{-1}$) was calculated empirically, similar to the demersal and pelagic fish groups.

The landing (0.01 t·km^{-2}·year^{-1}) for SAF functional group was taken from the combined landings of Sphyraenidae, Lutjanidae, Polynemidae, and Lethrinidae, from eight provinces and one municipality that border the Java Sea, for the period of 1974 to 1976 (DGF 1976, 1977, 1978). Diet composition was extracted from FishBase 98, and some modifications were made during balancing.

3.3.5 Marine mammals

As explained in section 3.1 above, for the purpose of this study, the marine mammals group are an aggregated group of resident dolphins and transient whales. Their membership was based on an exhaustive literature search (Klinowska 1991; Tas'an and Leatherwood 1984; Jefferson *et al.* 1993; White 1983, 1984; Bruyns 1971; International Whaling Commission 1984; Rice 1989; various scientific reports of Whales Research Institute; annual reports of International Whaling Commission). For a detailed species composition see Table 3.1. No studies were found that focused on the Java Sea; therefore the input parameters were calculated using primary data from similar areas and empirical calculations.

The biomass value (0.138 t·km^{-2}) was adapted from Trites *et. al.* (1997). The P/B ratio for the marine mammals group was estimated based on the rate of increase (r_{max}) of Spinner Dolphins (*Stenella longirostris*), which was estimated to be 9% (Perrin and Gilpatrick 1994). Using the assumption of Reilly and Barlow (1986) that production is half of r_{max}, a P/B ratio of 0.045 year^{-1} was therefore set for this group. The Q/B ratio (15.355 year^{-1}) of this group was calculated (Table 3.8) after Trites and Heise (1996), using resident dolphins as the primary species, and then averaged for all species.

Table 3.8 Deriving Q/B ratio for marine mammals with resident dolphins as the primary species, after the method of Trites and Heise (1996). Mean weight information was based on documentation provided in Trites and Pauly (1997).

No.	**Species**	**Sex**	**Mean weight (kg)**	**Food consumption (R, kg·day^{-1})**	**Daily ration** [a]	**Q/B ratio (year^{-1})** [b]
1.	*Tursiops* spp. (Bottlenose dolphins)	M	203.23	7.02	0.03	12.609
		F	171.89	6.14	0.04	13.039
2.	*Stenella longirostris* (Spinner dolphins)	M	43.13	2.03	0.05	17.192
		F	39.54	1.90	0.05	17.494
3.	*Orcaella brevirostris* (Irrawady dolphins)	M	107.15	4.21	0.04	14.332
		F	69.87	2.99	0.04	15.611
4.	*Neophocaena phocaenoides* (Finless porpoises)	M	44.54	2.08	0.05	17.082
		F	38.16	1.84	0.05	17.619
5.	*Sousa chinensis* (Indo-Pacific hump-backed dolphins)	M	153.55	5.61	0.04	13.337
		F	79.01	3.30	0.04	15.232
				Average Q/B ratio (year^{-1}) =		**15.355**

[a] Daily ration (fraction of body weight per day) was calculated as food consumption (kg·day^{-1}) divided by mean weight (kg).

[b] Q/B ratio (year^{-1}) in this case was calculated as daily ration multiplied by 365 days.

An account by Tas'an and Leatherwood (1984) indicated that dolphins were often caught as by-catch by local fishers, usually by purse seiners. However, the by-catch is not recorded in fisheries statistics[16]. Since marine mammals are caught, however, for the purpose of this model the smallest catch that can be accommodated (0.001 t·km^{-2}·year^{-1}) was entered.

Diet composition was taken from various sources on food and feeding habits of both baleen whales and toothed whales (Gales *et al.* 1992; Cheal and Gales 1992; Klinowska 1991; Perrin and Gilpatrick 1994; Perrin *et al.* 1989; Marsh *et al.* 1989; Jefferson *et al.* 1993; Ross *et al.* 1994; Kawakami 1980; Trites and Heise 1996; Kawamura 1977, 1980a, 1980b).

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[16] Similarly, it is known that whaling has been a traditional shore-based subsistence fishery using the old-style hand harpoon. This traditional fishery targets Sperm Whales in the eastern part of Indonesia; in particular in the coast of Lamalera and Lamakera (Savu Sea), and the Alor Islands of East Nusa Tenggara province (part of Lesser Sunda Islands) (Klinowska 1991; Barnes 1974). Again, this landing is not recorded explicitly in statistics.

3.4 Results and discussion

The construction of Ecopath model for the Java Sea provided basic information needed for the biological component of (fisheries) management, which will be further explored in the next chapter. Knowledge on how resource species impact on each other in an ecosystem context, for example, is crucial in giving an idea on how the ecosystem might respond to an externally imposed perturbation. An assessment of the ecosystem's maturity (and, therefore, stability) is also important in forecasting ecosystem recovery from perturbations.

A balanced Ecopath model produces a range of statistics which can be used to analyze the state of the ecosystem in question. These include among others EE (ecotrophic efficiency), GE (gross efficiency), flow to detritus, trophic levels, omnivory indices, respiration, assimilation, mortality coefficients, predation mortality coefficients, transfer efficiencies, network flow indices, and a variety of cycle and pathway analysis. The basic estimates generated by the Ecopath are discussed below, in light of some of the statistics noted above.

3.4.1 Basic estimation results

Final input parameters and results from model balancing are presented in Table 3.9. Table 3.10 presents the final diet compositions matrix, and Figure 3.1 presents simplified trophic fluxes within the system.

The Java Sea ecosystem spans over more than four trophic levels, with marine mammals and the fishery being the top predators occupying the highest trophic level, approximately 4.0. This level of fishery was also supported by the low value of its GE (0.0004, Table 3.11), indicative of fisheries for upper-level predators (Christensen and Pauly 1992a). The distribution of functional groups among trophic levels is relatively equal between low trophic levels (< 2.5) and intermediate trophic levels (2.5-3.5). There are 11 groups at low trophic levels, and 12 groups at intermediate trophic levels. The remaining five groups have trophic levels greater than 3.5. The relatively high number of groups located at similar trophic levels provides a situation where strong competition for resources occurs. In such circumstances, the direct impact of a fishery would be amplified throughout the entire system, by indirect interactions.

Table 3.9 Input and output (in brackets) parameters of the Ecopath model of the Java Sea, Indonesia, in mid-1970s

Functional Groups	**Biomass ($t \cdot km^{-2}$)**	**P/B ($year^{-1}$)**	**Q/B ($year^{-1}$)**	**EE**	**GE**	**Harvest ($t \cdot km^{-2} \cdot year^{-2}$)**	**Flow to detritus ($t \cdot km^{-2} \cdot year^{-2}$)**	**Trophic level**	**Omnivory index**	**R/A**	**R/B ($year^{-1}$)**
1. Benthic producers	153.000	11.885	-	(0.003)	-	0	(1812.255)	(1.000)	(0)	-	-
2. Phytoplankton	(4.154)	135.000	-	0.950	-	0	(28.036)	(1.000)	(0)	-	-
3. Small herb. zoopl.	2.430	60.225	220.000	(0.470)	(0.274	0	(291.454)	(2.000)	(0)	(0.544)	(71.775)
4. Lg. herb. zoopl.	0.560	20.000	70.000	(0.912)	(0.286	0.008	(16.666)	(2.000)	(0)	(0.524)	(22.000)
5. Carn. zoopl.	0.310	42.580	135.050	(0.238)	(0.315	0	(18.425)	(3.000)	(0)	(0.606)	(65.460)
6. Jelly fishes	0.100	5.011	25.050	(0.414)	(0.200	0.003	(0.795)	(3.000)	(0)	(0.750)	(15.029)
7. Benthic infauna	18.940	6.570	27.400	(0.537)	(0.240	0	(161.370)	(2.087)	(0.087)	(0.700)	(15.350)
8. SAF	0.377	(1.188)	9.280	0.750	(0.128	0.010	(0.812)	(3.483)	(0.251)	(0.840)	(6.236)
9. Macrozoobenthos	(2.455)	3.000	12.500	0.750	(0.240	0.001	(7.977)	(2.291)	(0.347)	(0.700)	(7.000)
10. LBS	20.000	0.100	0.500	(0.989)	(0.200	0.100	(2.021)	(2.070)	(0.085)	(0.750)	(0.300)
11. Juv. pen. shrimps	(0.556)	13.000	70.000	0.950	(0.186	0.008	(8.148)	(2.000)	(0)	(0.768)	(43.000)
12. Lg. pel. pred. (J)	(0.190)	3.350	(11.167)	0.500	0.300	0.116	(0.741)	(3.189)	(0.143)	(0.625)	(5.583)
13. Ad. pen. shrimps	(1.224)	5.000	28.945	0.950	(0.173	0.014	(7.392)	(2.214)	(0.185)	(0.784)	(18.156)
14. Misc. pelagics	(0.221)	(2.174)	10.870	0.950	0.200	0.117	(0.505)	(3.524)	(0.411)	(0.750)	(6.522)
15. Leiognathids	0.193	(3.674)	18.370	(0.510)	0.200	0.019	(1.057)	(2.912)	(0.235)	(0.750)	(11.022)
16. Crabs + Lobsters	(0.765)	4.000	21.900	0.950	(0.183	0.001	(3.504)	(2.460)	(0.308)	(0.772)	(13.520)
17. Cephalopods	(0.950)	3.100	20.318	0.950	(0.153	0.005	(4.009)	(3.161)	(0.122)	(0.809)	(13.154)
18. Decapterus spp.	(0.087)	(2.810)	14.050	0.950	0.200	0.071	(0.256)	(3.152)	(0.571)	(0.750)	(8.430)
19. Rastrelliger spp.	(0.044)	(4.248)	14.160	0.950	0.300	0.035	(0.135)	(2.630)	(0.233)	(0.625)	(7.080)
20. Clupeoids	(0.899)	3.350	17.620	0.950	(0.190	0.116	(3.318)	(2.940)	(0.099)	(0.762)	(10.746)
21. Small demersals	1.687	(2.568)	12.840	(0.553)	0.200	0.066	(6.270)	(3.120)	(0.172)	(0.750)	(7.704)
22. Med. demersals	0.046	(1.828)	9.140	(0.575)	0.200	0.015	(0.120)	(3.701)	(0.276)	(0.750)	(5.484)
23. Lg. dem. pred. (J)	(0.434)	(3.852)	12.840	0.500	0.300	0.066	(1.949)	(3.091)	(0.187)	(0.625)	(6.420)
24. Demersal rays	(0.009)	1.300	8.200	0.600	(0.159	0.006	(0.019)	(3.395)	(0.105)	(0.802)	(5.260)
25. Lg. pel. pred (A)	(0.104)	1.200	8.650	0.500	(0.139	0.051	(0.242)	(3.990)	(0.138)	(0.827)	(5.720)
26. Lg. dem. pred (A)	0.130	(0.920)	6.130	(0.174)	0.150	0.020	(0.258)	(3.885)	(0.212)	(0.812)	(3.985)
27. Marine mammals	0.138	0.045	15.355	(0.289)	(0.003	0.001	(0.428)	(4.088)	(0.045)	(0.996)	(12.239)
28. Detritus	120.000	-	-	(0.264)	-	0	(0)	(1.000)	(0.236)	-	-

Table 3.10 Diet composition matrix for all functional groups of the Java Sea model in mid-1970s. Values represent the proportion (on a weight or volume basis) each prey contributes to the diet of predator. All diet proportion sum up to 1 for each predator. See text for details.

PREDATOR PREY	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27
1. Benthic producers	-	-	-	-	-	0.010	0.150	0.100	-	-	0.001	-	-	-	-	-	-	0.002	0.020	0.030	-	-	-	-	-
2. Phytoplankton	0.700	0.700	-	-	0.170	-	0.290	0.040	0.800	-	-	0.060	0.010	-	-	0.233	0.370	0.080	-	-	0.072	-	-	-	-
3. Small herb. zoopl.	-	-	0.800	0.600	-	0.010	0.139	0.040	-	0.769	0.019	0.110	0.010	-	0.540	0.315	0.579	0.847	0.050	-	0.138	-	-	-	-
4. Lg. herb. zoopl.	-	-	0.200	0.400	-	0.010	0.001	0.010	-	0.001	0.001	0.040	0.010	-	0.010	0.028	0.001	0.001	0.010	-	0.006	-	-	-	0.001
5. Carn. zoopl.	-	-	-	-	-	-	0.070	0.010	-	0.020	-	0.030	0.010	-	-	-	-	0.020	0.020	-	-	-	-	-	-
6. Jellyfish	-	-	-	-	-	-	-	-	-	-	-	0.040	-	-	-	-	-	-	0.005	-	-	-	-	-	-
7. Benthic infauna	-	-	-	-	0.080	0.060	-	-	-	-	0.169	0.050	0.448	0.150	0.020	-	-	0.020	0.509	0.110	0.563	0.150	-	0.001	-
8. SAF	-	-	-	-	-	0.001	-	-	-	-	-	0.030	-	-	-	-	-	-	-	0.010	0.020	-	0.040	0.124	-
9. Macrozoobenthos	-	-	-	-	-	0.111	-	-	-	-	-	0.070	0.100	0.050	0.060	-	-	-	0.100	0.050	0.050	0.501	-	0.142	-
10. LBS	-	-	-	-	-	0.200	0.010	-	-	-	0.010	-	0.001	0.010	-	-	-	-	0.016	-	-	0.001	-	-	-
11. Juv. pen. shrimps	-	-	-	-	-	0.060	-	-	-	0.030	-	0.030	0.010	0.100	0.140	0.014	0.050	0.030	0.060	-	0.050	-	-	-	-
12. Lg. pel. pred. (J)	-	-	-	-	-	-	-	-	-	-	-	0.009	-	-	-	-	-	-	0.005	-	-	-	0.030	0.003	0.020
13. Ad. pen. shrimps	-	-	-	-	-	0.070	-	-	-	-	-	0.100	0.200	0.100	0.030	-	-	-	0.080	0.130	0.050	0.100	0.130	0.149	0.020
14. Misc. pelagics	-	-	-	-	-	-	-	-	-	-	-	0.005	-	-	-	-	-	-	0.004	-	-	-	0.150	-	0.050
15. Leiognathids	-	-	-	-	-	0.010	-	-	-	-	-	0.083	-	-	0.001	0.010	-	-	0.001	0.001	0.001	0.001	0.020	0.012	0.010
16. Crabs & Lobsters	-	-	-	-	-	0.110	-	-	-	-	-	0.005	-	-	0.100	-	-	-	0.022	0.130	-	0.098	0.001	0.050	-
17. Cephalopods	-	-	-	-	-	0.020	-	-	-	-	-	0.082	-	-	0.040	-	-	-	0.020	0.080	0.050	0.050	0.100	0.085	0.400
18. Decapterus spp.	-	-	-	-	-	-	-	-	-	-	-	0.030	-	-	-	-	-	-	-	-	-	-	0.004	-	0.040
19. Rastrelliger spp.	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	0.003	-	-	-	0.005	-	0.035
20. Clupeoids	-	-	-	-	-	-	-	-	-	0.180	-	0.122	-	-	0.020	0.400	-	-	0.015	0.010	-	-	0.470	0.002	0.209
21. Small demersals	-	-	-	-	-	0.170	-	-	-	-	-	0.088	0.001	-	0.027	-	-	-	0.010	0.400	-	0.099	-	0.204	0.209
22. Med. demersals	-	-	-	-	-	-	-	-	-	-	-	-	-	-	0.001	-	-	-	-	-	-	-	-	0.015	0.001
23. Lg. dem. pred. (J)	-	-	-	-	-	0.148	-	-	-	-	-	-	-	-	0.001	-	-	-	-	0.049	-	-	0.050	0.209	-
24. Demersal rays	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	0.001	-
25. Lg. pel. pred (A)	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	0.001	0.005
26. Lg. dem. pred (A)	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	0.001	-
27. Marine mammals	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	0.001	-
28. Detritus	0.300	0.300	-	-	0.750	0.010	-	0.800	0.200	-	0.800	0.016	0.200	0.590	0.010	-	-	-	0.050	-	-	-	-	-	-

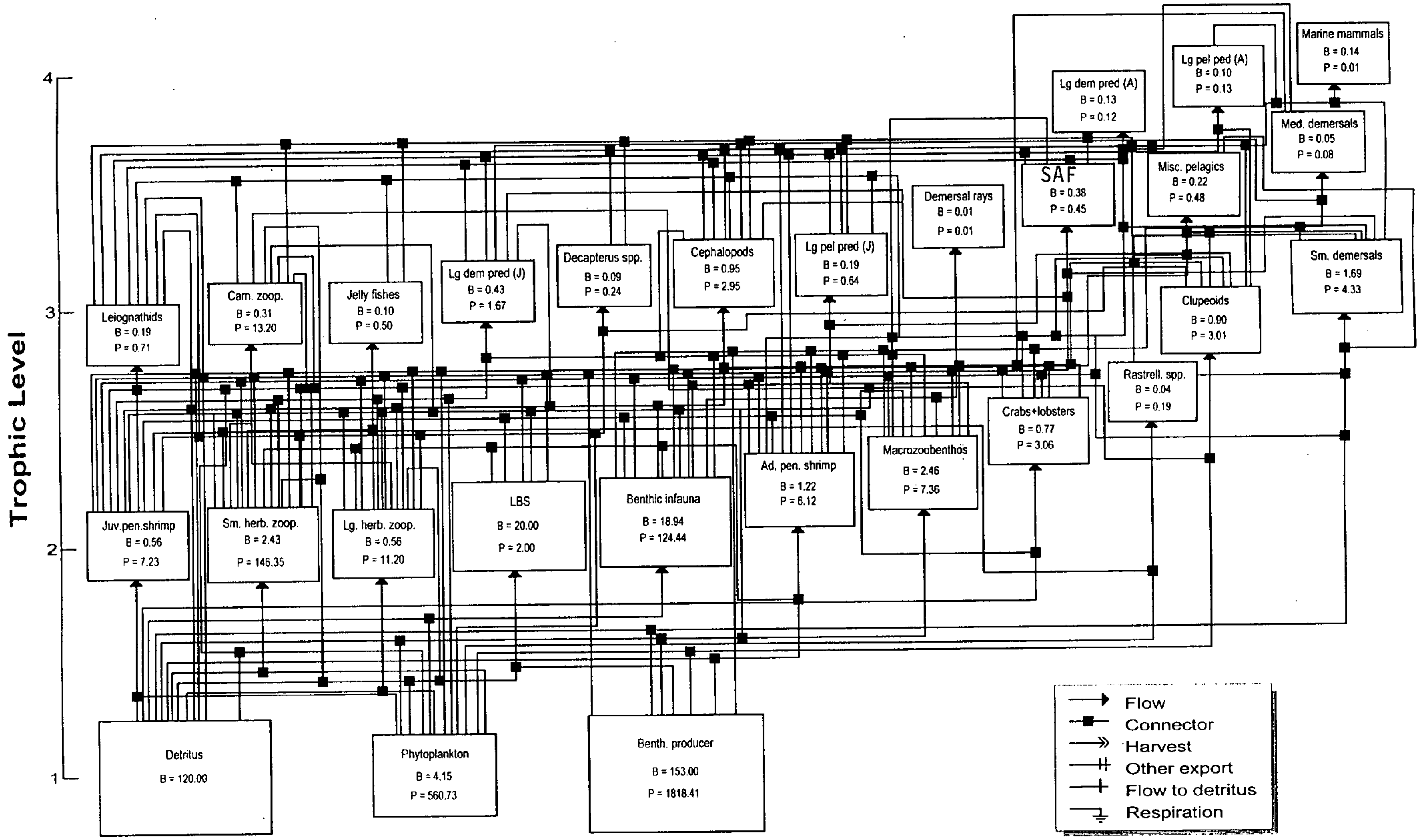

Figure 3.1 Simplified trophic flow diagram of the Java Sea ecosystem, Indonesia in mid-1970s, showing biomass (B, t·km^{-2}), production (P, t·km^{-2}·year^{-1}), and outflows (> 0.01 t·km^{-2}·year^{-1}) among functional groups, all arranged by their trophic levels. For reason of clarity, numerical labeling of flows, and symbols for harvest, other export, flows to detritus, cannibalism and respiration were omitted. Flows leaving a box from the top or the upper half of the box, represent consumption by other groups on that box. In contrast, flows (exiting other boxes) which were merged by connectors into one flow that enter a box from below, represent consumption by that box. The area of the sea is 471,000 km^2.

