ECOTROPHIC MODEL FOR AN ECOSYSTEM APPROACH FOR MANGROVE FISHERIES IN THAILAND

Sirapraphа Premcharoen

Thesis submitted for the Degree of

Doctor of Philosophy

in

Environmental Science

Environment Department

University of York

June 2009

Abstract

Estuaries, particularly coastal mangrove forests, have been focal points of human settlement and marine resource use throughout history, and they fulfill important socio-economic and environmental functions. Mangroves are currently threatened by various forms of exploitation and coastal development of which conversion to coastal aquaculture is one of the most serious in Thailand.

This study is focused on the Mae Klong Estuary, one of the four main mangrove estuaries in the Gulf of Thailand. The estuarine ecosystem of the Mae Klong is described through a mass-balance model (Ecopath) that includes 21 functional groups (state variables), representing 63 exploited fish and commercial invertebrate species as well as the energy (feeding) fluxes among them which pit artisanal fishers, using push net. The parameterization of the model is described in some detail, as are the implications of the ecological and multispecies interactions. The results emphasize the need of management and conservation between the two sectors of the fisheries and forestry, whose present trajectories tend toward further degradation of the Mae Klong ecosystem.

The Mae Klong Estuary supports a rich fish fauna in terms of number of species, abundance and biomass. A total of 63 fish species representing 25 families were recorded, with Clupeidae by far the most speciose (9 species). *Arius macronotacanthus* dominated the biomass of all fish and the biomass of all fish was maximum during the rainy season.

Three season-specific Ecopath models were developed and used to compare the biomass, production, consumption, biomass flows and higher order indices of ecosystem functioning of the Mae Klong Estuary in dry, hot and rainy seasons. Several higher order indices related to the ecosystem maturity indicators were computed for the Mae Klong models and were compared with other coastal ecosystems around the world. The results indicate that Mae Klong Estuary has a mixture of characteristics of a mature system (high total system throughput, ascendency and overhead) as well as an immature system (high PP/B and PP/R, low Finn's cycling index and mean path length), something which has been encountered for several estuarine systems.

The effects of harvesting "experiments" on shrimp groups on the biomass of other target fish species within the system using Ecosim revealed likely changes in some groups (mullet and croaker) which would have been difficult to predict from simple assumptions about species interactions and shows the power of multi-species models for fisheries planning.

Table of contents

List of tables

List of figures

Acknowledgements

This section of my thesis should probably be the most read, as there are so many people who have helped me on the road to my dissertation who require special mention. First and foremost, I would like to express my sincerest gratitude to my supervisor, Professor Dave Raffaelli, for his patience, encouragement, many insightful suggestions and keen supervision during the entire period of the study. He repeatedly went above and beyond the call of a supervisor's duties. I also express my cordial esteem and sincere appreciation to my TAC supervisors: Dr. Piran White and Dr. Rob Marchant and to my external examiner, Professor Mike Elliott for their valuable advice and instruction.

A study of this scale would be impossible without the assistance of my students for their enthusiastic help in the field and in the laboratory, working long days and long weeks to conduct the studies presented here. Thanks to Siriwan Suksri, Chakree Thes-arsen, Prapaporn Vitheesawat and Jumlong To-on.

I greatly appreciate the help from many people for their assistance with teaching me in computer skills and providing some software programs: Pitiwut Teerakittikul, Pravech Ajawatanawong, Anurak Sriariyawat, Phisut Apichayakul and Pete Rotpai.

Special thanks to my friends and fellow graduate students at the Environment Department: Leanne Mason and Ruth Thurstan for providing many fond memories, and their constant encouragement and willingness to lend an ear when I was in need. Thanks to Dr Julie Hawkins, Professor Callum Roberts, Drs Bryce and Jo Beukers-Stewart, Dr. Shezlin Hason, Philip James, Samina Khali, Debbie Coldwell and Jo Hardstaff for their friendly greetings and interest in my progress. To Dave Hay for providing technical support, Kathryn Addison, Maroula Hill and Clarie Hughes for their assistance with administrative matters, suggestions on university-wide policies and their perpetual smiles and willingness to help me. I also would like to thank the staff of Environment Department for their hospitality which made me feel at home.

Thanks are also to my Thai friends in York and around UK for their constant encouragement and support: Apichat Thammultri, Taweesak Kritjaroen, Sittipol Viboonthanakul, Sasitsaya Saengphueng, Chutima Tan, Weenat Ganjanarode, Paramita Punwong, Thongrob Ruenbanthoeng, Somphop Rodamporn and Kulapa Kuldilok.