3.4.2 Trophic impact assessment

Figure 3.2 presents the trophic interactions of the ecosystem under study herein. Although it is not always graphically apparent (i.e., because of the scale), all functional groups responded negatively to an increase in their own biomass. This is due to increased within-group competition for food resources.

The fishery responded positively to an increased biomass of all functional groups in the system (including unfished groups), with the exception of the following: benthic infauna (i.e., -0.03), SAF (i.e., -0.02), crabs and lobsters (i.e., -0.01), and cephalopods (i.e., -0.02). This happened because of the negative impacts these groups had on the fished groups, either through direct or indirect (i.e., cascade) effects.

The fishery was positively affected mostly by lower and intermediate trophic levels, as indicated by catch increases with the increased biomass of phytoplankton (TL = 1.0), small herbivorous zooplankton (TL = 2.0), LBS (TL = 2.1), *Rastrelliger* spp. (TL = 2.6), clupeoids (TL = 2.9), juvenile large demersal predators (TL = 3.1), juvenile large pelagic predators (TL = 3.2), and *Decapterus* spp. (TL = 3.2). The only high trophic level group that positively affected the fishery was the miscellaneous pelagics (TL = 3.5). The fishery experienced its largest positive increases from increases in primary productivity and zooplankton biomasses.

As indicated earlier, the fishery and the marine mammals group occupied the same trophic level (i.e., 4.1), hence, competition is very likely[17]. The trophic impact routine confirms this, although the effect appears slight. Perhaps the effect was attenuated because the marine mammal group is highly aggregated. An increase in marine mammal biomass slightly decreased the fishery (i.e., -0.05), while an increase in the fishery produced slightly less impact on the marine mammals (i.e., -0.01).

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[17] Trites *et al.* (1997) indicated a 'food web competition' between fisheries and marine mammals, which suggests that competition between fisheries and marine mammals is more a matter of indirect competition for primary production, than of direct competition for prey.

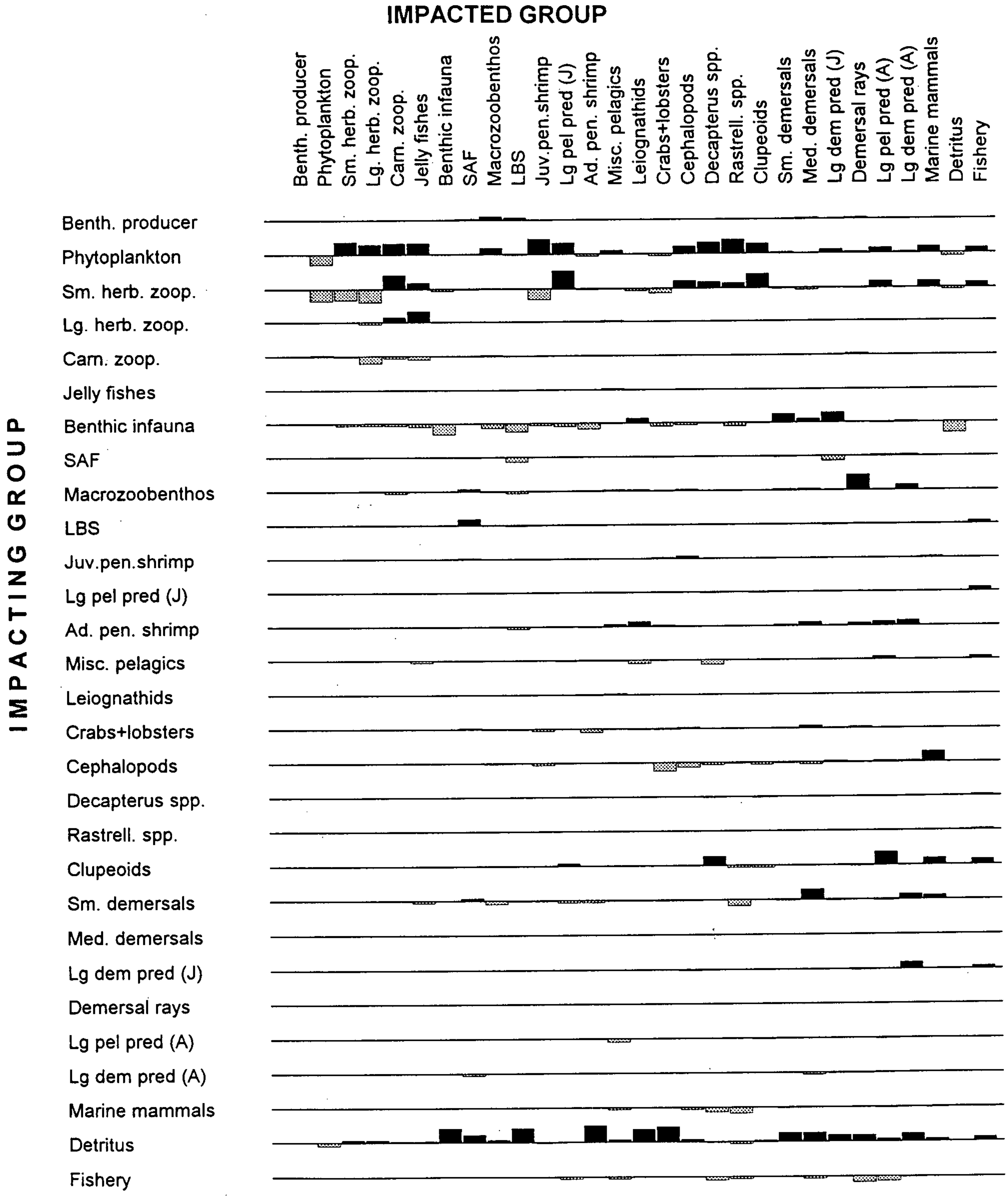

Figure 3.2 Results of trophic impact routine, showing relative impacts of an increase of the groups listed on the left of the histogram (rows) have on the groups listed above the histogram (columns). Positive impacts (biomass increases) are black and above the lines, and negative impacts (biomass decreases) are gray and below the lines.

3.4.3 Maturity analysis of the Java Sea: a comparative study

A system for measuring the of maturity of an ecosystem was first described by Odum (1969). Ulanowicz (1986) developed this further to include a new interpretation of ecosystem growth and development, using concepts mainly drawn from thermodynamics and information theory[18].

Key concepts that are parts of Odum and Ulanowicz's theories are available as Ecopath routines (Christensen and Pauly 1992a, 1992b). These routines were used by Christensen (1994, 1995) and Christensen and Pauly (1993) to compare the maturity and stability levels of 41 aquatic ecosystems represented by Ecopath models. A comparison between the Java Sea ecosystem, and these 41 aquatic systems, can provide a relative assessment of the maturity of the Java Sea system, and perhaps elucidate possible responses to perturbation (see section 1.3 in Chapter 1).

Table 3.11 presents the summary statistics, while Table 3.12 and Table 3.13 list network flow indices and transfer efficiencies, respectively, for the Java Sea model in mid-1970s.

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[18] As ecosystems mature, there is an increase in its growth and development. The increase in its growth involves an increase in its size, and is measured as the sum of all flows, *viz.*, total system throughput of all compartments (i.e., thermodynamic concepts). While an increase in its development is represented by an increase in the organization and information content of the flows, and is quantified using the 'average mutual information' (i.e., information theory). Therefore, when ecosystems mature, total system throughput and information are assumed to increase. See Ulanowicz (1986) for more in-depth discussion.

Table 3.11 Summary statistics computed by Ecopath for the Java Sea model, Indonesia, in mid-1970s

Property (units)	
Sum of all consumption (t•km^{-2}•year^{-1}):	1349.02
Sum of all exports (t•km^{-2}•year^{-1}):	875.99
Sum of all respiratory flows (t•km^{-2}•year^{-1}):	627.99
Sum of all flows into detritus (t•km^{-2}•year^{-1}):	2378.16
Total system throughput (t•km^{-2}•year^{-1}):	5231.16
Sum of all production (t•km^{-2}•year^{-1}):	2715.60
Mean trophic level of the fishery:	4.062
Gross efficiency (catch/net primary production):	0.0004
Calculated total net primary production (t•km^{-2}•year^{-1}):	2379.13
Total primary production/total respiration (P_p/R):	3.79
Net system production (t•km^{-2}•year^{-1}):	1751.15
Total primary production/total biomass (P_p/B) (year^{-1}):	11.33
Total biomass/total throughput (year^{-1}):	0.04
Total biomass (excluding detritus) (t•km^{-2}):	210.00
Total catches (t•km^{-2}•year^{-1}):	0.849
Connectance Index:	0.283
System Omnivory Index:	0.146

Table 3.12 Network flow indices of the Java Sea as computed by Ecopath

SOURCE	Ascendency		Overhead		Capacity	
	Flowbits	**%**	**Flowbits**	**%**	**Flowbits**	**%**
Import	0.0	0.0	0.0	0.0	0.0	0.0
Internal flow	4144.1	22.1	8867.9	47.3	13012.0	69.3
Export	991.2	5.3	1280.0	6.8	2271.3	12.1
Respiration	1260.6	6.7	2221.2	11.8	3481.8	18.6
Totals	**6396.0**	**34.1**	**12369.1**	**65.9**	**18765.1**	**100.0**
Finn's Cycling Index	**7.4 (% of total throughput)**					
Finn's Mean path length	**3.48**					

Table 3.13 Transfer efficiencies (TE) and flows at each discrete trophic level in the Java Sea model

Trophic Level	**TE (%)**	**OUTFLOWS**			**INFLOWS**	
		Consumption by predators (t·km^{-2}·year^{-1})	**Consumption by fisheries (t·km^{-2}·year^{-1})**	**Sum of all outflows (t·km^{-2}·year^{-1})**	**Throughput (t·km^{-2}·year^{-1})**	**% of TE due to F***
II	10.7	124.21	0.19	124.40	1166.70	0.15
III	11.6	13.89	0.47	14.36	124.17	3.27
IV	11.4	1.41	0.18	1.59	13.93	11.32
V	9.2	0.11	0.02	0.13	1.42	15.39
VI	8.1	0.007	0.002	0.009	0.11	22.22
VII	7.7	0.0	0.0	0.0	0.01	0.0

Proportion of total flow originating from:
detritus = 0.42
primary producers = 0.58

Note: * F = fishing mortality (year^{-1}).

In the mid-1970s, the Java Sea was moderately dependent on detritus (Table 3.13); only less than half of the flows originated from detritus. Odum (1969) stated that as ecosystems mature, they should become more dependent on detrital flows than on flows from primary producers.

Odum (1971) indicated that the ratio between total primary production and total respiration (P_p/R) is an excellent functional index of the relative maturity of an ecosystem. This ratio would approach 1 as systems mature. In their comparative study of 41 aquatic ecosystems, Christensen and Pauly (1993, Figure 2) found that the bulk of P_p/R ratios were in the range between 0.8 to 3.2, although the extreme values ranged from < 0.8 to > 6.4. The P_p/R ratio of the Java Sea was 3.8 (Table 3.11).

In their account, Christensen and Pauly (1993) indicated that ecosystems with very high P_p/R ratios usually will either have problem in model parameterization (specifically problems with quantification of assimilation rates, and hence indirectly of respiration), or have omitted bacterial activity from the system. In the first case, usually the ratio between total export and system throughput exceeds 0.3 (Christensen and Pauly 1993). In the Java Sea, this ratio was found to be 0.17 (Table 3.11). Hence, problems of model parameterization are not likely to be the cause, which lead us then to the second case, i.e., omission of bacterial activity. In this study, bacterial activity was not included, which in this case overestimate the P_p/R ratio. The Java Sea ecosystem acts as a detrital sink for the surrounding islands, from which the run-off (i.e., suspended solids and particulate solids) comes primarily from agricultural sources. This provides an explanation for the high P_p/R value obtained for this model.

As discussed by Margalef (1968), the ratio between total system productivity and total system biomass (P/B) is high in developing systems and low in mature systems. Christensen and Pauly (1993, Figure 3) ranked the P/B ratios of 41 aquatic ecosystem according to the maturity ranking of Odum (1969). Compared to these 41 aquatic systems, the Java Sea, having a total P/B of 8.34 $year^{-1}$ (Table 3.11), is placed at an intermediate level of maturity.

Another measure of maturity is cycling, which is assumed to increase as systems mature (Odum 1969). Finn (1976) quantified this using an index now called Finn's Cycling Index (FCI), which expresses the percentage of the total throughput actually recycled in the system. As maturity *sensu* Odum (1969) was shown to be related with stability *sensu* Rutledge *et al.* (1976), Christensen and Pauly (1993, Figure 6) showed that when FCI is plotted against system overhead for a large number of ecosystems, they provide a parabolic correlation. The apex or inflection point of the parabola is the optimal stable point. Values on either side of the inflection are moving away from stability. When the FCI (7.4%, Table 3.12) and the system overhead (65.9%, Table 3.12) of the Java Sea are plotted onto the parabolic relation of Christensen and Pauly (1993, Figure 6), they placed the Java Sea as a relatively stable ecosystem. Thus, the Java Sea ecosystem is less stable than the Gulf of Thailand and the Brunei coast, but more stable than the Gulf of Mexico continental shelf and Monterey Bay.

Christensen and Pauly (1993, Figure 8) also plotted the FCI of 41 aquatic ecosystems against their total primary production/total respiration (P_p/R) ratios, and indicated that P_p/R ratio moves toward unity and FCI increases as ecosystems mature. When the Java Sea's indices of FCI (7.4%, Table 3.12) and P_p/R (3.79, Table 3.11) are plotted onto this correlation (Christensen and Pauly 1993, Figure 8), they placed the Java Sea at an intermediate level of maturity.

In terms of the correlation between FCI (7.4%, Table 3.12) and mean path length (3.48, Table 3.12), the Java Sea fell within the intermediate range of maturity among all the ecosystem models described in Christensen and Pauly (1993, Figure 9).

Following Lindeman's (1942) definition of trophic transfer efficiencies (TE) and using a method to calculate TE described in Christensen and Pauly (1992), Christensen and Pauly (1993) calculated TE for all 41 aquatic ecosystem models and plotted them all against their respective discrete trophic levels (Ibid., Figure 16). Average TE by trophic levels was as follows: 10% for the herbivores/detritivores, 11% for the next trophic level, and lower efficiency (7.5 - 9.0%) on the higher trophic levels. The overall mean TE was 9.2%. The TEs of the Java Sea in mid-1970s were similar (Table 3.13).

In summary, the Java Sea ecosystem model in mid-1970s behaves as can be expected from a tropical shelf system. Furthermore, it can be regarded as moderately mature and relatively stable. The impact of the fishery was low to moderate in comparison with the fisheries of other systems. Therefore, it is anticipated that the Java Sea ecosystem should be moderately resilient to perturbations.

CHAPTER 4:

DYNAMIC MULTISPECIES MODELLING OF THE JAVA SEA

4.1 Preparing an Ecosim run

Ecosim was designed to operate using largely the same data set as for the Ecopath model. The one major addition is that in ecosystems where trophic ontogeny was explicitly defined, juvenile-adult links must be established, and various recruitment parameters entered (see text in section 4.1.2 for further details).

4.1.1 Allocating catches to different gear types

As outlined in Chapter 2, for the purpose of Ecosim simulations in this study, all of the 28 types of gear that operate in the study area were categorized into three groups, *viz.,* trawlers, purse seiners and small scale gears. Thus, all catches from the Java Sea were allocated to each gear type, by functional group. The allocation process was not straightforward. One of the main reasons for this is the national fisheries statistics system of Indonesia does not maintain record on species composition of the catch per gear, but only on the total catch per gear and the total catch per species[19]. Thus, in order to allocate the catch by functional group for each gear type, the following procedure was used.

First, the composition of trawlers' catches in this study was estimated by using the catch composition from the R/V *Mutiara IV* trawl survey in the Java Sea (Martosubroto and Pauly 1976; Pauly *et al.* 1996; Torres *et al.* 1996). Each of these values were then applied to the total average annual landings of all trawlers (0.107 t·km^{2}·year^{-1})[20] operating in the Java Sea

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[19] Catch data from landing sites are recorded and collated by officers at fish auction places (TPI, *Tempat Pelelangan Ikan*), which are then reported to Fisheries Service Offices at district level. Once compiled at district level, these data are then reported to the Fisheries Service Offices at provincial level, which will make summary report (out of reports from different districts), to be reported to the national fisheries statistics division in DGF. Consequently, if one needs more elaborate data such as on species composition of the catch by gear, should go directly to the Fisheries Service Offices at the provincial or district level (see e.g., Martosubroto 1982).

[20] According to DGF (1976, 1977, 1978), in mid-1970s, trawlers accounted for 12.6% of the total catch from the Java Sea (0.849 t·km^{-2}·year^{-1}, Table 3.9 in Chapter 3). Thus, the total trawlers' landing for that period is 0.107 t·km^{-2}·year^{-1}.

from 1974 to 1976, giving the estimated catch per functional group caught by trawlers in mid-1970s.

Second, the catch composition of purse seiners for the simulation was estimated using the catch composition of the purse seine fishery of the northern coast of Central Java in 1991[21]. Similar to allocation of catch composition of trawlers, these values were then applied to the total average annual landing of all purse seines (0.074 t·km^{-2})[22], and the estimated catch by functional group for purse seine was then calculated.

Third, the allocation of catch composition for small scale gears (by functional group) were estimated as the difference between the total catch for each functional group (from the Java Sea, Table 3.9 in Chapter 3) and the estimated catches by trawlers and purse seiners.

Finally, the catch by functional group caught by each gear were estimated by applying the proportions obtained above to the reported total catches of each group (Table 4.1).

4.1.2 Defining links for juvenile-adult split pools

In the present study, there are three pairs of functional groups that are ontogenetically linked. These are comprised of the penaeid shrimps (juvenile and adult), large pelagic predators (juvenile and adult) and large demersal predators (juvenile and adult).

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[21] Kindly provided by Dr. Akhmad Fauzi of the Centre for Coastal Resource Management, Bogor Agricultural University, Indonesia; who obtained the data directly from the provincial Fisheries Service Office in Semarang, Central Java.

[22] In mid-1970s purse seiners accounted for 8.7% (DGF 1976, 1977, 1978) of the total catch from the Java Sea (0.849 t·km^{-2}·year^{-1}, Table 3.9 in Chapter 3). Thus, the total purse seiners' landing for that period is 0.074 t·km^{-2}·year^{-1}.

Table 4.1 Catch proportion of each gear type by functional group, used for Ecosim. See text for details.

No.	Functional Group	Catch from the Java Sea ($t\cdot km^{2}\cdot year^{-1}$)	Catch by trawlers ($t\cdot km^{2}\cdot year^{-1}$)	Catch by purse seiners ($t\cdot km^{2}\cdot year^{-1}$)	Catch by small scale gears ($t\cdot km^{2}\cdot year^{-1}$)	Catch proportion [a] of trawls	Catch proportion [a] of purse seines	Catch proportion [a] of small scale gears
1.	Misc. pelagics	0.117	0.0037	0.01166	0.10116	0.03213	0.1	0.86788
2.	Lg. pel. pred. (juvenile)	0.116	0	0	0.116	0	0	1
3.	Clupeoids	0.116	0.0001	0.00578	0.10976	0.00093	0.05	0.94908
4.	*Decapterus* spp.	0.071	0.0000	0.01414	0.05654	0	0.2	0.8
5.	Small demersal	0.066	0.0321	0	0.03440	0.48269	0	0.51731
6.	Lg. dem. pred. (juvenile)	0.066	0	0	0.066	0	0	1
7.	Lg. pel. pred. (adult)	0.051	0.0002	0.03313	0.01728	0.00423	0.65439	0.34138
8.	*Rastrelliger* spp.	0.035	0.0000	0.00708	0.02832	0	0.2	0.8
9.	Lg. dem. pred. (adult)	0.02	0.0051	0	0.01541	0.24774	0	0.75226
10.	Leiognathids	0.019	0.0043	0	0.01448	0.22810	0	0.77190
11.	Medium demersal	0.015	0.0018	0	0.01347	0.11703	0	0.88297
12.	Adult penaeid shrimps	0.014	0.0082	0	0.00548	0.6	0	0.4
13.	SAF	0.01	0.0100	0	0.00007	0.99274	0	0.00726
14.	LBS	0.1	0.04	0	0.06084	0.39165	0	0.60835
15.	Juv. penaeid shrimps	0.008	0	0	0.008	0	0	1
16.	Lg. herb. zoopl.	0.008	0.0000	0	0.00779	0	0	1
17.	Demersal rays	0.006	0.0001	0	0.00568	0.00933	0	0.99067
18.	Cephalopods	0.005	0.0021	0.00109	0.00222	0.39253	0.2	0.40747
19.	Jellyfish	0.003	0.0000	0	0.00346	0	0	1
20.	Macrozoobenthos	0.001	0.0000	0	0.00138	0	0	1
21.	Crabs & lobsters	0.001	0.0001	0	0.00050	0.17563	0	0.82437
22.	Marine mammals	0.001	0	0.001	0	0	1	0
Absolute total catch ($t\cdot km^{-2}\cdot year^{-1}$) =		**0.849**	**0.107**	**0.074**	**0.668**			
Relative total catch (%) =		**100%**	**12.6%**	**8.7%**	**78.7%**			

[a] Catch proportions were calculated by functional group *i* that was caught by each gear *j*. Total proportion sum up to 1 for each functional group.

As mentioned in section 1.5.2.1, the three additional parameters required by Ecosim are: the curvature parameter of von Bertalanffy growth model (K, in year^{-1}), the weight at which juveniles becomes adults (W_k in kg), and the age at which juveniles become adults (T_k, in years). Table 4.2 presents these three additional parameters for each of the three pairs of juvenile-adult pools simulated in the present study.

Table 4.2 Parameters defining the transition from juveniles to adults in Ecosim split pools.

Functional Groups	**W_k (kg)**	**T_k (year^{-1})**	**K (year^{-1})**
Penaeid shrimps	0.010 [a]	0.5 [a]	0.928 [b]
Large pelagic predators	1.291 [c]	2.5 [c]	0.401 [c]
Large demersal predators	1.492 [d]	3.0 [e]	0.453 [e]

[a] From A. Courtney (Queensland Department of Primary Industry, Southern Fisheries Centre, *pers. comm.*, 1998);

[b] Calculated from L_∞ and ϕ' data of penaeid shrimps in Pauly *et al.* (1993);

[c] Represented by *Euthynnus affinis* (FishBase 1998);

[d] Represented by *Carcharhinus sorrah* (FishBase 1998);

[e] Represented by *Carcharhinus sorrah* (Davenport and Stevens 1988).

4.2 Results and discussion

4.2.1 Ecosystem impact of management scenario 1

For fishery management scenario 1, a 50 year simulation was undertaken. Here, fishing pressure by trawlers, purse seiners and small scale gears increased gradually over time (Figure 4.1). Predation control was set as top-down, intermediate and donor-controlled (Figure 4.2).

Under top-down control (Figure 4.2, upper panel), a trophic cascade effect was evident. Increased fishing caused the biomass of individual groups to decline quickly and dramatically. Most notable among these groups were medium demersals, large demersal predators (adults and juveniles), *Decapterus* spp., and large pelagic predators (adult and juveniles). Though not dramatic, decline in the biomass of those groups also occurred in the intermediate and bottom-up type control.

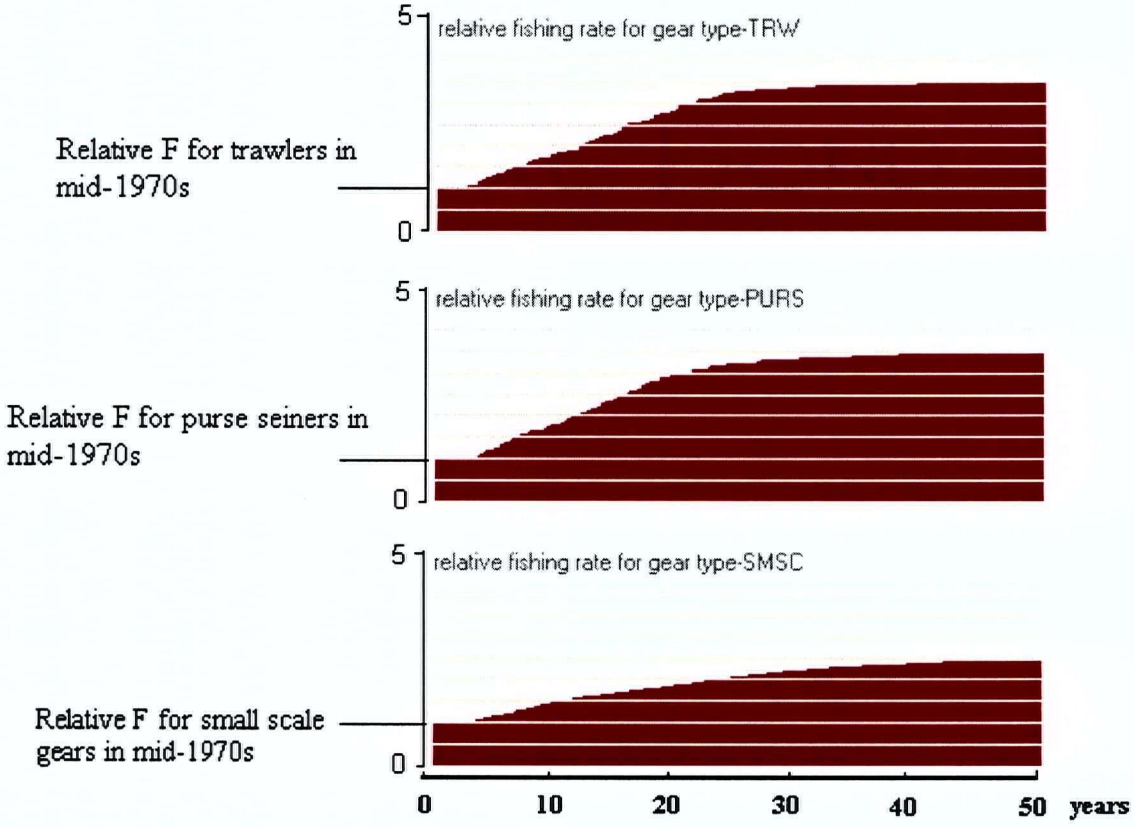

Figure 4.1 Patterns of fishery management scenario 1: "what would have happened to the ecosystem if the trawl ban had never been implemented ?". Here, fishing rates for trawlers and purse seiners were increased moderately throughout 50 years, up to three and a half times the level in mid-1970s. On the other hand, throughout 50 years, fishing rate for small scale gears was only increased slightly, up to two and a half times the level in mid-1970s.

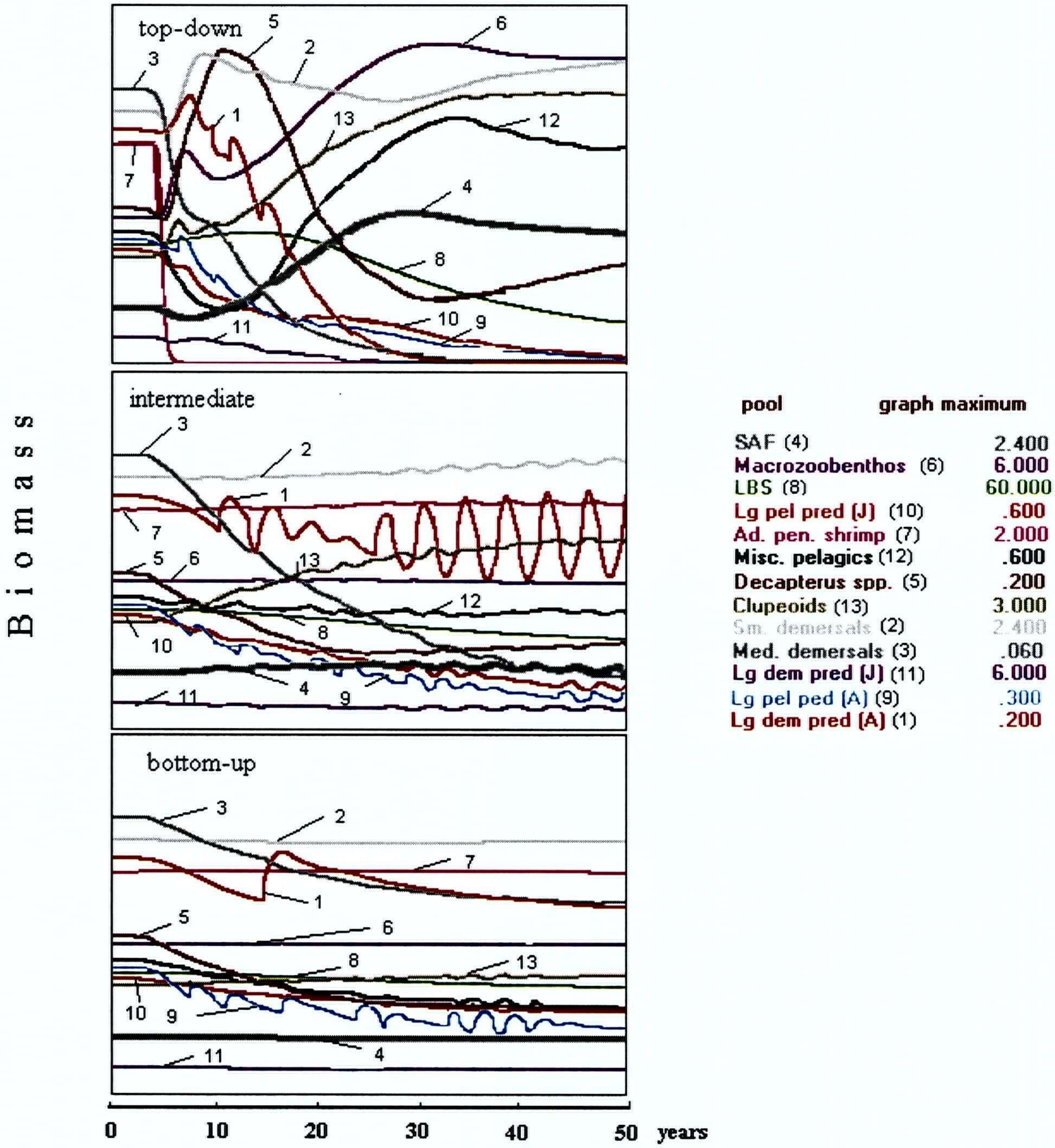

Figure 4.2 Dynamic simulations of fishery management scenario 1, under top-down, intermediate and bottom-up controls. The colored lines are biomass, and the thick dark gray line is SAF (structured-associated fish). See text for details and Fig. 4.1 for patterns of scenario 1.

In contrast, the biomass of smaller organisms such as small demersals, clupeoids, macrozoobenthos, large herbivorous zooplankton and jellyfish had a tendency to increase over time under both the top-down and intermediate control assumptions. This was mainly due to a decrease in predation pressure on them, as higher trophic level groups were fished out.

Under top-down control, both adult and juvenile penaeid shrimp biomass exhibit a drastic decline as fishing pressure is increased. The decline occurred prior to any large increase of the biomass of their predators. From Figure 4.2 (upper panel), it is not clear whether the declines were caused by trophic cascade or fishing. From shorter simulation (10 years, Figure 4.3), however, it was evident that an oscillation in the phytoplankton (the major component in the diet of juvenile penaeid shrimps) and detritus biomasses induced a more drastic oscillation in the biomass of juvenile penaeid shrimps. The oscillation of the juveniles in turn affects the adult shrimps, through recruitment. A concurrent, but minor, increase in the biomass of penaeid shrimps' predators (e.g., leiognathids and miscellaneous pelagics in this case) as a result of decreased predation from higher tropic levels, served to amplify the impact on penaeid shrimps. In the intermediate and bottom-up controls, however, penaeid shrimps are relatively stable over time. Thus, under top-down control, penaeid shrimp groups are sensitive to change.

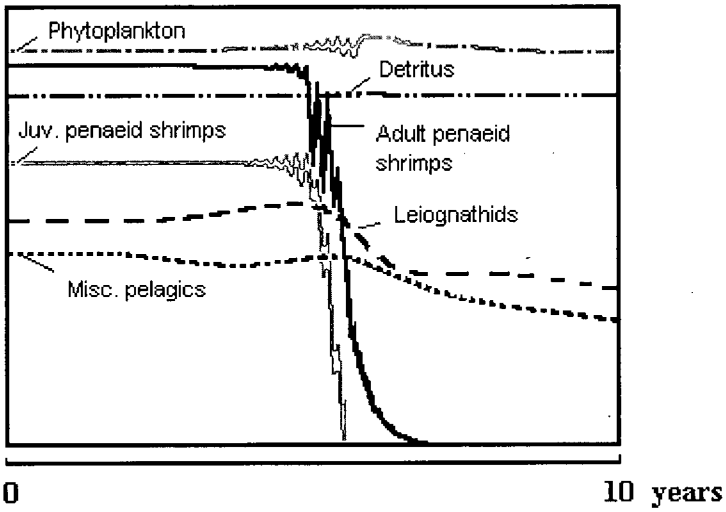

Figure 4.3 Dynamic simulations of scenario 1 (top-down control) are zoomed in for the first 10 years to show trophic cascade interactions. See text for details.

Despite the increase of fishing pressure due to trawling, under the top-down assumption the biomass of LBS did not decrease as expected. One possible cause is that the catch rate of LBS was underestimated in this model. Further, LBS is a component of the diet of many groups in the system. Thus, the trophic cascade effects on LBS may, in aggregate, have been stronger than the effect of increased fishing pressure. Further evidence that the cascade effect was a more influential factor in the biomass of LBS than the impact of the trawl fishery can also be inferred from the results. Here, I make a distinction between the direct impact of fishing, and the indirect cascade effects caused by some level of fishing effort. As fishing rates increased, the SAF (directly impacted) decreased slightly. In contrast, however, the LBS (also directly impacted by fishing) increased slightly as predation pressure was reduced. Thus, the indirect cascade effects were more influential, at least in the beginning. After about 15 years of the simulation, however, the biomass of the LBS began a gradual decline, due in part to both the combined effects of increased fishing pressure and low recovery rate over time (direct), and an increase in the SAF associated with declining predation (indirect).

This simulation suggests that, in the short term, the fishery impacts (negative) on the LBS are negligible relative to trophic impacts (positive). Over time and under top-down control, however, both impacts are additive and negative. The loss of LBS did not negatively impact the SAF group, possibly because predation pressure was decreased and the increased availability of other diet items for SAF were more than compensatory. This holds true for all but very extreme fishing rates (see scenario 4 in section 4.2.4).

Sainsbury *et al.* (1997) indicated that the relationship between LBS and SAF is of a habitat and behavioral nature[23]. Hence, it is important to note here that these results are, at least in part, an artifact of the method. LBS may be far more important as a refuge from predation rather than as a food source for SAF. Thus, the loss of LBS over time might not be compensated for by increasing food sources and decreased predation on the SAF.

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[23] In their account, after 5 years of experiment, it was estimated that the probability associated with the effect of bottom habitat on the carrying capacity of demersal species was 0.62. This was calculated using a Bayesian updating method described in Sainsbury (1991).

Under the intermediate type control (Figure 4.2 middle panel), where trophic cascades are less pronounced, fishing pressure due to trawling provided a direct negative impact on LBS. The biomass of LBS decreased gradually as fishing increased. Meanwhile, however, SAF showed a slight oscillation but a gradual increase over time. Here, the removal of predation pressure and increased availability of other diet items for SAF still play a role, though very minimal.

In summary, a negative impact, as a result of trawling on LBS and SAF occurred under bottom-up control (Figure 4.2, lower panel). Albeit not apparent in the graphs, increased fishing pressure exerted direct negative impact on both LBS and SAF.

4.2.2 Ecosystem impact of management scenario 2

In fishery management scenario 2 (Figure 4.4), the model was simulated for 50 years to mimic the status-quo, under all three energy flow assumptions (Figure 4.5). Here, the trawl fishery was gradually phased out starting in 1980. Purse seiners quickly replaced the trawlers, because many trawlers had their vessels refitted for use with purse seiners. Concurrently, small scale gears also increased.

Under all three energy flow assumptions, the patterns of biomass changes (Figure 4.5) were similar, but more pronounced, to those in scenario 1 (Figure 4.2). In the top-down assumption, for example, the *Decapterus* spp. biomass was more heavily impacted by the more rapid increase of purse seining. Large demersal predators increased more rapidly and peaked at higher biomass (compared to scenario 1) right after the ban, but declined faster and had smaller biomasses during later years of the simulation. The ban did not affect the large pelagic predators at the beginning. However, at later years of the simulations, they declined rapidly and had smaller biomasses. In turn, this gave an opportunity for their prey species to increase to higher abundance and at a faster rate, relative to scenario 1.

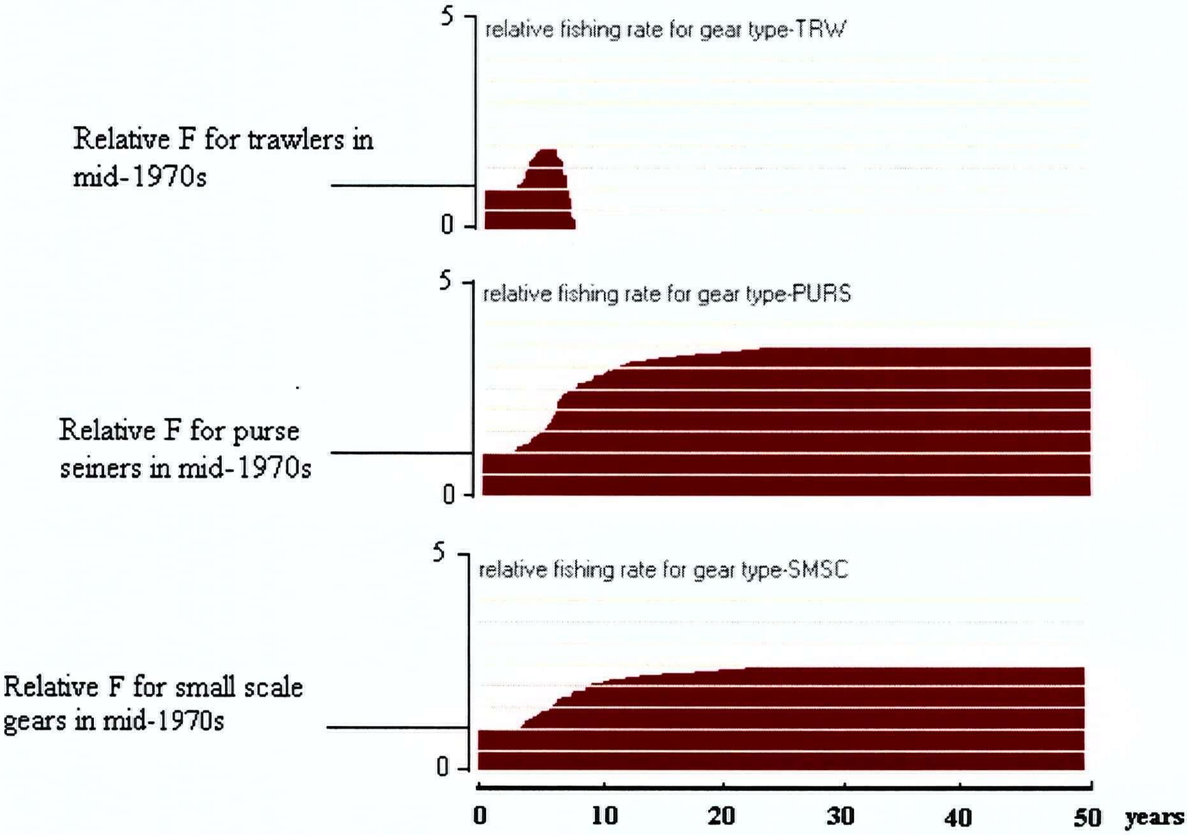

Figure 4.4 Patterns of fishery management scenario 2: "what can be expected to happen to the ecosystem and the fisheries if the status-quo is maintained ?". Here, trawling was gradually decreased from 1980 to approximately 1982, as per the decree. Fishing rate for purse seiners increased quickly after the trawl ban, and continued to increase moderately (up to three and a half times the level in mid-1970s). In 50 years, fishing rate for small scale gears also increased up to two and a half time the level in mid-1970s.