I am grateful to Zoology Section, Faculty of Liberal Arts and Science, Kasetsart University at Kamphaeng Saen, who allowed me to use lab facilities to analyse thousands of fish samples, at any time and any day, holiday or weekend.

I wish to express my gratitude to Royal Thai Government for providing financial support in the form of a 4-year scholarship, including university fees and living expenses. I am also grateful to the Environment Department, University of York for financial support in the form of a grant covering field work expenses.

And last, but not definitely not least, my deepest gratitude is to Mom, Dad, Pa Lek, Na Daeng, Na Jaew and dearest sister, Jeab, who have been extremely supportive, patient and understandable right the way through and for instilling in me the belief that I can accomplish anything I set my mind to. This thesis I dedicate to you, my future success will pay back your sacrifice.

In the sunny, beautiful summer in York
Sirapapha Premcharoen
June 2009

"So long as water moves, so long as fins press against it, as long as weather changes and man is fallible, fish will remain in some measure unpredictable"
Roderick Haig-Brown

Declaration

I declare that all the work contained within this thesis is my own unless otherwise stated.

CHAPTER 1

General introduction

1.1 Introduction

Estuaries, which include mangroves, mudflats, and the lower reaches and mouths of the rivers, are dynamic systems in which environmental fluctuations and changing species compositions are common (Wilson and Sheaves 2001; Vidthayanon and Premcharoen 2002). Estuaries have traditionally been the focus of human settlement and activity by virtual of their highly productive fisheries and shell fisheries, trade routes and ports and their vital link to the terrestrial hinterland (Valiela et al. 2001). Because of this they experience large scale impacts of land claim, habitat destruction, over-exploitation, pollution and eutrophication (Micheli 1999; McIsaac et al. 2001). Despite these disturbances, estuaries are very productive environments in general (Rybarczyk et al. 2003), and are used by fish and invertebrate species for reproduction, feeding, and sheltering from predators (Barry et al. 1996; Layman and Silliman 2002; Francis et al. 2005). The food webs, and the pathways of energy flow within webs, are temporally variable in estuaries due to changes in river flow, water temperature, water column stratification, salinity gradients, large-scale seasonal changes in biota, for example due to migration of birds and fish and ontogenetic changes in feeding strategies of many species. Many processes and patterns are common to all estuaries, but some are determined by local conditions, which therefore makes every estuary unique and special (Raffaelli 1992). In ecosystems dominated by this variation, the resilience of the food web may depend largely on how energy flows through the system (Hunter and Price 1992), with many estuarine food webs appearing to be highly resilient, as they remain generally intact despite the challenges of an extremely dynamic and disturbed environment (Day et al. 1989).

There has been considerable research carried out on European and North American estuaries, but some of the larger tropical estuaries are less well understood (Kennish 2002). Tropical estuaries differ markedly from temperate estuaries in the presence of dense and complex forests (mangroves) that characterise the shore-line, forming forests of salt-tolerant species, with complex food web and ecosystem dynamics (Valiela et al. 2001). Mangroves provide an environmental function (Table1.1), a number of ecosystem

Table 1.1 Environmental function of mangroves (Macintosh and Ashton 2002)

Regulation functions	Protection against harmful cosmic influences Local and global energy balance Chemical composition of the atmosphere Chemical composition of the oceans Local and global climate Run-off flood prevention Water catchment and groundwater recharge Prevention of soil erosion, sediment control Topsoil formation, maintenance of fertility Solar energy fixation, biomass production Storage /recycling of nutrients Storage /recycling of water Biological control mechanisms Migration and nursery habitats Biological (and genetic) diversity
Production functions	Oxygen Water (drinking, irrigation) Food Genetic resources Raw materials for construction fuel and energy Biochemical fodder and fertilizer Ornamental resource
Carrier functions (providing space and a suitable substrate)	Human habitation (indigenous settlements) Cultivation (fish, carps, cattle) Recreation and tourism Nature protection
Information functions	Aesthetic information Spiritual and religious information Historic information (heritage value) Cultural and artistic inspiration Scientific and educational information

goods and services, including supporting, regulating, provisioning and cultural services (Aksornkoae et al. 1985; Twiley 1997; Barnes et al. 1998; Macintosh and Ashton 2002). Many of these have changed in status over the past years (Ong 1995; Macintosh et al. 2002), due to trade-offs in land use which may be economically beneficial in the short term, but reduce the value of other services, such as protection from floods and storms, including Asia tsunamis in eleven countries around the Indian Ocean (Dahdouh-Guebas et al. 2005; Stobutzki and Hall 2005), cyclone Orissa in India in 1999 and cyclone Sidr in Bangledesh in 2007 (Porteus 2008), including cyclone Nargis in Burma in 2008 (Thomalla et al. 2008).

This thesis focuses on one such tropical estuarine system, the inner Gulf of Thailand, a major mangrove estuarine area in Southeast Asia, which is one of the most productive areas compared to others within the wider Gulf (Menasveta 1976). However a little information available on the food webs and trophic organization of the ecosystem (Chong et al. 1990; Poovachiranon and Satapoomin 1994; Sasekumar et al.1994; Hajisamae et al. 1999). This area is covered with mangrove forests and the shoreline is muddy alluvium. The effluents from the four main rivers, Tha Chin, Mae Klong, Chao Phraya and Bang Pakong, transport a large amount of silt annually to create deltas. Although this region covers a vast area of mudflats, the mangrove forest has declined due to human impacts. There are many reasons for the destruction of mangrove forests, including increasing population pressure, coastal development, mining, conversion to salt pounds and agriculture overharvesting of the forests for timber and fuel, but the largest factor in recent years has been the widespread expansion of aquaculture ponds into mangrove forests (Aksornkoae 1985). Several fishing and rural communities depend on the fish and shellfish in mangroves as a source of income and food security; when mangrove forests are destroyed, a significant decrease in local fish catches may result. Thailand has major offshore fisheries, which represent a significant portion of national income and depends partly on mangroves (FAO 2007). Thailand lost more than half of its mangrove forest area between 1961 (372,000 ha) and 1993 (168,000 ha) (Thailand Environment 2000), so that only 0.45% of mangrove forests remains in the inner part of the Gulf (Sudara et al. 1994). The conversion of mangroves to shrimp farming has been particularly evident in Thailand over the past 25 years (Huitric et al. 2002; Barbier 2003).

Exploring the trophodynamics of mangrove ecosystems provides insights into their resilience or "ecosystem health" as well as into fish assemblages which remain an important provisioning service for mangrove ecosystems (Vega-Cendejas and Arreguín-Sáchez 2001, Vega-Cendejas 2003). Several studies of mangroves associated with fish and fisheries in Thailand have clearly demonstrated that mangroves maintain estuarine water quality and play crucial roles in the life cycle of many species of fish (Vathanachai, 1979; Monkolprasit, 1994; Boonruang and Satapoomin, 1997; Janekitkarn et al. 1999; Vidthayanon and Premcharoen 2001, 2002; Ikejima et al. 2003). Mangrove areas are utilized and exploited in many ways that result in several resources and environmental problems, such as fertility decline, salt intrusion, reduced forest products and environmental degradation, many of which influence, either directly or indirectly, fish and fisheries of estuarine and diadromous species (Snidvongs 1982). At the same time, fishing activities have been proposed as the most superficial human disturbance to the Gulf of Thailand ecosystem (Vibunpant et al. 2003).

Mangrove forests are important for marine coastal food webs because they provide food (via detritus) to both estuarine and ocean consumers, serve as habitat for early life history stages, juveniles and adults of estuarine and many marine species, and they play an important role in the regulation of estuarine biogeochemical cycles (Vega-Cendejas and Arreguin-Sanchez 2001). To understand mangrove food webs better, a study of multispecies interactions is needed which includes trophic fluxes and efficiencies of energy assimilation as well as energy transfer and dissipation (Vega-Cendejas 2003). These are reflected by the diversity, abundance, distribution and persistence of the biological components which are ultimately regulated by primary productivity (Oksanen et al. 1981; Oksanen 1983), environmental variability (Pimm and Kitching 1987) and a combination of both (Persson et al. 1992). Due to their temporal and biological complexity, it is difficult to understand the structure of food web and trophic interactions by direct observation (Schoenly and Cohen 1991; Niquil et al.1999) and ecosystem–level experiments are difficult to replicate (Carpenter 1990), so have a modelling approach is adopted.

Ecosystem modelling is an alternative to experimental approaches that can be used to predict ecosystem responses to perturbations and to identify higher-level properties of the ecosystem that are not readily estimated empirically (Straile 2002). Ecosystem models which are well-parameterised with field data allow realistic baseline conditions to be constructed so that future model predictions can be compared. This approach helps resource managers and scientists in determining the effects of anthropogenic impacts on ecosystems. Using model simulations to characterize the structure and function of an ecosystem, as well as identifying vulnerable and critical species, allows monitoring goals to be formulated and management becomes more efficient. Ecosystem models can also be used to explore the economic benefits of estuaries, which is often needed to evaluate benefits versus costs of various management alternatives. The importance of models for ecological forecasting in the development of regulatory policy is well recognized (Clark et al. 2001).