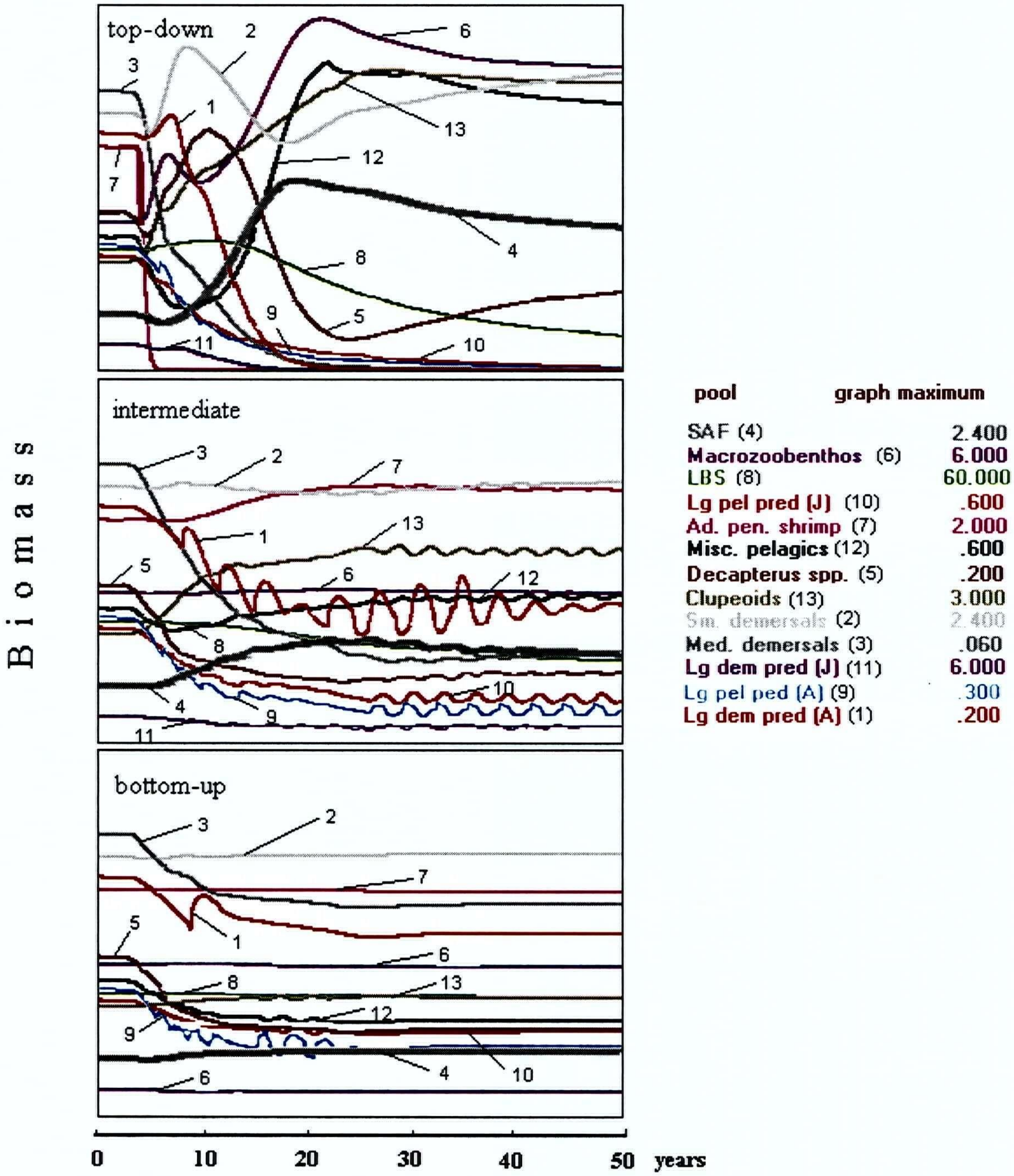

Figure 4.5 Dynamic simulations of fishery management scenario 2, under top-down, intermediate and bottom-up controls. The thick dark gray line is SAF (Structure-associated fish), and the other colored lines are biomass of other groups. See text for details and Fig. 4.4 for patterns of scenario 2.

In this scenario and under top-down control (Figure 4.5, upper panel), the SAF (directly impacted by the trawl) increased at a higher rate relative to scenario 1. In contrast, LBS declined faster relative to scenario 1, despite the trawl ban. Here it is shown that combined effects from the trawl ban and the decline of large demersal and pelagic predators (due to increased fishing), created a situation where their prey (SAF) could attain higher relative abundance and at a faster rate. In turn, the decrease of LBS due to predation by SAF occurred at a faster rate, relative to scenario 1.

A decrease in biomass was noted for the penaeid shrimps (Figure 4.5, upper panel), similar to scenario 1. The ban of trawling did not result in a recovery of the penaeid shrimps stock (Figure 4.5, upper panel). However, under the intermediate control assumption (Figure 4.5, middle panel), the biomass of the shrimps gradually and constantly increased to a level slightly higher than its initial biomass following the trawl ban. This could be a result of the combination of the trawl ban and increased fishing on penaeid shrimps' predators.

Nevertheless, natural ecosystems most likely have a mixture of high (top-down control) and low (bottom-up control) vulnerability, and for a tropical ecosystem with complex trophic interactions, it is difficult to make simple and clear predictions about ecosystem responses to changing fishing patterns (Walters *et al.* 1997). In this model, although it is demonstrated that intermediate control presents a more optimistic scenario of recovery and stabilization patterns for penaeid shrimps, this trend should be viewed with caution.

4.2.3 Ecosystem impact of management scenario 3

This fishery management scenario simulated the revocation of the trawl ban in the year 2000 (Figure 4.6), as an exploration of a precautionary approach for the Indonesian government's plan for trawl ban revocation (see Chapter 1 and footnote 9). The trawlers fishing rate was increased gradually. Concurrently, it was also simulated that after the trawl ban revocation marked the beginning of the conversion of purse seiners and small scale gears to trawlers (motivated by the historical records of trawling profits). Thus, fishing rates of purse seiners and small scale gears were decreased slightly and gradually.

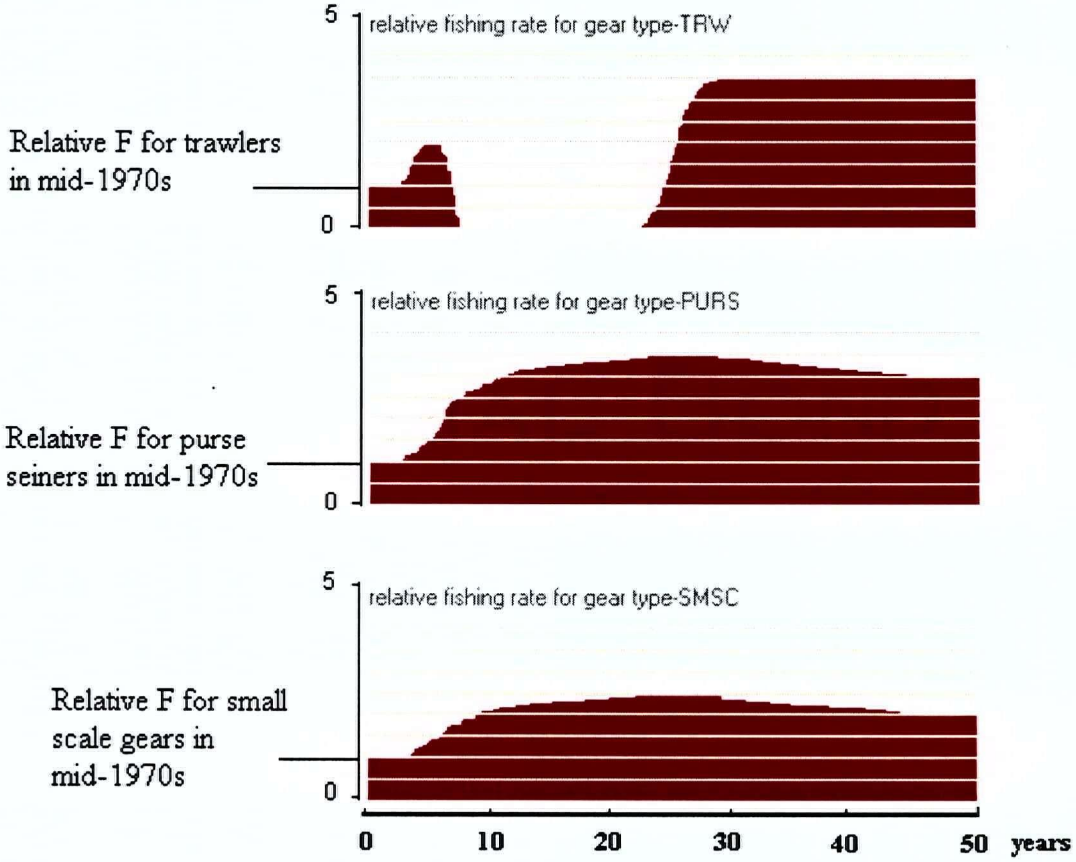

Figure 4.6 Patterns of fishery management scenario 3: "what would happen to the ecosystem and the fisheries if the trawl ban were to be revoked ?". In this scenario, trawling was re-intoduced in the year 2000, and its fishing rate was increased gradually up to three and a half time the level in mid-1970s. However, it was also simulated that those trawlers who would cheat, would start trawling a bit earlier (see footnote 9). Purse seiners, on the other hand, were decreased slightly due to conversion of purse seiners to trawlers. Likewise for small scale gears.

Simulation results (Figure 4.7) show that upon the re-introduction of trawlers in the ecosystem and some decline in purse seine and small scale fishing, several functional groups showed marked sharp decrease or increase in biomass.

Under top-down control (Figure 4.7, upper panel), SAF declined following the trawl ban revocation and the decline of purse seine and small scale gear fishing rate. A similar decline also occurred for clupeoids, miscellaneous pelagics, and macrozoobenthos. Other groups that increased are small demersals and *Decapterus* spp. Thus, at least initially, under the top-down type control, the revocation of the trawl ban had a negative effect on any group that was directly impacted by the trawl such as the SAF and macrozoobenthos. However, a positive effect also occurred for some groups, due to an interplay of indirect cascade effects (as occurred in previous scenarios) and the compensatory decrease in the effort by other gears. The latter could be an indication that the current effort level of purse seine and small scale fishing has been near the maximum, or indeed exceeds the maximum sustainable rates for these groups (see e.g., McElroy 1991, Nurhakim *et al.* 1987, and Dwiponggo 1987).

An interesting trend occurred in the simulated patterns of LBS biomass, in both the top-down and intermediate control. Despite the trawl ban revocation, the rate of decline of LBS (Figure 4.7, upper and middle panels) remained to scenario 2 (Figure 4.5, upper and middle panels). Here, though again possibly an artifact of the method, a pronounced trophic cascade occurred with the LBS and its predators. An indirect impact resulting from cascade effects (the increase of other predators of LBS such as small demersals) compensated the predation relief obtained from the decrease of SAF. In contrast, unlike in scenario 2 (Figure 4.5, lower panel), under the bottom-up control of scenario 3 (Figure 4.7, lower panel), a negative impact, resulting from the revocation of the trawl ban, on LBS was made evident, albeit to a lesser degree.

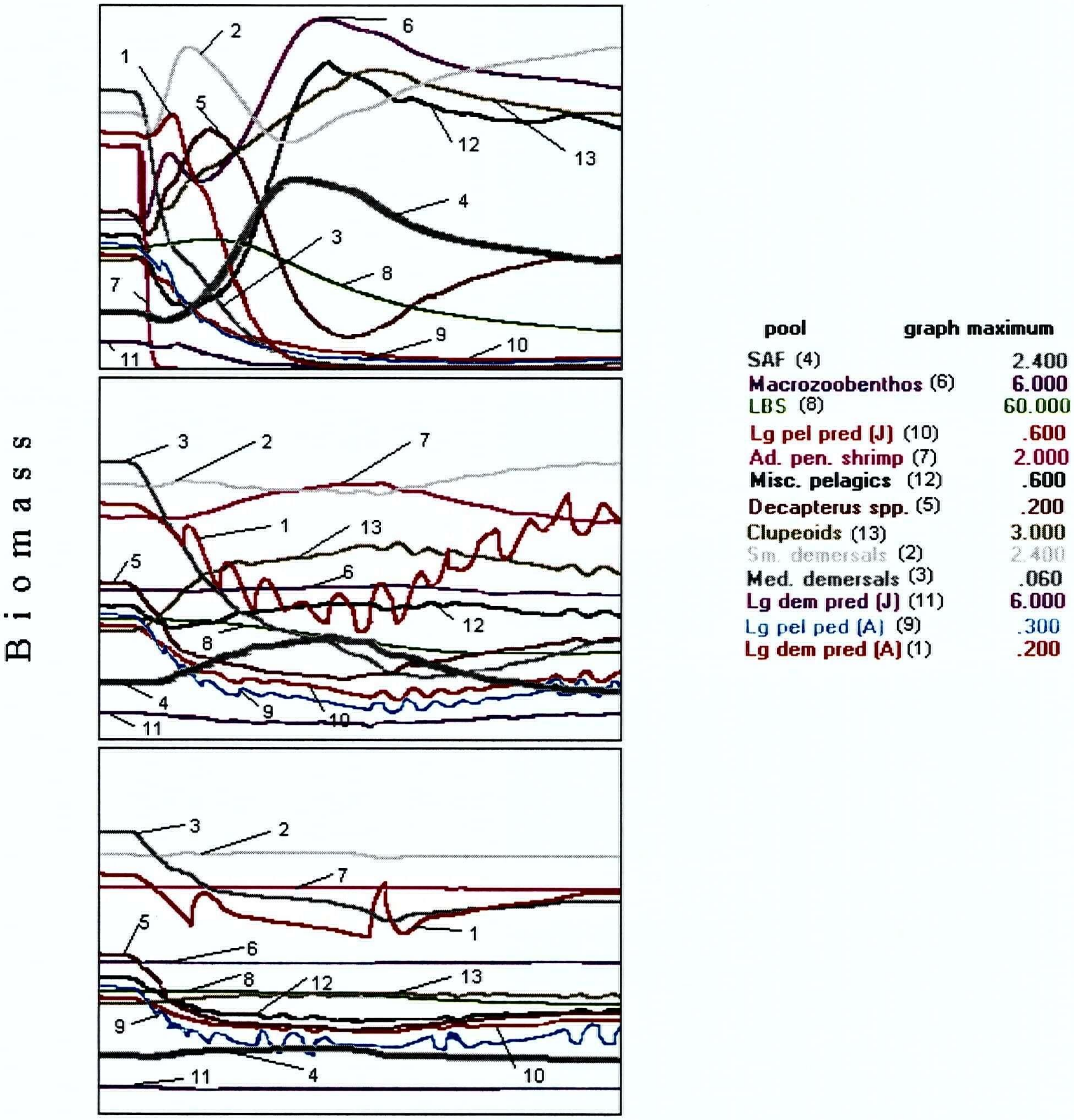

Figure 4.7 Dynamic simulations of fishery management scenario 3, under top-down, intermediate and bottom-up controls. The thick dark gray line is SAF (structure-associated fish), and the other colored lines are biomass of other groups. See text for details and Fig. 4.6 for patterns of scenario 3.

Under intermediate control (Figure 4.7, middle panel), scenario 3 resulted in a decrease in the penaeid shrimps biomass (which increased gradually and constantly to a level higher than its initial biomass, following the trawl ban). A decrease in biomass also occurred with the SAF. In contrast, *Decapterus* spp., large demersal and pelagic predators, medium demersals and small demersals increased, due in part to the decline in purse seiners and small scale gears.

Under bottom-up control (Figure 4.7, lower panel), an increase was also observed in large demersal and pelagic predators, miscellaneous pelagics, medium demersals and *Decapterus* spp. while at the same time, SAF and LBS declined. Thus, despite increased fishing pressure as a result of the revocation of the trawl ban, relief from the fishing pressure of purse seine and small scale gears allowed some fished groups to increase their biomass.

Some insights can be drawn from these ecosystem simulations. Scenario 1 showed that persistently elevated fishing pressure created an 'alternative stable state' (after Holling 1973) that usually involved the loss of at least some parts of the initial food web structure. Here, it is noted that the Java Sea ecosystem fluctuated dramatically (especially in top-down control) for a time, and then achieved a new state of equilibrium. As predicted by Ecopath (see section 3.4.3 in Chapter 3), the Java Sea is moderately stable and resilient. The response behavior of the ecosystem during the simulations showed wide range in the amplitude of the fluctuations and managed to absorb impact caused by continued increase in fishing pressure. Here, with one exception (i.e., penaeid shrimps that were driven to very low biomass), under the assumption of top-down control, many species groups still persist in the system, although at much lower biomass levels by the end of the simulation. In scenario 1, it is also noted that this 'new' alternative stable (or equilibrium) state indicated that small-bodied and lower trophic level biota attain higher biomass (relative to their initial biomass) by the end of the simulation, while large-bodied and high trophic level biota attain lower biomass.

The fluctuations indicated above were even more pronounced in scenario 2, in which the gap or amplitudes between the two trends was wider (i.e., the increase and decrease of biomass level were more marked; see Figure 4.2 and 4.5 for comparison). Hence, with wider fluctuations and larger amplitude, in scenario 2 the ecosystem was less stable relative to scenario 1 (see Holling 1973). It was also shown in scenario 2 that, overall, the 42-year long trawl's absence (i.e., from 1982 to 2024) did not provide enough opportunity for the heavily impacted fished groups to recover. The failure to recover is partly due to the concurrent and continual increase of fishing pressure from purse seiners and small scale gears.

In scenario 3, it was demonstrated that the simulated trawl ban revocation abruptly shifted the 'alternative stable state' attained under scenario 2, to yet another alternative stable state. It was also found that, under scenario 3, small-bodied and lower trophic level biota in general attained higher biomass (relative to scenario 2). Large-bodied and higher trophic level biota attained lower biomass, except with the large demersal and pelagic predators that are heavily influenced by the delay-differential relationships of split pools. Maturity analysis of the Java Sea (see section 3.4.3 in Chapter 3) indicated that the system is moderately mature, relatively stable and hence, moderately resilient to perturbations. Here, it is shown that, at least temporarily for these ~25 years of simulated perturbation (i.e., trawl ban revocation, from year 2000 to 2024), and despite the abrupt shift, the system still managed to absorb the perturbation with no dramatic change in or loss of structural integrity of the system. From the results of the simulation of scenario 3, it can be inferred that indeed small scale gears and purse seiners have been exerting a high fishing pressure. A test of this was modelled by introducing a slight decline in their fishing pressures, with the result that some groups increased immediately[24].

4.2.4 Ecosystem impact of management scenario 4

In this scenario, an approach was applied to examine the predator-prey relationships between LBS and other functional groups in the ecosystem that was simulated through a dramatic increase in the fishing mortality of LBS. The approach taken was to assume intermediate energy flow control and then, by greatly increasing the fishing mortality, eliminate the LBS (Figure 4.8). Fishing rates for other functional groups were not changed The time period for the simulation was also set at 50 years.

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[24] Model validation for this mass-balance and simulation models was applied by comparing the predicted yield values (estimated by Ecosim) with the observed yield values (from DGF database of 1975 to 1994). See Appendix 2 for details.

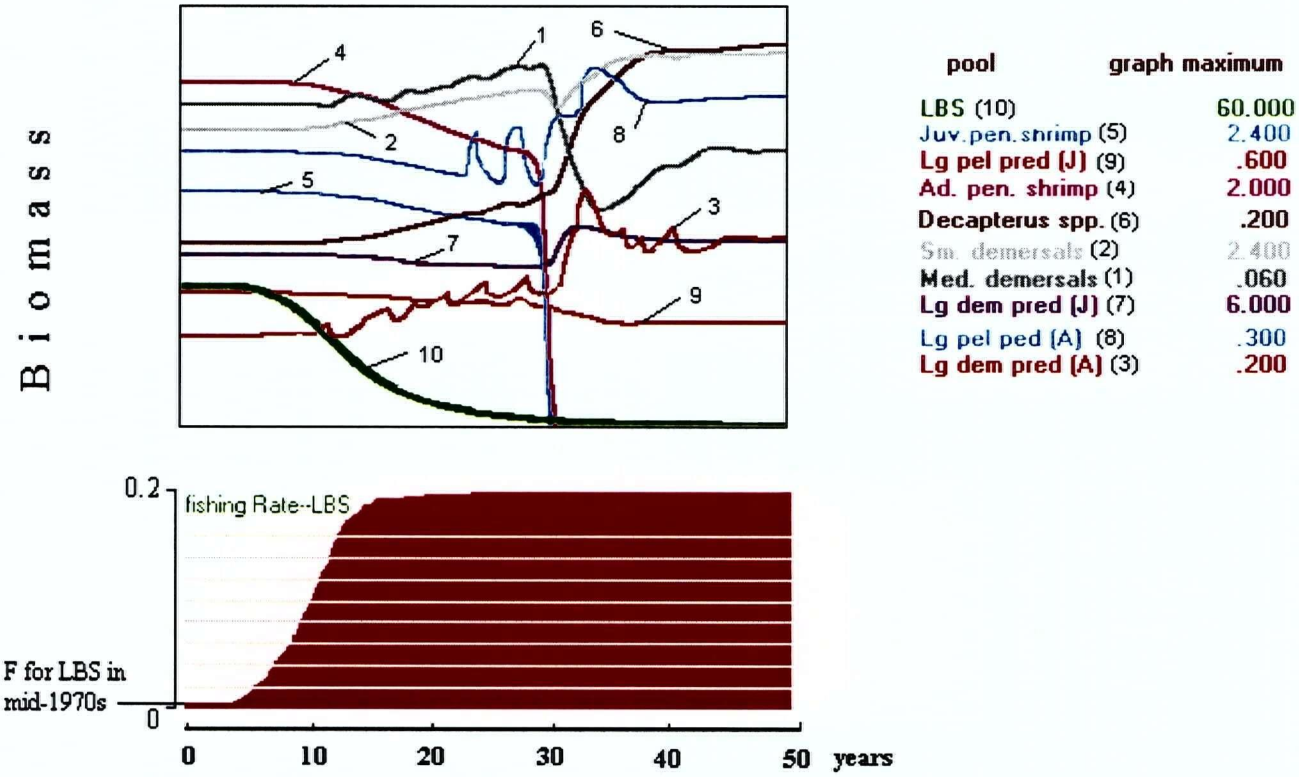

Figure 4.8 Dynamic simulations and patterns of fishery management scenario 4: "what would be the impact to the ecosystem if the LBS was eliminated through dramatic increase of trawling ?". Here, fishing rate on LBS was increased dramatically (due to trawling), approximately 32 times the level in mid-1970s. The thick green line is the biomass of LBS, while the colored lines are biomass of other groups.

The simulation exhibited classic trophic cascade effects (Figure 4.8). Plotting the percent change of all biomass over fishing mortality change (Figure 4.9 and 4.10) indicated that some species were positively impacted, while others were negatively impacted. In Figure 4.9 it was shown that upon the elimination of LBS, biomass increase in *Decapterus* spp. was observed to be over 100%. Similar respond was also observed in adult large demersal predators. High increase in the latter group was attributable to the delay-differential relationship between the juvenile and the adult split pools. Here, the juvenile demersal predators (increased by about 60%) benefit from the decrease of larger demersal fish, such as SAF and medium demersals. Hence, an increased recruitment to the adult pool was observed. Increases in other individual biomass group were less than 40% (Figure 4.9).

Judging from the wide amplitude of the biomass change in *Decapterus* spp., this group can be considered to be one of the most unstable groups when the LBS group is eliminated. *Decapterus* spp., as small shoaling pelagic fish are short-lived fish and have high productivity. Csirke (1988) indicated that small shoaling pelagic fish like *Decapterus* spp. are known to be sensitive to changes, especially in terms of its density-dependence of the catchability coefficient. In scenario 4, it was shown that even trophic cascade alone may highly impacted *Decapterus* spp.

The simulation also showed that penaeid shrimps were the most negatively impacted group by the elimination of LBS (Figure 4.10). Both adult and juvenile penaeid shrimps biomass decreased up to 100% compared to its initial biomass level, effective extermination. Other groups that were severely impacted were SAF (about -95%), and miscellaneous pelagics (about -65%). The remainder of the groups decreased in the range of 10% to 30% (Figure 4.10).

This scenario also showed that the relationships between SAF, small demersals and LBS, as studied by Sainsbury *et al.* (1997)[25], can be emulated in this model, using merely predator-prey relationship. As LBS continued to decrease through trawler-induced mortality, SAF biomass decreased and small demersals increased.

Scenario 4 also provided some insights that the emergence of an alternative 'stable state' or states, as predicted by Holling (1973), may unfortunately involve the gain of economically low-valued species (such as small demersals and *Decapterus* spp.), at the cost of the loss of some economically high-valued species (such as penaeid shrimps and SAF in this case). This would negatively affect the fisheries.

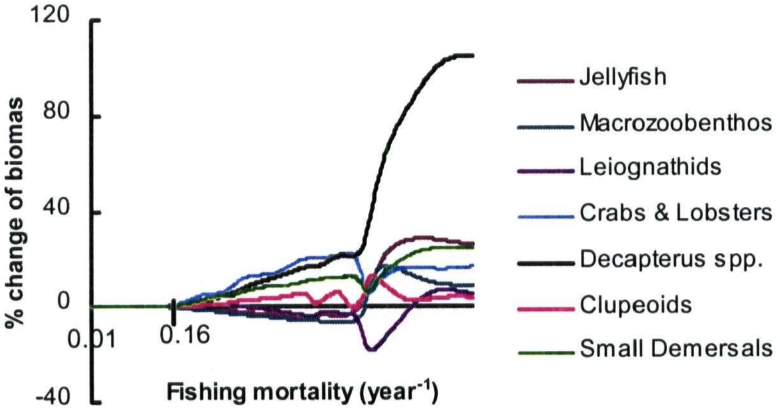

Figure 4.9 Plots of % positive change in biomass versus change in fishing mortality, from scenario 4.

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[25] In their experimental research on trawl-induced habitat modification in an Australian multispecies trawl fishery, Sainsbury *et al.* (1997) found that the abundance of economically high-valued lethrinids and lutjanids declined with the development of trawl fishing, and that the abundance of small-bodied and economically low-valued fish such as *Nemipterus* and *Saurida* in particular increased. Similarly, they also found that in the trawled area, large epibenthos (sponges, > 25 cm, where *Lethrinus* and *Lutjanus* are usually abundantly found) had a very low abundance, and vice versa. In the present study, the economically high-value Lethrinids and Lutjanids are represented by SAF, while the economically low-value fish are represented by small demersals. The large epibenthos are represented by LBS.

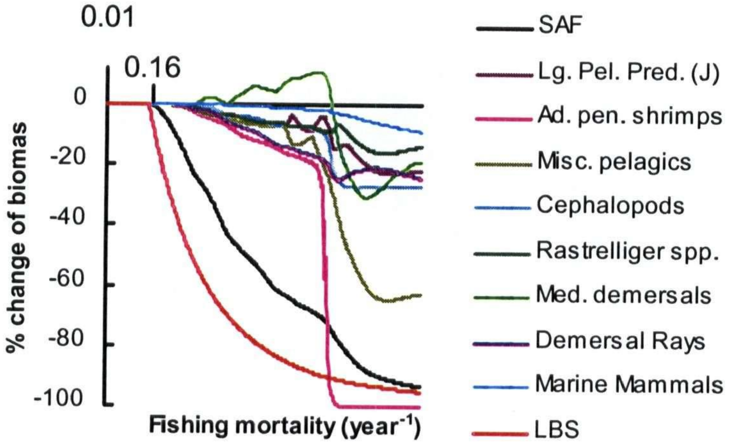

Figure 4.10 Plots of % negative change in biomass versus change in fishing mortality from scenario 4.

4.2.5 Testing the working hypotheses

In Chapter 1, four working hypotheses were put forward. Here, these four hypotheses were tested using results from all management scenarios.

Hypothesis 1: due to the trawl ban, there was a dramatic shift in ecosystem structure, from demersal to pelagic species.

This hypothesis was not supported. Scenario 2 (Figure 4.5) suggested that after the trawl ban, the ecosystem progressed so that large size organisms declined, and small size organisms increased, regardless of whether they were a demersal or pelagic group. There was no evidence to suggest that pelagic species would become the dominant groups in the ecosystem. Though landing statistics are dominated by pelagic species, this was true even before the trawl ban. This suggests only that gear specialized for pelagic species is more successful, either because of numerical abundance, or because it is more efficient.

Hypothesis 2: the impact of trawling on living bottom structure (LBS) and several groups of demersal species that depend on it can be emulated with trophic relationship.

This hypothesis was strongly supported by scenario 4 (Figure 4.8, 4.9 and 4.10). In this scenario, LBS dramatically decreased, negatively impacting the SAF. The hypothesis was also supported by scenario 3 (Figure 4.7), wherein the biomass of SAF and other trophically dependent groups declined after the trawl ban was revoked.

Hypothesis 3: right after the ban, there was a short period of rapid growth in pelagic populations (due to the transfer of primary production from the demersal to the pelagic subsystems), followed by a decline due to increasing pelagic fishing.

This hypothesis was only supported by smaller size pelagic fish. In scenario 2, the biomass of larger pelagic species continued to decline with the increasing level of purse seine fishing, regardless of the trawl ban. Smaller pelagic species such as clupeoids, however, continuously increased. In scenario 2, only *Decapterus* spp. exhibited the trend proposed in this hypothesis. However, since the trophic cascade effect was very strong, it is very likely that this trend was a result of reduced predation on *Decapterus* spp., as this response also occurred in scenario 1.

Hypothesis 4: demersal populations recovered after the ban, except for those species that depend on bottom living structures to survive.

This hypothesis was supported by the demersal populations, although after a short period, these populations had to succumbed to the increasing fishing pressure from the small scale and purse seine fisheries. Results from scenario 2 (Figure 4.5) and 3 (Figure 4.7) suggest that the levels of fishing attributable to small scale gears and purse seiners were too strong to allow demersal populations to recover after the ban. In contrast, this hypothesis was not supported by the species that depend on living bottom structures (*viz.,* SAF). In here, due to indirect cascade effects, the SAF recovered faster than other demersal functional groups. In this case, the removal of predation pressure and fishing pressure combined were enough to overcome the loss of food resources associated with the loss of the LBS.

4.2.6 Economic implications under different policy regimes

Here, as explained in Chapter 1, the impacts of both "status-quo" (scenario 2) and "lift the ban" (scenario 3) options, under differing discount rates, were translated into their respective discounted net economic values.

The ecosystem/economic analysis were applied only to the following groups: large herbivorous zooplankton, living bottom structure-associated biota, adult penaeid shrimps, miscellaneous pelagics, leiognathids, crabs and lobsters, cephalopods, *Decapterus* spp., *Rastrelliger* spp., clupeoids, small demersals, medium demersals, demersal rays, adult large pelagic predators, and adult large demersal predators.

The International Financial Statistics Yearbook (International Monetary Fund 1998) reported that in 1997, Indonesian GDP (Gross Domestic Product of all sectors) was Rp. 624,337 billions, or approximately US$ 208 billions. Fisheries sector (for the whole Indonesia) accounted for approximately 2% of the GDP (Dr. A. Fauzi, Centre for Coastal Resource Management, Bogor Agricultural University, *pers. comm*, 1999), resulting a value of US$ 4.2 billions for 1997 Indonesian Fisheries GDP. Given the level of 1997 Indonesian Fisheries GDP and assuming normal economy scenario, under the status-quo option the predicted average annual total gross revenue from the Java Sea fishery for the period of 1997 to 2024 (Rp. 3,038 billions) was 24% of the Indonesian Fisheries GDP. For the same period, the Java Sea fishery's contribution to the Indonesian Fisheries GDP was predicted to be 27% by adopting the 'lift the ban' option (Rp. 3,338 billions). On the other hand, by taken into account the current economic crisis into the prediction (i.e., shock scenario), under the status-quo option the predicted average annual total gross revenue from the Java Sea fishery for the period of 1997 to 2024 (Rp. 10,129 billions) was 81% of the 1997 Indonesian Fisheries GDP. For the same period, the Java Sea fishery's contribution to the Indonesian Fisheries GDP was predicted to be 89% by adopting the 'lift the ban' option (Rp. 12,487 billions).

From Table 4.3, it was evident that the net benefits that were obtained were higher in lower discount rates. This suggests that the higher the discount rate (r), the less time fishers would like to wait to get the benefits. Therefore, they would exert as much capital (*viz.,* fishing effort) as they could to fish. This behavior might occur under extreme economic situation such as the one that exists in Indonesia right now. In other words, no matter how good and well-meaning the regulations are, when fishers are trapped under such circumstances, destructive fishing behaviors will occur.

Table 4.3 Discounted net economic benefit (US$ x 10^6), under both 'status-quo' and 'lift the ban' options; and under three different discount rates. See text for details [a] .

	Status-quo option			**Lift the ban option**		
Year	**r = 3%**	**r = 5%**	**r = 10%**	**r = 3%**	**r = 5%**	**r = 10%**
1995	369	248	96	369	248	96
1996	360	238	87	360	238	87
1997[b]	360	233	82	362	234	82
1998	349	222	74	360	229	77
1999	341	213	68	366	228	73
2000[c]	334	204	62	380	233	71
2001	326	196	57	384	230	67
2002	312	184	51	373	219	61
2003	305	176	47	364	210	56
2004	297	168	43	349	198	50
2005	283	158	38	334	186	45
2006	280	153	35	325	177	41
2007	272	146	32	314	168	37
2008	260	137	29	301	158	33
2009	257	132	27	293	151	30
2010	250	126	24	281	142	27
2011	239	118	22	273	135	25
2012	236	115	20	264	128	22
2013	230	109	18	253	120	20
2014	220	103	16	246	115	18
2015	217	99	15	237	109	16
2016	211	95	14	227	102	15
2017	202	89	12	219	97	13
2018	199	86	11	210	91	12
2019	194	82	10	203	86	11
2020	185	77	9	200	83	10
2021	183	75	9	192	78	9
2022	178	71	8	186	74	8
2023	170	67	7	183	72	8
2024	168	65	6	177	68	7
SUM =	6,008	3,031	662	6,768	3,432	712

[a] Conversion rate from US$ was Rp. 3,000 for normal scenario, and Rp. 10,000 for shock scenario.

[b] This period marked the start of trawling activities conducted by trawlers that cheated.

[c] Trawl ban revocation was simulated to be officially implemented in this year.

The higher estimated total discounted net benefit obtained under the lift the ban option as shown in Table 4.3 was indeed the reason why this issue started in the first place. By opting for the lift the ban option, there would be an increase in the total discounted net benefit of 12.7%, 13,2% and 14.5%, for r = 3%, r = 5% and r = 10%, respectively. In both normal and shock scenarios, the predicted total discounted net benefit were similar when they were expressed in US$ (Table 4.3). However, when converted to Rupiah, using the current average 'economic crisis' exchange rate of Rp.10,000, the resulting nominal values are very large. Here is one of the issues that was put forward by those who favor the revocation. Many argue that the high price that would be obtained from the trawl landing (notably from shrimps) - should the trawl ban be revoked - may be used to empower the fishing communities, in particular small-scale fishers (Aziz 1996). Nevertheless, without considering other related aspects surrounding trawl fishing, either ecological, social, political and even more in-depth economic analysis, this expectation may only serve as a highliner illusion.

To better illustrate the impact of the trawl ban revocation from the economic perspective, all predicted annual discounted net benefit from 'status-quo' and 'lift the ban' options under normal economy scenario and all discount rates were plotted (Figure 4.11). Here it is shown that immediately after the trawl ban revocation, there will be an increase of the discounted net benefit of approximately 14% - 15%. However, given the food web interactions in the ecosystem coupled with the effect of discounting, the simulations showed that by re-introducing the trawl, eventually the level of the discounted net benefit of the lift the ban option approached the level of the status-quo's. In fact, under the 10% discount rate, the simulation showed that the two lines intersect at approximately the 25th year (i.e., the end of year 2024).

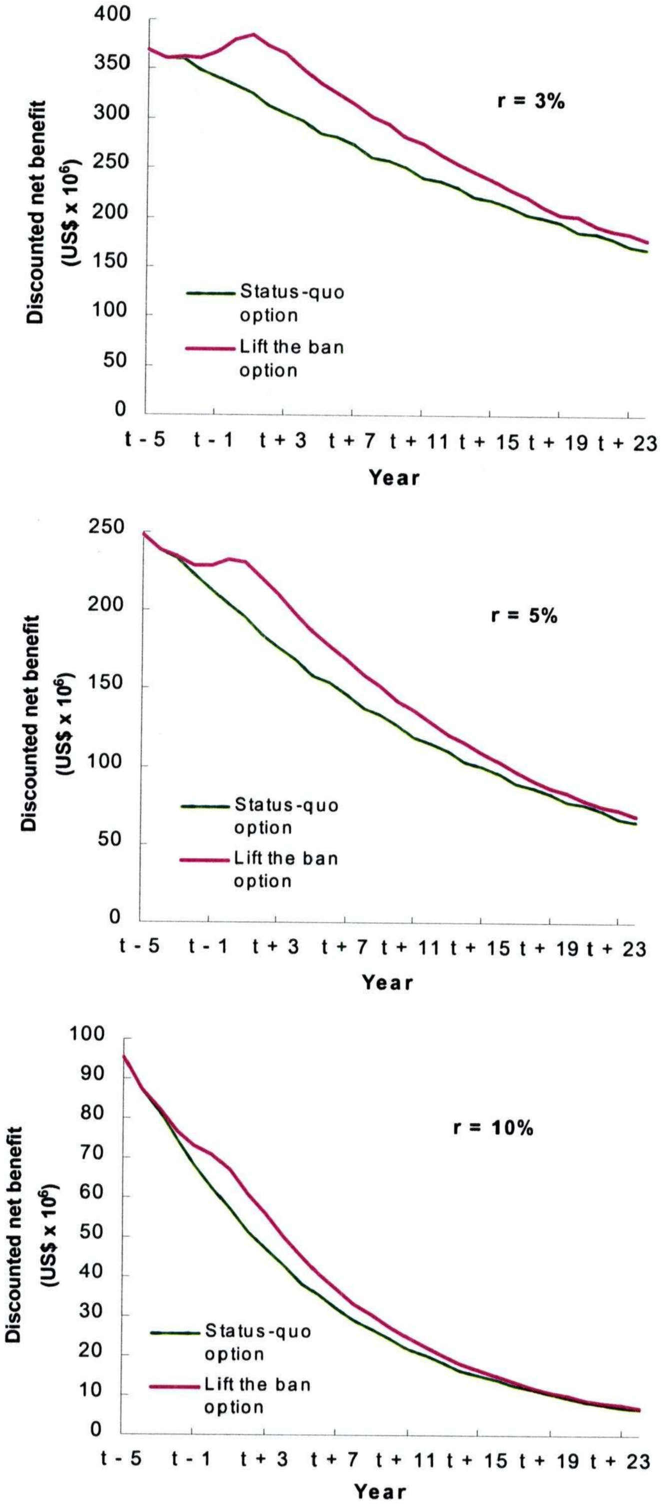

Figure 4.11 Plots of discounted net benefits from the beginning of year t - 5 (i.e., 1995) to the end of year t + 24 (i.e., 2024) based on Table 4.3. Discount rates were varied from 3% to 10%.

Though this economic analysis is very elementary, the main aim is to provide some insights on the potential economic impacts for the Java Sea fisheries, based on ecosystem dynamics perspective, should the trawl ban be revoked. Results showed that with a proper account of food chain interactions, the net benefit that would be obtained from lifting the trawl ban would last only for a relatively short period. As discussed earlier, revoking the trawl ban would reduce ecosystem stability. Thus, while some gains were obtained by implementing the trawl ban revocation, they only lasted for a while. Meanwhile the repercussion of the fishing impact towards ecosystem integrity might further reduce the economic viability of the already highly-exploited fishery (see e.g., Dwiponggo 1987; McElroy 1991).

The analysis could be elaborated upon, for example by employing actual price and cost for every year, rather than using a projection of constant 1974 price and cost proportion based on a study in the early 1970s. In addition, the present analyses could be extended to investigate the impact of government subsidies on both the ecology and economics of the Java Sea fishery. Given the fluctuation of the country's economy, these two extensions will very likely reveal a more pronounced negative impact to the economics of the Java Sea fishery as a result of the revocation of the trawl ban, than found in this analysis.

CHAPTER 5:

SUMMARY AND CONCLUDING COMMENTS:

QUO VADIS THE 1980 INDONESIAN TRAWL BAN ?

Lack of law enforcement has always been a chronic problem in almost every aspect of the public domain in Indonesia. This includes fisheries management. Therefore, monitoring the implication of a policy and testing how well it works in different situations are unheard of. This, unfortunately, has led to a number of policies being implemented without adequate scientific research or public participation - resulting an atmosphere of reactive policy making rather than proactive policy making. The 1980 trawl ban can be considered as one of those reactive policies.

As described in Chapter 1, the trawl ban has now been under the spotlight for some time. Although only a few scientific studies was conducted prior to the trawl ban[26], perhaps, as radical and reactive as it was, the ban was a blessing in disguise. After all, the artisanal fishers needed an instant remedy for the turmoil that engulfed them at that time. Now, nineteen years later, the socio-political situation in the country has changed dramatically. So have the state of the marine ecosystem and the sophistication of fishing technology. The Indonesian people are now more aware about what role they can have in policy making. Marine environmental degradation (especially in the western part of the country) is not news anymore (e.g., Soegiarto 1981; Anggraini 1990; Cesar *et al.* 1997). Advances in fishing technology have been continuously planned and pursued (e.g., Potier and Nurhakim 1995; Petit *et al.* 1995). Consequently, calls for trawl ban revocation (e.g., Aziz 1996; *Kompas Online* [Jakarta] 1998) cannot be answered without thorough studies of the potential implications both on the marine environment and the people that depend on it.

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[26] A study by Martosubroto (1982) that focused on the demersal resources of the Java Sea from 1975 to 1979, indicated that fishing pressure on the demersal resources in the Java Sea prior to the ban was not as heavy as in the Gulf of Thailand. The study also emphasized that any decline of the demersal stocks, including that of shrimps, had not been the sole responsibility of the trawlers; and that increased fishing in that period has had only a minor effect on the trophic structure of the demersal communities of the sea.

The controversy surrounding the trawl ban revocation gave rise to the research question in Chapter 1: *"is it really biologically safe to revoke the trawl ban, and furthermore, would it benefit the fisheries?"*.

The Java Sea served as a good choice for this research, as it is one of the most heavily exploited, and one of the best studied regions in the country. The first rationale is important in forming the basis for a sound research approach in ecosystem sustainability. Also, it is important to avoid being faced with 'no data' or 'not enough data' - a common problem for developing countries. Lastly, considering the skewed population distribution in the archipelago, selecting the Java Sea for this study even gave it added value, since the Java Sea also serves as the source of fish for the majority of coastal communities in the western part of the country.

From the research question, an objective was formulated to direct this research, i.e., to evaluate the effect of the 1980 trawl ban on the sustainability of the Java Sea ecosystem. More specifically: (1) to explore the behavior of a marine ecosystem subjected to drastic human perturbation (i.e., trawling), evaluate its recovery after the removal of the perturbation, and then to see whether it is ecologically defensible to re-impose a similar perturbation; and (2) to perform an economic implication analysis of this trawl ban revocation for the Java Sea fisheries in order to evaluate options for policy makers reviewing the trawl ban in Indonesia. As the research question suggests analyses at an ecosystem scale, an appropriate suite of methods was applied, comprising Ecopath modelling (Christensen and Pauly 1992, 1996), Ecosim modelling (Walters *et al.* 1997) and ecosystem/economic impact analysis (Sumaila 1998; Pitcher *et al.* in press).

Chapter 2 reviewed the Java Sea fisheries. The picture taken from this chapter was of multispecies (214 species of finfish alone) and multigear (28 gear types) fisheries, supporting mostly artisanal (small-scale) fishers. The overall trend over the past twenty five years indicated that fisheries are continuing to grow in terms of the landings, the number of gears and vessels (especially, the outboard engine powered vessels), and the number of fishers (in particular, the part-time small scale fishers). Easy entry into the small scale fishery sector has been made possible by low capital requirements. This trend is disturbing. Despite the increase in landings, the quality of the landings seem to be declining, as more and more smaller fish (i.e., lower trophic level and usually less expensive) are caught over the years, such as the clupeoids (e.g., anchovies, sardines, and sardinellas) and small demersals (e.g., ponyfish and lizardfish). The symptoms of the trends observed from the Java Sea fishery over those years fit the causal patterns of Malthusian overfishing (Pauly 1997).

To test the hypothesis of Pauly *et al.* (1998) that fishing globally has moved deeper down the food web, the mean trophic levels of the landings from the Java Sea, from 1974 to 1994 were estimated and plotted. The trend revealed a fluctuations in mean trophic levels (TL), but over the years, a gradual and a very slow shifting to lower mean TL (Figure 5.1).

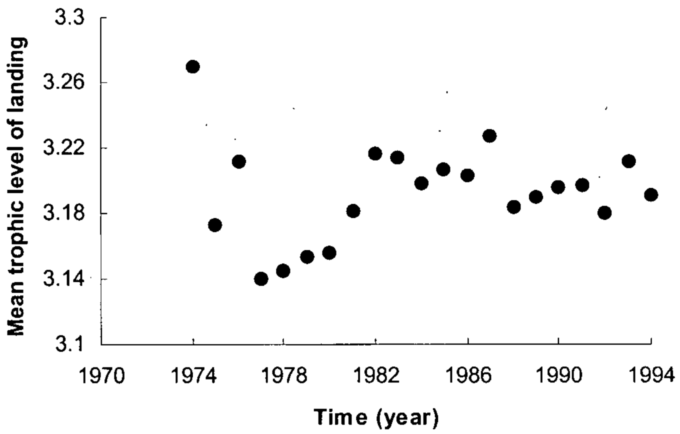

Figure 5.1 Trend of mean trophic levels in Java Sea fishery landings, from 1974 to 1994. Landings from juvenile predator fish, LBS and marine mammals are not reported by the statistics, hence, are not included in this trend (data from DGF, various years; mean TLs were calculated as in Pauly *et al.* [1998], with TL estimates derived from the model presented in Chapter 3).

From the plots, it is unclear what caused the big amplitude change in mean trophic levels between 1974 to 1977. Probably it was because the early stages of fishery development concurrently employed trawls, purse seines and small scale gears. Between 1977 and 1982, mean trophic levels increased gradually, coincident with a rapid increase in purse seine and small scale gear fisheries and with the rapid decline of trawl fishery which started in 1980 and culminated in 1982. Further increases in purse seine and small scale gear fisheries (see

Figure 2.9 in Chapter 2) may well account for fluctuations of mean trophic levels between 1983 and 1987. This pattern of fluctuations and an overall shift to a lower mean TL (compared to the previous year) was repeated in the period of 1988 to 1991, and again from 1992 to 1994. Compared to the mean trophic level at the initial phase of fishery development in the mid-1970s, the mean trophic level in 1994 had dropped by 0.1. The indication is even clearer, when mean trophic levels are plotted against the level of catch (Figure 5.2). This is an indication that fishing-down-the-food-web (Pauly *et al.* 1998) may just have started for the Java Sea fisheries.

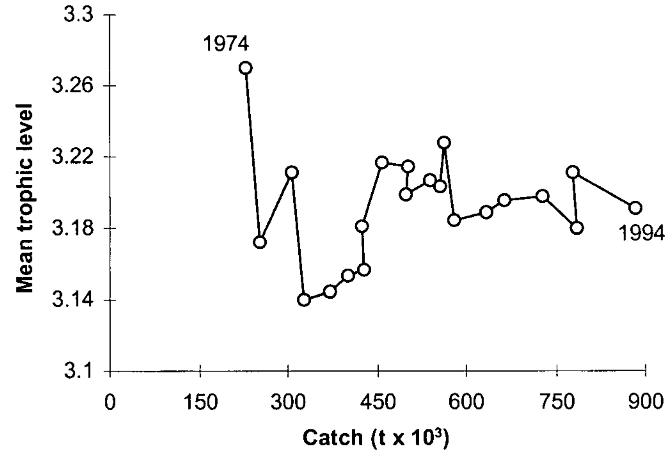

Figure 5.2 Plots of mean trophic level of landings versus the landings of the Java Sea fishery, from 1974 to 1994. Landings from juvenile predator fish, LBS and marine mammals are not included in this plot (data from DGF, various years; mean TL were calculated as in Pauly *et al.* [1998], based on TL estimates derived from the model presented in Chapter 3).

Chapter 3 presents the first fully parameterized mass-balanced Ecopath model (Christensen and Pauly 1992, 1996) for the Java Sea. The building of this Ecopath model, the first of its kind in Indonesia, required the gathering and the utilization of a large amount of old and new data about Indonesian marine resources that was scattered between various research and government institutes in the country, and perhaps underutilized (see critiques in Pauly 1986, 1996).

Ecopath allows us to pool and integrate diverse knowledge about small pieces of the ecosystem, so as to increase overall knowledge of the whole system. For the Java Sea, which supports a tropical multispecies and multigear fishery, this means a better understanding of how the entire ecosystem works. The resulting Ecopath model comprised 28 functional groups and suggests that the Java Sea ecosystem in mid-1970s was moderately mature and relatively stable. This implies that the ecosystem should be moderately resilient to perturbations. The model also indicated that the fishery in the mid-1970s targeted apex predators and the impact of the fishery was low to moderate in comparison with the fisheries of other aquatic systems (Christensen and Pauly 1993).

In Chapter 4, a multispecies analyses using the Ecopath model as input was done by using Ecosim, a dynamic multispecies simulation model (Walters *et al.* 1997). Ecosim modelling allows us to ask "what-if" questions. Four "what-if" scenarios were simulated in order to explore the research objectives: (1) what would have happened to the ecosystem if the trawl ban had never been implemented ? (2) what can be expected to happen to the ecosystem and the fisheries if the status-quo is maintained ? (3) what would happen to the ecosystem and the fisheries if the trawl ban were to be revoked ? and (4) what would be the impact to the ecosystem if the living bottom structure (LBS, composed of sponges, gorgonians, sea fans, and other living organisms that are attached to the sea floor) was eliminated through dramatic increase of trawling ? Ecosystem dynamics were explored in relation to those "what-if" questions.

Results suggested that persistently elevated fishing pressure created an 'alternative stable state' (Holling 1973) that involved the loss of at least some parts of the initial food web structure. In scenario 1, it is also noted that this 'new' alternative stable (or equilibrium) state indicated that small-bodied and lower trophic level biota attain higher biomass (relative to their initial biomass) by the end of the simulation, while large-bodied and higher trophic level biota attain lower biomass.

Scenario 2 produced more impact to the ecosystem. Here, higher amplitude fluctuations of individual biomass were observed. It was also shown that although the trawl ban managed to let the shrimps biomass recover slightly, overall, it has not provided enough opportunity for most of the heavily impacted fished groups to recover. The failure to recover is partly due to the concurrent and continual increase of fishing pressure from purse seiners and small scale gears.

Scenario 3 demonstrated that the simulated trawl ban revocation abruptly shifted the 'alternative stable state' attained under scenario 2, to yet another alternative stable state, in a relatively short time. Additionally, under scenario 3, small-bodied and lower trophic level biota generally attained higher biomass (relative to scenario 2). Large-bodied and higher trophic level biota declined, except for the large demersal and pelagic predators that were affected by delay-differential effects. In this scenario, it was also observed that, as the trawl fishing increased, the amount of small demersals (which feed on shrimps) also increased. Thus, in addition to trawling, over time shrimp biomass decreased due to predation. Here, it is shown that, at least temporarily for these 25 years of simulated perturbation, and despite the abrupt shift, the system still managed to absorb the perturbation with no sudden dramatic change in or loss of structural integrity. From scenario 3, it can also be inferred that small scale gears and purse seines have indeed been exerting a high fishing pressure.

Scenario 4 indicated that elimination of LBS through trawl-induced mortality has a profound impact on the integrity of the whole ecosystem. This scenario also provided some insights that the emergence of an alternative 'stable state' (Holling 1973) due to increased trawling, may involve the increase of economically low-valued species, at the expense of the loss of some economically high-valued species.

Finally, using the ecosystem dynamics predicted by Ecopath/Ecosim, an economic implication analysis (Sumaila 1998; Pitcher *et al.* in press) was attempted. Results indicated that short-term gains were predicted for the trawl ban revocation. Eventually, however, a revocation of the trawl ban would create a greater impact on ecosystem integrity, and further reduce the economic viability of the already highly-exploited fishery.

The disastrous physical impact of trawling activities on marine ecosystems have been well documented (see e.g., Jones 1992; Sainsbury *et al.* 1997; Engel and Kvitek 1998). A recent study by Watling and Norse (1998) went so far as to compare the destructive effects of trawling on the seabed to forest clearcutting. The socio-political impacts of the trawl (and its ban) in Indonesia have also been well observed and documented (Bailey 1986, 1987c, 1988, 1997; Taryono 1998). These studies indicated that the trawl ban has been well received and institutionalized among the small scale fishers. Additionally, the ban has served as a confirmation of traditional resource use rights of small scale fishers in Indonesia. The positive impact of the trawl ban on the profitability of small scale fishers, at least at the aggregate level, has also been documented (Chong *et al.* 1987). Pauly (1988) re-iterated the inefficiency of using the traditional legislative or administration means in Southeast Asia to 'disentangle' the small scale inshore fisheries from the trawl fisheries. Drawing from these research and the present study, it can be concluded that revoking the trawl ban will simply mean heralding another tragedy of the commons in the Indonesian fisheries.

REFERENCES

Aliño, P.M., L.T. McManus, J.W. McManus, C.L. Nañola, Jr., M.D. Fortes, G.C. Trono, Jr. and G.S. Jacinto. 1993. Initial parameter estimations of a coral reef flat ecosystem in Bolinao, Pangasinan, Northwestern Philippines; p. 252-258. *In* V. Christensen and D. Pauly (eds.). Trophic models of aquatic ecosystems. ICLARM Conf. Proc. 26.

Allen, K.R. 1971. Relation between production and biomass. *J. Fish Res. Board Can.*, 28(10):1573-1581.

Amaratunga, T. 1983. The role of cephalopods in the marine ecosystem. p. 379-415. *In* J.F. Caddy (ed.). Advances in assessment of world cephalopod resources. *FAO Fish. Tech. Paper* 231.

Amir, I. 1991. Fauna sepon (Porifera) dari terumbu karang Genteng Besar, Pulau-pulau Seribu [Sponges of the Genteng Besar reefs, Thousand Islands]. *Oseanologi di Indonesia* 24: 41-54. [In Indonesian].

Amir, I. 1992. A comparison of sponge fauna of exposed and sheltered reef flats in eastern Indonesia. *Mar. Res. Ind.* 28: 1-12.

Amir, I. and A. Budiyanto. 1996. Mengenal spons laut (Demospongiae) secara umum [General introduction to marine sponges]. *Oseana* 2: 15-31. [In Indonesian].

Anggraini, E. 1990. Potensi dan sistem pengusahaan ikan-ikan karang di zona penyangga Taman Nasional Laut Kepulauan Seribu, Daerah Khusus Ibukota Jakarta [Biological observations, and harvesting and marketing of reef fishes in the buffer zone of Thousand Islands Marine National Park, DKI Jakarta]. Laporan Praktek Ketrampilan Lapang. Fakultas Perikanan, Institut Pertanian Bogor. 133 p. [In Indonesian].

Arreguín-Sánchez, F., E. Valero-Pacheco and E.A. Chávez. 1993. A trophic box model of the coastal fish communities of the southwestern Gulf of Mexico. p. 197-205. *In* V. Christensen and D. Pauly (eds.). Trophic models of aquatic ecosystems. ICLARM Conf. Proc. 26.

Ayodhyoa, H. and Diniah. 1989. Handbook Perikanan Indonesia [Handbook of Indonesian Fisheries]. Fakultas Perikanan, Institut Pertanian Bogor. 148 p. [In Indonesian].

Aziz, K.A. 1996. Pukat Harimau sebagai salah satu alat pemanfaatan sumberdaya perikanan [Trawls as fishing gears]. Unpublished. Presented at Scientific Discussion on Fisheries Resources Utilization in the Indonesian EEZ and the Problems of Trawl Fishing. Bogor, January 8, 1996. 8 p. [In Indonesian].

Bailey, C. 1987a. Overview of Indonesian marine capture fisheries. p. 1-9. *In* C. Bailey, A. Dwiponggo and F. Marahudin (eds.). Indonesian Marine Capture Fisheries. ICLARM Stud. Rev. 10.

Bailey, C. 1987b. Marine fisheries management and development: policies and programs. p. 89-102. *In* C. Bailey, A. Dwiponggo and F. Marahudin (eds.). Indonesian Marine Capture Fisheries. ICLARM Stud. Rev. 10.

Bailey, C. 1987c. Government protection of traditional resource use rights: the case of Indonesian fisheries. p. 292-308. *In* D.C. Korten (ed.). Community management: Asian experience and perspective. Kumarian Press, West Harford.

Bailey, C. 1988. The political economy of marine fisheries development in Indonesia. *Indonesia* 46: 25-38.

Bailey, C. 1997. Lesson from Indonesia's 1980 trawler ban. *Mar. Pol.* 21(3): 225-235.

Bailey, C. and A. Dwiponggo. 1987. Indonesian marine fisheries: structure and change. p. 64-88. *In* C. Bailey, A. Dwiponggo and F. Marahudin (eds.). Indonesian Marine Capture Fisheries. ICLARM Stud. Rev. 10.

Bailey, C., A. Dwiponggo and F. Marahudin. 1987. Indonesian Marine Capture Fisheries. ICLARM Stud. Rev. 10. 196 p.

Barnes, R.D. 1982. Invertebrate Zoology. Fourth edition. Holt-Saunders International Editions, Tokyo. 1089 p.

Barnes, R.H. 1974. Lamalerap: a whaling village in eastern Indonesia. *Indonesia* 17: 136-159

Brett, J.R. and T.D.D. Groves. 1979. Physiological energetics. p. 279. *In* W.S. Hoar, D.J. Randall and J.R. Brett (eds.). Fish Physiology, Vol. 8. Bioenergetics and Growth. Academic Press, New York.

Bruyns, W.F.J.M. 1971. Field guide of whales and dolphins. N.V. Uitgeverij V.H. C.A. Mees., Amsterdam. 258 p.

Bundy, A. 1997. Assessment and management of multispecies, multigear fisheries: a case study from San Miguel Bay, the Philippines. Ph.D. thesis. University of British Columbia, Vancouver, B.C., Canada. 396 p.

Butcher, J. 1996. The marine fisheries of the Western archipelago: towards an economic history, 1850 to the 1960s. p. 24-39. *In* D. Pauly and P. Martosubroto (eds.). Baseline studies of biodiversity: the fish resources of Western Indonesia. ICLARM Stud. Rev. 23.

Carpenter, S.C. and J.F. Kitchell. 1993. The Trophic Cascade in Lakes. Cambridge University Press, New York. 385 p.

Carpenter, S.C., J.F. Kitchell, and J.R. Hodgson. 1985. Cascading trophic interactions and lake productivity. *Bioscience* 35: 634-639.

Cesar, H., C.G. Lundin, S. Bettencourt and J. Dixon. 1997. Indonesian coral reefs: an economic analysis of a precious but threatened resource. *Ambio* 26(6): 345-350.

Cheal, A.J. and N.J. Gales. 1992. Growth, sexual maturity and food intake of Australian Indian Ocean Bottlenose Dolphins, *Tursiops truncatus*, in captivity. *Aust. J. Zool.* 40: 215-223.

Chong, K.C., A. Dwiponggo, S. Ilyas and P. Martosubroto. 1987. Some experience and highlights of the Indonesian trawl ban: bioeconomics and socioeconomics. p. 458-477. *In* Indo-Pacific Fishery Commission (IPFC). Symposium on the exploitation and management of marine fishery resources in Southeast Asia. Darwin, 16-19 February 1987. Regional Office for Asia and the Pacific (RAPA), Food and Agriculture Organization of the United Nations, Bangkok.

Chong, V.C. and A. Sasekumar. 1981. Food and feeding habits of the White Prawn *Penaeus merguensis*. *Mar. Ecol. Prog. Ser.* 5(2): 185-191.

Christensen, V. 1994. On the behavior of some proposed goal functions for ecosystem development. *Ecol. Modelling* 75/76: 37-49.

Christensen, V. 1995. Ecosystem maturity - towards quantification. *Ecol. Modelling* 77: 3-32.

Christensen, V. 1998. Fishery-induced changes in a marine ecosystem: insight from models of the Gulf of Thailand. *J. Fish Biol.* 53 (Supplement A): 128-142.

Christensen, V. 1998. The new Ecopath with Ecosim version 4.0 alpha. p. 9 - 10. *In* D. Pauly (ed.). Use of Ecopath with Ecosim to evaluate strategies for sustainable exploitation of multi-species resources. *Fisheries Centre Research Report* 6(2).

Christensen, V. and D. Pauly. 1993. Trophic models of aquatic ecosystems. ICLARM Conf. Proc. 26, 390 p.

Christensen, V. and D. Pauly. 1992a. A guide to the ECOPATH II software system (version 2.1). ICLARM Software 6, 72 p.

Christensen, V. and D. Pauly. 1992b. ECOPATH II - a software for balancing steady-state ecosystem models and calculating network characteristics. *Ecol. Modelling* 61: 169-185.

Christensen, V. and D. Pauly. 1993. Flow characteristics of aquatic ecosystems, p. 338-352. *In* V. Christensen and D. Pauly (eds.). Trophic models of aquatic ecosystems. ICLARM Conf. Proc. 26, 390 p.

Christensen, V. and D. Pauly. 1995. Fish production, catches and the carrying capacity of the world oceans. *Naga*, the ICLARM Quarterly 18(3): 34-40.

Chullasorn, S. and P. Martosubroto. 1986. Distribution and important biological features of coastal fish resources in Southheast Asia. *FAO Fish. Tech. Paper* 278. 84 p.

Collier, W.L., H. Hadikoesworo, and S. Saropie. 1977. Income, employment and food systems in Javanese coastal villages. Center for International Studies. Ohio University. Papers in International Studies, Southeast Asia Series No. 44. 152 p.

Csirke, J. 1988. Small shoaling pelagic fish stock. p. 271-302. *In* J.A. Gulland (ed.). Fish population dynamics. Second edition. John Wiley & Sons Ltd., New York

Dall, W., B.J. Hill, P.C. Rothlisberg and D.J. Sharples. 1990. The Biology of the Penaeidae. *Adv. Mar. Biol.* 27. 495 p.

Davenport, S. and J.D. Stevens. 1988. Age and growth of two commercially important sharks (*Carcharhinus tilstoni* and *C. sorrah*) from Northern Australia. *Aust. J. Mar. Freshwater Res.* 39: 417-433

de Wilde, P.A.W.J., W.W. Kastoro, E.M. Berghuis, I. Aswandy, I. Al Hakim and A. Kok. 1989. Structure and energy demand of the benthic soft-bottom communities in the Java Sea and around the islands of Madura and Bali, Indonesia. *Neth. J. Sea Res.* 23(4):449-461.

Deriso, R.B. 1980. Harvesting strategies and parameter estimation for an age-structured model. *Can. J. Fish. Aquat. Sci.* 37: 268-282.

Directorate General of Fisheries (DGF). Fisheries statistical data of the Java Sea, from 1974 to 1994. Digitized from hard-copy documents for the purpose of this study.

Dudley, R.G. and K.C. Harris The fisheries statistical system of Java, Indonesia: operational realities in a developing country. *Aquaculture and Fisheries Management* 18: 365-374

Dudley, R.G. and K.C. Harris. 1984. Field review of the standard system of fishery statistic for Indonesia with recommendation for its improved operation. A report to the Directorate General of Fisheries, Indonesia as part of the USAID's Small Scale Fisheries Development Project. 40 p.

Dwiponggo, A. 1979. Review of the demersal resources and fisheries in the Java Sea. Marine Fisheries Research Institute. Jakarta. 36 p.

Dwiponggo, A. 1987. Indonesia's Marine Fisheries Resources. p. 10 - 63. *In* C. Bailey, A. Dwiponggo and F. Marahudin (eds.). Indonesian Marine Capture Fisheries. ICLARM Stud. Rev. 10.

Dwiponggo, A. 1988. Recovery of overexploited demersal resources and growth of its fishery on the North Coast of Java. *Indonesian Agricultural Research and Development Journal* 10(4): 1-14.

Dwiponggo, A. T. Hariati, S. Banon, M.L. Palomares, and D. Pauly. 1986. Growth, mortality and recruitment of commercially important fishes and penaeid shrimps in Indonesian waters. ICLARM Tech. Rep. 17, 91 p.

Emery, K.D. 1969. Distribution pattern of sediments on the continental shelves of Western Indonesia. ECAFE Tech. Bull. 2: 79-82.

Emery, K.D., E. Uchupi, J. Sunderland, H.L. Uktolseja and E.M. Young. 1972. Geological structure and some water characteristics of the Java Sea and adjacent continental shelf. ECAFE Tech. Bull. 6: 197-223.

Engel, J. and R. Kvitek. 1998. Effects of otter trawling on a benthic community in Monterey Bay National Marine Sanctuary. *Conserv. Biol.* 12(6): 1204-1214.

Finn, J.T. 1976. Measures of ecosystem structure and function derived from analysis of flows. *J. Theor. Biol.* 56: 363-380.

FishBase. 1998. FishBase 98 CD-ROM. ICALRM, Manila.

Froese, R. and D. Pauly. 1998. FishBase 98: concepts, design and data sources. ICLARM, Manila, Phillippines. 293 p.

Froese, R., S.M. Luna and E.C. Capuli. 1996. Checklist of marine fishes of Indonesia, compiled from published literature. p. 217-275. *In* D. Pauly and P. Martosubroto (eds.). Baseline studies of biodiversity: the fish resources of Western Indonesia. ICLARM Stud. Rev. 23.

Gales, R., D. Pemberton, M. Clarke and C.C. Lu. 1992. Stomach contents of long-finned Pilot Whales (*Globicephala melas*) and Bottlenose Dolphins (*Tursiops truncatus*) in Tasmania. *Mar. Mamm. Sci.* 8(4): 405-413.

Greze, V.N. and O. Kinne. 1978. Production of animal population. p. 89-114. *In* O. Kinne (ed.). Marine Ecology: a Comprehensive, Integrated Treatise on Life in Oceans and Coastal Waters. Vol. 4. Dynamics. John Wiley & Sons, New York.

Groombridge, B. 1982. The IUCN Amphibia - Reptilia Red Data Book. Part 1: Testudines, Crocodylia and Rhynchocephalia. International Union for Conservation of Nature and Natural Resources. Gland, Switzerland.

Gunawan, G.A. 1986. Indigeous fishing crafts in Java, Indonesia. p. 409-412. *In* J.L. Maclean, L.B. Dizon and L.V Hosillos (eds.). The First Asian Fisheries Forum. Asian Fisheries Society, Manila, Philippines.

Hall, D.J., W.E. Cooper and E.E. Werner. 1970. An experimental approach to the production dynamics and structure of freshwater animal communities. *Limnology and Oceanography* 15: 839-928.

Hannesson, R. 1993. Bioeconomic analysis of fisheries. John Wiley & Sons Ltd., New York. 138 p.

Harting, P. 1870. Mémoire sur le genre *Potérion.* Societé des Arts et des Sciences d'Utrecht. 40 p.

Holling, C.S. 1973. Resilience and stability of ecological systems. *Ann. Rev. Ecol. Syst.* 4: 1-23.

Hunter, M.D. and P.W. Price. 1992. Playing chutes and ladders: heterogeneity and the relative roles of bottom-up and top-down forces in natural communities. *Ecology* 73(3): 724-732.

International Monetary Fund. 1998. International Financial Statistics Yearbook. Vol. 51. Washington D.C.

International Whaling Commission. 1984. Summary of available live-capture information (Appendix 3). *Rep. Int. Whal. Comm.* 34: 155-157.

Jefferson, T.A., S. Leatherwood, and M.A. Webber. 1993. Marine mammals of the world. *FAO Species Identification Guide.* 320 p.

Jones, J.B. 1992. Environmental impact of trawling on the seabed: a review. *New Zealand J. Mar. Freshwat. Res.* 26: 59-67.

Kastoro, W.W., B. Sudibjo, A. Aziz, I. Aswandy, and I. Al Hakim. 1990. Pengamatan komunitas benthos di Teluk Jakarta. (Observation of benthic communities in Jakarta Bay). p. 50-64. *In* M. Djajasasmita, A. Budiman, Irawati, and H.B. Munaf (eds.). Prosiding Seminar Nasional Biologi Dasar I. Bogor, 14 Pebruari 1989. Pusat Penelitian dan Pengembangan Biologi. Lembaga Ilmu Pengetahuan Indonesia (LIPI), Jakarta. [In Indonesian].

Kawakami, T. 1980. A review of Sperm Whale food. *Sci. Rep. Whales Res. Inst.* 32: 199-218

Kawamura, A. 1977. On the food of Bryde's whales caught in the South Pacific and Indian Oceans. *Sci. Rep. Whales Res. Inst.* 29: 49-58.

Kawamura, A. 1980a. A review of food of balaenopterid whales. *Sci. Rep. Whales Res. Inst.* 32: 155-197

Kawamura, A. 1980b. Food habits of the Bryde's whales taken in the South Pacific and Indian Oceans. *Sci. Rep. Whales Res. Inst.* 32: 1-23

KEPPRES No. 39/1980 tentang penghapusan jaring trawl (Presidential Decree No. 39/1980 pertaining to trawl ban). [In Indonesian].

Kitchell, J.F., E.A. Eby, X. He, D.E. Schindler, and R.M. Wright. 1994. Predator-prey dynamics in an ecosystem context. *J. Fish. Biol.* 45: 209-226.

Klinowska, M. 1991. Dolphins, Porpoises and Whales of the World. The IUCN Red Data Book. Gland, Switzerland. 429 p.

Kompas Online [Jakarta]. 17 June 1998. Pukat Harimau akan dihidupkan lagi [The trawl ban will be revoked]. http://www.kompas.com. [In Indonesian].

Kusuma-Atmadja, M. 1991. Sovereign rights over Indonesian natural reources: an archipelagic concept of rational and sustainable resource management. *Mar. Pol.* 15(6): 383-412.

Larkin, P.A. and W. Gazey. 1982. Applications of ecological simulations models to management of tropical multispecies fisheries. p. 123-139. *In* D. Pauly and G.I. Murphy (eds.). Theory and Management of Tropical Fisheries. ICLARM Conf. Proc. 9.

Lie, U. 1968. A quantitative study of benthic infauna in Puget Sound, Washington U.S.A., in 1963-1964. *FiskDir. Skr. Ser. HavUnders.* 14(5): 229-556.

Lindeman, R.L. 1942. The trophic-dynamic aspect of ecology. *Ecology* 23: 399-418.

Longhurst, A. R., and D. Pauly. 1987. Ecology of Tropical Oceans. Academic Press, San Diego. 407 p.

LON-LIPI. 1977a. Monitoring Teluk Jakarta. (The monitoring of Jakarta Bay). Laporan No. 6, Pelayaran KM Samudera, 26-29 Januari 1977. Kantor Menteri Negara Pengawasan Pembangunan and Lingkungan Hidup. Proyek Penelitian Masalah Pengembangan Sumberdaya Laut dan Pencemaran Laut. Jakarta. 119 p. [In Indonesian].

LON-LIPI. 1977b. Pemonitoran Teluk Jakarta. (The monitoring of Jakarta Bay). Laporan No. 8, Pelayaran KM Samudera, 10-13 Agustus 1977. Panitia Perumus dan Perencanaan Kerja bagi Pemerintah di Bidang Pengelolaan Lingkungan Hidup. Kantor Menteri Negara Pengawasan Pembangunan and Lingkungan Hidup. Proyek Penelitian Masalah Pengembangan Sumberdaya Laut dan Pencemaran Laut. Jakarta. 136 p. [In Indonesian].

LON-LIPI. 1978a. Pemonitoran Teluk Jakarta. (The monitoring of Jakarta Bay). Laporan No. 9. Pelayaran KM Samudera, 26-29 November 1977. Panitia Perumus dan Perencanaan Kerja bagi Pemerintah di Bidang Pengelolaan Lingkungan Hidup. Proyek Penelitian Masalah Pengembangan Sumberdaya Laut dan Pencemaran Laut. Jakarta. 120 p. [In Indonesian].

LON-LIPI. 1978b. Pemonitoran Teluk Jakarta. (The monitoring of Jakarta Bay). Laporan No. 10. Pelayaran KM Samudera, 17-20 Januari 1978. Panitia Perumus dan Perencanaan Kerja bagi Pemerintah di Bidang Pengelolaan Lingkungan Hidup. Proyek Penelitian Masalah Pengembangan Sumberdaya Laut dan Pencemaran Laut. Jakarta. 116 p. [In Indonesian].

LON-LIPI. 1980. Pemonitoran perairan pantai utara Jawa. (The monitoring of the northern coast of Java). Laporan No. 1. Pelayaran KM Samudera, 18-31 Desember 1979. Kantor Menteri Negara Pengawasan Pembangunan and Lingkungan Hidup. Proyek Penelitian Masalah Pengembangan Sumberdaya Laut dan Pencemaran Laut. Jakarta. 159 p. [In Indonesian].

LON-LIPI. 1981a. Pemonitoran pantai utara Jawa (The monitoring of the northern coast of Java). Laporan No. 3. Pelayaran KM Bawal Putih, 29 Januari - 20 Pebruari 1981. Kantor Menteri Negara Pengawasan Pembangunan and Lingkungan Hidup. Proyek Penelitian Masalah Pengembangan Sumberdaya Laut dan Pencemaran Laut. Jakarta. 171 p. [In Indonesian].

LON-LIPI. 1981b. Pemonitoran pantai utara Jawa (The monitoring of the northern coast of Java). Laporan No. 4. Pelayaran KM Bawal Putih, 16 Agustus - 12 September 1981. Kantor Menteri Negara Pengawasan Pembangunan and Lingkungan Hidup. Proyek Penelitian Masalah Pengembangan Sumberdaya Laut dan Pencemaran Laut. Jakarta. 190 p. [In Indonesian].

LON-LIPI. 1981c. Pemonitoran pantai selatan Kalimantan. (The monitoring of the southern coast of Kalimantan). Laporan No. 1. Pelayaran KM Bawal Putih, 29 Januari - 20 Pebruari 1981. Kantor Menteri Negara Pengawasan Pembangunan and Lingkungan Hidup. Proyek Penelitian Masalah Pengembangan Sumberdaya Laut dan Pencemaran Laut. Jakarta. 113 p. [In Indonesian].

LON-LIPI. 1981d. Pemonitoran pantai selatan Kalimantan. (The monitoring of the southern coast of Kalimantan). Laporan No. 2. Pelayaran KM Samudera, 16 Agustus - 12 September 1981. Kantor Menteri Negara Pengawasan Pembangunan and Lingkungan Hidup. Proyek Penelitian Masalah Pengembangan Sumberdaya Laut dan Pencemaran Laut. Jakarta. 116 p. [In Indonesian].

LON-LIPI. 1982a. Pemonitoran perairan estuaria Teluk Banten. (The monitoring of the Banten Bay estuary). Laporan No. 2. Juni 1981 dan Maret 1982. Kantor Menteri Negara Pengawasan Pembangunan and Lingkungan Hidup. Proyek Penelitian Masalah Pengembangan Sumberdaya Laut dan Pencemaran Laut. Jakarta. 119 p. [In Indonesian].

LON-LIPI. 1982b. Pemonitoran perairan pantai selatan Kalimantan. (The monitoring of the southern coast of Kalimantan). Laporan No. 3. Pelayaran KM Samudera, 26 Maret - 9 April 1982. Kantor Menteri Negara Pengawasan Pembangunan and Lingkungan Hidup. Proyek Penelitian Masalah Pengembangan Sumberdaya Laut dan Pencemaran Laut. Jakarta. 106 p. [In Indonesian].

LON-LIPI. 1983a. Pemonitoran perairan pantai selatan Kalimantan. (The monitoring of the southern coast of Kalimantan). Laporan No. 4. Pelayaran KM Madidihang, 28 Mei - 3 Juni 1983. Kantor Menteri Negara Pengawasan Pembangunan and Lingkungan Hidup. Proyek Penelitian Masalah Pengembangan Sumberdaya Laut dan Pencemaran Laut. Jakarta. 116 p. [In Indonesian].

LON-LIPI. 1983b. Pemonitoran perairan pantai selatan Kalimantan. (The monitoring of the southern coast of Kalimantan). Laporan No. 5. Pelayaran KM Samudera, 1-4 Juli 1983. Kantor Menteri Negara Pengawasan Pembangunan and Lingkungan Hidup. Proyek Penelitian Masalah Pengembangan Sumberdaya Laut dan Pencemaran Laut. Jakarta. 121 p. [In Indonesian].

Losse, G.F. and A. Dwiponggo. 1977. Report on the Java Sea southeast monsoon trawl survey, June - December 1976. Special Report/Contribution of the Demersal Fisheries Project No. 3. Jakarta. 119 p.

Malovitskaya, L.M. 1971. Production of species of mass-occurrence copepods in the Gulf of Guinea. *Tr., Atl. Nauchno-Issled. Inst. Rybn. Khoz. Okeanogr.* 37: 401-405.

Malthus, T.R. 1798. An essay on the principle of population. Reprinted 1970 by Penguin Books, Harmondsworth, Middlesex, England.

Margalef, R. 1968. Perspective in ecological theory. University Press, Chicago. 111 p.

Marsh, H., R. Lloze, G.E. Heinsohn, and T. Kasuya. 1989. Irrawaddy Dolphins, *Orcaella brevirostris* (Gray, 1866). p. 101-118. *In* S.H. Ridgway and S.R. Harrison (eds.). Handbook of Marine Mammals, vol. 4. River Dolphins and the Larger Toothed Whales. Academic Press, London.

Martosubroto, P. 1982. Fishery dynamics of the demersal resources of the Java Sea. Ph.D. thesis. Dalhousie University, Halifax, Nova Scotia, Canada. [In microfiche].

Martosubroto, P. 1987. The status of management of the marine fishery resources of Indonesia. p.125-131. *In* Indo-Pacific Fishery Commission (IPFC). Symposium on the exploitation and management of marine fishery resources in Southeast Asia. Darwin, 16-19 February 1987. Regional Office for Asia and the Pacific (RAPA), Food and Agriculture Organization of the United Nations, Bangkok.

Martosubroto, P. 1989. Indonesian Fishery. p. 29-35, 82-86, 98-99; + App. 8 - 32a. *In* Marine fishery production in the Asia-Pacific region. Regional Office for Asia and the Pacific (RAPA), Food and Agriculture Organization of the United Nations, Bangkok. Publication No. 1989/6.

Martosubroto, P. 1996. Structure and dynamics of the demersal resources of the Java Sea, 1975-1979. p. 62-76. *In* D. Pauly and P. Martosubroto (eds.). Baseline studies of biodiversity: the fish resources of Western Indonesia. ICLARM Stud. Rev. 23.

Martosubroto, P. and D. Pauly. 1976. *R/V Mutiara 4* survey data November 1974 to July 1976. Mar. Fish. Res. Rep. (Special Report)/Contribution of the Demersal Fisheries Project No. 2, Jakarta. 135 p.

Matson, P. and M.D. Hunter. 1992. The relative contribution of top-down and bottom-up forces in population and community ecology. *Ecology* 73(3): 723.

McElroy, J.K. 1991. The Java Sea purse seine fishery: a modern-day 'tragedy of the commons'? *Mar. Pol.* July 1991: 255- 271.

Morgan, J.R. and D.W. Fryer. 1985. The marine geography of Southeast Asia. p. 9-32. *In* G. Kent and M.J. Valencia (eds.). Marine Policy in Southeast Asia. University California Press, Berkeley. 425 p.

Nogasawa, S. and R. Marumo. 1972. Feeding of a pelagic chaetognath, *Sagitta nagae* in Suruga Bay, central Japan. *J. Oceanogr. Soc. Japan*, 28(5): 181-186.

National Development Information Office. 1992. Indonesia Source Book 1992. Jakarta.

National Development Information Office. 1994. Indonesia Source Book 1994. Jakarta.

National Development Information Office. 1996. Indonesia Source Book 1996. Jakarta.

Nishiwaki, M., T. Kasuya, N. Miyazaki, T. Tobayama, and T. Kataoka. 1979. Present distribution of the Dugong in the world. *Sci. Rep. Whales Res. Inst.* 31: 133-141.

Nurhakim, S., S. Banon, M. Potier and T. Boely. 1987. Study on the big purse seiners fishery in the Java Sea: II. Evolution and structure of the Javanese purse seiners' fleet. *J. Pen. Per. Laut* 40: 65-76.

Nybakken, J.W. 1982. Marine Biology: an ecological approach. Harper & Row Publishers, New York. 446 p.

Odum, E.P. 1969. The strategy of ecosystem development. *Science* 164:262-270.

Olivieri, R.A., A. Cohen and F.P. Chavez. 1993. An ecosystem model of Monterey Bay, California. p. 315-322. *In* V. Christensen and D. Pauly (eds.). Trophic models of aquatic ecosystems. ICLARM Conf. Proc. 26.

Opitz, S. 1996. Trophic interactions in Caribbean coral reefs. ICLARM Tech Rep. 43, 341 p.

Pauly, D. 1979. Theory and management of tropical multispecies stocks: a review with emphasis on the Southeast Asian demersal fisheries. ICLARM Stud. Rev. 1, 35 p.

Pauly, D. 1980. On the interrelationships between natural mortality, growth parameters, and mean environmental temperature in 175 fish stocks. *J. Cons. Int. Explor. Mer*, 39(2): 175-192.

Pauly, D. 1986. On using other people's data. *Naga*, ICLARM Quarterly 11(1): 6-7.

Pauly, D. 1988. Fisheries research and the demersal fisheries of Southeast Asia. p. 329-348. *In* J.A. Gulland (ed.) Fish population dynamics. Second edition. John Wiley & Sons Ltd., New York.

Pauly, D. 1989. Fisheries resources management in Southeast Asia: why bother? p. 1- 9. *In* T.E. Chua and D. Pauly (eds.). Coastal area management in Southeast Asia: policies, management strategies and case studies. ICLARM Conf. Proc. 19.

Pauly, D. 1994. On the sex of fish and the gender of scientists: a collection of essays in fisheries science. *Fish and Fisheries Series* 14. Chapman and Hall, London. 250 p.

Pauly, D. 1996. Biodiversity and the retrospective analysis of demersal trawl surveys: a programmatic approach. p. 1-6. *In* D. Pauly and P. Martosubroto (eds.). Baseline studies of biodiversity: the fish resources of Western Indonesia. ICLARM Stud. Rev. 23.

Pauly, D. 1997. Small-scale fisheries in the tropics: marginality, marginalisation, and some implications for fisheries management. p. 40-49. *In* E.K. Pikitch, D.D. Huppert, and M.P. Sissenwine (eds.). Global trends: fisheries management. AFS Symposium 20, Bethesda, Maryland.

Pauly, D. and P. Martosubroto. 1996. Baseline studies of biodiversity: the fish resources of Western Indonesia. ICLARM Stud. Rev. 23, 312 p.

Pauly, D. and V. Christensen. 1993. Stratified models of large marine ecosystems: a general approach and an application to the South China Sea. p. 148-174. *In* K. Sherman, L.M. Alexander and B.D. Gold. (eds.). Large Marine Ecosystems: stress, mitigation and sustainability. AAAS Press, Washington, DC.

Pauly, D., M.L. Soriano-Bartz and M.L.D. Palomares. 1993. Improved construction, parameterization and interpretation of steady-state ecosystem models. p. 1-13. *In* V. Christensen and D. Pauly (eds.). Trophic models of aquatic ecosystems. ICLARM Conf. Proc. 26.

Pauly, D., P. Martosubroto and J. Saeger. 1996. The *Mutiara* 4 surveys in the Java and Southern South China Seas, November 1974 to July 1976. p. 47-54. *In* D. Pauly and P. Martosubroto (eds.). Baseline studies of biodiversity: the fish resources of Western Indonesia. ICLARM Stud. Rev. 23.

Pauly, D., V. Christensen and V.J. Sambilay. 1990. Some features of fish food consumption estimates used by ecosystem modellers. ICES CM 1990/G:17. 8 p.

Pauly, D., V. Christensen, J. Daalsgaard, R. Froese, and F. Torres, Jr. 1998. Fishing down marine food webs. *Science* 279: 860-863.

Pauly, D., V. Sambilay, Jr. and S. Opitz. 1993. Estimates of relative food composition by fish and invertebrate populations, required for modelling the Bolinao reef ecosystem, Philippines. p. 236-251. *In* V. Christensen and D. Pauly (eds.). Trophic models of aquatic ecosystems. ICLARM Conf. Proc. 26.

Perrin, W.F. and J.W. Gilpatrick, Jr. 1994. Spinner Dolphin *Stenella longirostris* (Gray, 1828). p. 99-128. *In* S.H. Ridgway and S.R. Harrison (eds.). Handbook of Marine Mammals, vol. 5. The First Book of Dolphins. Academic Press, London.

Perrin, W.F., N. Miyazaki and T. Kasuya. 1989. A dwarf form of the Spinner Dolphin (*Stenella longirostris*) from Thailand. *Mar. Mamm. Sci.* 5(3): 213-227.

Petit, D., F. Gerlotto, N. Luong, and D. Nugroho (eds.). 1995. Java Sea Pelagic Fishery Assessment Project (ALA/INS/87/17). Akustikan I Workshop Report. Jakarta, 5-10 December 1994. Scientific and Technical Document No. 21. 117 p.

Pitcher, T. J., R. Watson, N. Haggan, S. Guénette, R. Kennish, and U.R. Sumaila. 1999. Marine reserves and the restoration of fisheries and marine ecosystems in the South China Sea. Essential Fish Habitat and Marine Reserves, William R. and Lenore Mote International Symposium in Fisheries Ecology, Sarasota, Florida. *Bull. Mar. Sci. (in press).*

Polovina, J.J. 1984a. Model of a coral reef ecosystem. I. The ECOPATH model and its application to French Frigate Shoals. *Coral Reefs* 3: 1-11

Polovina, J.J. 1984b. An overview of the ECOPATH model. *Fishbyte* 2(2): 5-7.

Polovina, J.J. 1993. The first ECOPATH. p. vii-viii. *In* V. Christensen and D. Pauly (eds.). Trophic models of aquatic ecosystems. ICLARM Conf. Proc. 26.

Potier, M. and B. Sadhotomo. 1995. Seiners fisheries in Indonesia. p. 49-66. *In* M. Potier and S. Nurhakim (eds.). Biology, Dynamics, Exploitation of the Small Pelagic Fishes in the Java Sea. Report of the BIODYNEX Seminar, Jakarta, March 1994. PELFISH Project, Jakarta. 281 p.

Power, M.E. 1992. Top-down and bottom-up forces in food webs: do plants have primacy ? *Ecology* 73(3): 733-746.

Priyono, B.E. and B. Sumiono. 1997. The marine fisheries of Indonesia, with emphasis on the coastal demersal stocks of the Sunda shelf. p.38-46. *In* G. Silvestre and D. Pauly (eds.). Status and management of tropical coastal fisheries in Asia. ICLARM Conf. Proc. 53.

Reilly, S.B. and J. Barlow. 1986. Rates of increase in Dolphin population size. *Fishery Bulletin* 84(3): 527-533.

Republik Indonesia. 1968. Rencana Pembangunan Lima Tahun Pertama, 1968/1969 - 1972/1973. [First Five Year Development Plan, 1968/1969 - 1972/1973]. Republik Indonesia, Jakarta. [In Indonesian].

Rice, D.W. 1989. Sperm Whales, *Physeter macrocephalus* Linnaeus, 1758. p. 177-233. *In* S.H. Ridgway and S.R. Harrison (eds.). Handbook of Marine Mammals, vol. 4. River Dolphins and the Larger Toothed Whales. Academic Press, London .

Ross, G.J.B., G.E. Heinsohn and V.G. Cockcroft. 1994. Humpback Dolphins, *Sousa chinenses, S. plumbea* and *S. teuszii.* p. 23-42. *In* S.H. Ridgway and S.R. Harrison (eds.). Handbook of Marine Mammals, vol. 5. The first book of Dolphins. Academic Press, London.

Rutledge, R.W., B.L. Bacore, and R.J. Mulholland. 1976. Ecological stability: an information theory viewpoint. *J. Theor. Biol.* 57: 355-371.

Sainsbury, K.J. 1991. Application of an experimental approach to management of a tropical multispecies fishery with highly uncertain dynamics. *ICES Mar. Sci. Symp.* 193: 301-320.

Sainsbury, K.J., R.A. Campbell, R. Lindholm, and A.W. Whitelaw. 1997. Experimental management of an Australian multispecies fishery: examining the possibility of trawl-induced habitat modification. p.107-112. *In* E.K. Pikitch, D.D. Huppert, and M.P. Sissenwine (eds.). Global trends: fisheries management. AFS Symposium 20, Bethesda, Maryland.

Sardjono, I. 1980. Trawlers banned in Indonesia. ICLARM Newsl. 3(4):3.

Schuster, W. H. and R. Djajadiredja. 1952. Local common names of Indonesian fishes. W.V. Hoeve, Bandung, Indonesia. 276 p.

Shushkina, E.A. and I.I. Kisliakov. 1975. An estimation of the zooplankton production in an equatorial part of the Pacific Ocean in the Peruvian upwelling. *Tr. Inst. Okeanol. im. P. P. Shrishova, Akad. Nauk SSSR.* 102: 384-395.

Silvestre, G., S. Selvanathan and A.H.M. Salleh. 1993. Preliminary trophic model of the coastal fisheries recources of Brunei Darussalam, South China Sea. p. 300-306. *In* V. Christensen and D. Pauly (eds.). Trophic models of aquatic ecosystems. ICLARM Conf. Proc. 26.

Soegiarto, A. 1981. The development of a marine park system in Indonesia. p. 287-291. *In* Proceedings of the Fourth International Coral Reef Symposium, Manila. Vol. 1.

Sparre, P. 1991. Introduction to multispecies virtual population analysis. *ICES Mar. Sci. Symp.* 193: 12-21.

Stewart, D.J., J.F. Kitchell and L.B. Crowder. 1981. Forage fishes and their salmonid predators in Lake Michigan. *Trans. Am. Fish. Soc.* 110: 751-763.

Strømme, T. 1992. NAN-SIS: software for fishery survey data logging and analysis. User's manual. *FAO Computerized Information Series (Fisheries)* 4. 103 p.

Su, M.S. and I.C. Liao. 1981. Preliminary studies on the distribution and the stomach contents of some commercial prawns from the coast of Tungkang, Taiwan. p. 57-71. *In* I.C. Liao and R. Hirano (eds.). Proceedings of ROC-Japan Symposium on Mariculture. TML Conference 1. Tungkang Marine Laboratory, Tungkang, Pingtung, Taiwan.

Sujastani, T. 1981. The state of Indonesian marine fishing resource exploitation. *Indones. Agric. Res. Dev. J.* 3(1): 13-20.

Sumaila, U.R. 1998. Ecosystem/economic impact analysis. p. 83-89. *In* R. Bonfil, G. Munro, U.R. Sumaila, H. Valtysson, M. Wright, T. Pitcher, D. Preikshot, N. Haggan, and D. Pauly (eds.). Distant Water Fleets: an Ecological, Economic and Social Assessment. *Fisheries Centre Research Reports* 6(6).

Taryono. 1998. Pencabutan pelarangan 'trawl': mungkinkah? (Is it possible to revoke the trawl ban?). Suara Pembaruan Daily, July 28, 1998. http://www.suarapembaruan.com. [In Indonesian].

Tas'an and S. Leatherwood. 1984. Cetaceans live-captured for Jaya Ancol Oceanarium, Djakarta, 1974-1982. *Rep. Int. Whal. Comm.* 34: 485-489.

Tiews, K. 1973. Fishery development and management in Thailand. *J. Fish. Res. Board Can.* 30: 2333-2334

Torres, F., Jr., A. Cabanban, S. Bienvenida, J. McManus, M. Prein and D. Pauly. 1996. Using the NAN-SIS and FiSAT Software to create a trawl survey database for Western Indonesia. p. 276-283. *In* D. Pauly and P. Martosubroto (eds.) Baseline studies of biodiversity: the fish resources of Western Indonesia. ICLARM Stud. Rev. 23.

Trites, A. and D. Pauly. 1998. Estimates of mean body weights for marine mammals from measurements of maximum body lengths. *Can. J. Zool.* 76(5): 886-896.

Trites, A. and K. Heise. 1996. Marine Mammals. p. 25-30. *In* D. Pauly and V. Christensen (eds.). Mass-balance models of Northeastern Pacific ecosystem. *Fisheries Centre Research Reports* 4(1): 131 p.

Trites, A., V. Christensen and D. Pauly. 1997. Competition between fisheries and marine mammals for prey and primary production in the Pacific Oceans. *J. Northw. Atl. Fish. Sci.* 22: 173-187.

Ulanowicz, R.E. 1986. Growth and Development: Ecosystem Phenomonology. Springer-Verlag. New York. 203 p.

Ulanowicz, R.E. and J.S. Norden. 1990. Symmetrical overhead in flow and networks. *Int. J. Systems Sci.* 21(2): 429-437.

Unar, M. 1972. Review of the Indonesian shrimp fishery and its present development. Research Report on Marine Fisheries No. 1, Marine Fisheries Research Institute, Jakarta.

Van Soest, R.M. 1989. The Indonesian sponge fauna: a status report. *Neth. J. Sea Res.* 23(2): 223-230.

Walters, C.J. 1996. Suggested improvements for Ecopath modelling. p. 82-86. *In* D. Pauly and V. Christensen (eds.). Mass-balance models of North-Eastern Pacific ecosystems. *Fisheries Centre Research Report* 4(1).

Walters, C.J., D. Pauly, and V. Christensen. 1998. Ecospace: prediction of mesoscale spatial patterns in trophic relationships of exploited ecosystems, with emphasis on the impacts of marine protected areas. ICES CM 1998/S:4. 20 p.

Walters, C.J., V. Christensen and D. Pauly. 1997. Structuring dynamic models of exploited ecosystems from trophic mass-balance assessments. *Rev. Fish Biol. Fish.* 7: 139-172.

Wassenberg, T.J. and B.J. Hill. 1987. Natural diet of the Tiger Prawns *Penaeus esculentus* and *P. semisulcatus*. *Aust. J. Mar. Freshwat. Res.* 38: 169-182.

Watling, L. and E.A. Norse. 1998. Disturbance of the seabed by mobile fishing gear: a comparison to forest clearcutting. *Conserv. Biol.* 12(6): 1180-1197.

White, A. 1983. Valuable and vulnerable resources. p. 26-39. *In* J.R. Morgan and M.J. Valencia (eds.). Atlas for marine policy in Southeast Asia. University of California Press, Berkeley. 144 p.

White, A. 1984. Vulnerable marine resources, coastal reserves, and pollution: a Southeast Asian perspective. p. 170-174. *In* J.A. McNeely and K.R. Miller (eds.). National Parks, Conservation, and Development: the role of protected areas in sustaining society. Smithsonian Institution Press, Washington, DC.

Widodo, J. 1976. A check-list of fishes collected by *Mutiara 4* from November 1974 to November 1975. Lap. Pen. Per. Laut. Special Report/Contribution of the Demersal Fisheries Project No. 1: 47-77.

Winberg, G.G. 1956. Rate of metabolism and food requirements of fishes. *Transl. Fish. Res. Board Can.* 253.

Wood, E.M. 1987. New Studies in Biology: Subtidal Ecology. Edward Arnold (Publishers) Ltd., London.

Wyrtki, K. 1961. Physical oceanography of the Southeast Asian waters. Scripps Ints. Oceanogr., Naga Rep. 2, 195 p.

Yamamoto, T. 1978. Basic economic structure of marine fishery in Sumatra, Java and Bali. Simposium Modernisasi Perikanan Rakyat, Jakarta, 27-30 June 1978. Lembaga Penelitian Perikanan Laut, Jakarta.

Yamamoto, T. 1980. A standard statistical system for current fishery statistics in Indonesia. A report prepared for the fisheries development and management project, Indonesia. Rome, FAO. FI:DP/INS/72/064, Field Document 7, 79 p.

Zachman, N. 1973. Fisheries development and management in Indonesia. *J. Fish. Res. Board Can.* 30 (12/2):2335-2340.

APPENDICES

Appendix 1. Presidential Decree No. 39/1980 (English translation)

Logo of the Presidential Seal

Decree of the President of the Republic of Indonesia
No. 39, Year 1980

Pertaining to the Trawl Ban

The President of the Republic of Indonesia

Considers: the fact that in managing the sustainability of the demersal resources, and in supporting the production increase for artisanal fishers, and in avoiding social tensions; it is deemed necessary to ban any fishing activity that uses trawls;

Refers to:

1. Chapter 4, article (1); and chapter 33, article (3) of the 1945 Constitution;
2. The Decree of the People Representative Council of the Republic of Indonesia No. IV/MPR/1978;
3. Coastal Fisheries Ordinance No. 144/1927;
4. Act No. 4/1960 pertaining to Indonesian Waters (State Gazette No. 22/1960; Additional State Gazette No. 1942)
5. Government Regulation No.64/1957
6. Presidential Decree No. 7/1979 pertaining to REPELITA III, 1979/1980 - 1983/1984

DECIDES:

Stipulates: The Decree of the President of the Republic of Indonesia pertaining to the Trawl Ban

Chapter 1

(1) To phase out all fishing activities that use trawls
(2) The terminology of "trawl" in this decree is to include any gear that resembles or is considered as trawl. Details on this aspect will be stipulated further.

Chapter 2

As of July 1, 1980 and up to July 1, 1981, all vessels that use trawls will be totally phased out, so that the remaining trawlers left will be 1,000 units.

Chapter 3

The phasing out as stipulated in Chapter 2, will be conducted as follows:

A. First phase:
   a. As of July 1, 1980, and up to September 30, 1980, phasing out will be carried out to trawlers that operate and domicile in areas surrounding Java and Bali islands;
   b. On October 1, 1980, all trawling activities are forbidden in areas surrounding Java and Bali islands.
   c. For trawlers that operate and domicile in areas surrounding Sumatra island, the ban will take into effect on January 1, 1981.

B. Second phase:
   From October 1, 1980 to July 1, 1981, in areas other than those stated in the first phase, all trawlers will be phased out up to 1,000 units.

Chapter 4

The phasing out of trawlers as stipulated in Chapter 3 B above, and other considerations with respect to the 1,000 units of trawl, will be further regulated.

Chapter 5

(1) All ex-trawler vessels can still be used to fish, as long as they use gears other than trawls.
(2) Owners of ex-trawler vessels as stated in article (1) above who do not wish to continue their business in fisheries, can sell their vessels to other parties or to the government, to be used as non-trawl vessels.
(3) For those who sell their trawl vessels to the government, appropriate compensation will be given.
(4) The ex-trawler vessels that were sold to the government, will be distributed to fishers cooperatives, to be utilized as non-trawl vessels.
(5) The ex-trawler vessels as stated in article (4) above will be distributed to the fishers cooperatives through credit program. Two other credits will also be added to the package, namely a credit for gear conversion, and a credit for capital.

Chapter 6

(1) The conduct of this decree at regional level, will refer to further regulations that will be stipulated by the Minister of Agriculture.
(2) The Minister of Agriculture will further regulate:
   a. Details on the terminology of trawls;
   b. The phasing out process of trawlers;
   c. License renewal for trawlers that have not been phased out.

(3) The Minister of Agriculture with other related ministers, will further regulate the government's buyback process of ex-trawler vessels, price taxation, and the process of distributing the ex-trawler vessels to fishers cooperatives.

Chapter 7

(1) To reduce the decrease of prawn production due to the trawl ban, a national prawn program will be encouraged.
(2) The Minister of Agriculture with other related ministers, will further regulate the national prawn program; in order to support the trawlers phasing out process.

Chapter 8

Fishing vessels that violate this decree and other related decrees and regulations that will be stipulated in the future, will be considered as conducting illegal fishing; and can be sued in court, as regulated in Chapter 15 of the Coastal Fisheries Ordinance No. 144/1927.

Chapter 9

This Presidential Decree takes into effect on the date of the stipulation.

Stipulated in Jakarta
on July 1, 1980

The President of the Republic of Indonesia

Signed

SOEHARTO

Appendix 2 Model validation of the Ecopath and Ecosim models, using comparison of predicted yield values (estimated by Ecosim) and observed yield values (from DGF fisheries database of the Java Sea, 1975 to 1994) for the five highest yields. Coefficient correlations were also estimated. Results indicated that the first two groups with the highest yields (i.e., clupeoids and miscellaneous pelagics) showed very close trends between the predicted and the observed values.