Many different types of modelling approaches are available, and here I will focus on food web and ecosystem models that allow the modelling of perturbations over several different time scales. Ecopath with Ecosim software (EwE) (Pauly et al. 2000; Christensen and Walters 2004), is a mass-balance modelling approach that has been widely used to quantitatively describe aquatic systems and to assess the impacts that fishing activities and environmental factors have on marine ecosystems (Christensen and Pauly 1993; Pauly et al. 2000; Christensen and Walters 2004). Ecopath is a steady-state model that estimates energy or biomass flows among food web functional groups. Ecosim, which uses Ecopath files, can be used to explore the consequences of changes in some functional groupings (Walters et al. 1997). Ecopath also allows for identification of the key components of the ecosystem, as well as estimation of higher-order indices such as capacity, throughput, and ascendancy, thought to relate ecosystem resilience (see chapter 4). The Ecopath model was originally derived from an approach first developed by Polovina (1984) to estimate biomass and food consumption of the different elements of an ecosystem. It has subsequently been combined with various approaches from theoretical ecology (Ulannowicz 1980, 1986) for analysis of flows between ecosystem components (Christensen and Pauly 1992a, b). Ecopath models have been applied to many different systems throughout the world (Christensen and Pauly 1993; Christensen 1995); including mangroves in South America (Wolff et al. 2000; Vega-Cendejas and Arreguín-Sáchez 2001; Rivera-Arriaga 2003; Vega-Cendejas 2003; Vidal and Basurto 2003; Velasco and Castello 2005; Avila Foucat 2006), West Africa (Longonje 2008) and Northwest Africa (Amorim et al. 2004), South Asia (Mustafa 2003; Mohamed et al. 2005) and Sotheast Asia (Bundy and Pauly 2001; Garces et al. 2003; Nurhakim 2003).

There have been a few studies in the wider Gulf of Thailand using Ecopath and Ecopath with Ecosim during the 1990s (see Chapter 4 for detail). However, there have been no models describing energy fluxes in a mangrove ecosystem in the inner Gulf of Thailand, to date.

1.2 Aim and research questions

The aim of the present study was to investigate fish assemblages in the area in order to construct mass balance models (Ecopath) for evaluating the ecosystem health of the Mae Klong and for exploring fisheries scenarios. This aim was to answer the following specific questions:

- Are there seasonal and spatial differences in fish assemblages in the area?
- How does the mangrove estuary ecosystem function in terms of trophic interactions and the amount of energy transferred?
- How does the structure and function of the food web vary among the seasons?
- Can the mass balance model approach be used to understand the ecosystem effects of the decline in biomass of fisheries resources in the area?
- Does the mass balance model approach has the potential for evaluating ecosystem health of mangrove estuary?

To achieve these aims of study, it was decided to construct the Mae Klong Mangrove Estuary trophic model for the inner Gulf of Thailand in order to quantify its structure and function, to determine its flow of energy and the role of the fish community in transferring energy from the mangrove areas to adjacent ecosystems, to describe the ecosystem impact of fishing and mangrove deforestation, and to analyse how the structure and function of the food web varies among the seasons during the period of study. The results of this study will help in evaluating the effectiveness of mass balance models in describing ecosystems in general and specifically assist mangrove estuary management in Thailand, including strengthening our understanding of both forestry and fisheries. Moreover, the outputs of such models could also be used as tools for diagnosing ecosystem health. Based on the results of the diagnoses, effective policy decisions regarding management of ecosystems will become possible.

1.3 Outline of chapters in thesis

The thesis is structured as follows: In Chapter 2, I present a broad description of the study area, the Mae Klong Estuary and the Inner Gulf of Thailand, including hydrological, physical and biological characteristics. I also review the coastal fisheries situation and fishery resources in Thailand and describe the status of mangroves in Thailand. In Chapter 3, I investigate the fish assemblages in this system in order to establish the ecological groups within the area. Biomasses and diets from stomach contents of fish are also examined and the information derived in this chapter is used in Chapter 4. In Chapter 4, I construct a mass-balance model using Ecopath with Ecosim version 5 and 6 (EwE , www.ecopath.org) based primarily on the ecological groups results come Chapter 3, biomass and stomach content analysis of fish from the present study, as well as using data from the surveys conducted in the Gulf of Thailand during 1973 and 1993 (Viboonpun et al. 2003) together with FishBase (www.fishbase.org) and the literature reports for species groups in the similar area (see Table 4.3). Three season-specifically, steady-state Ecopath models are constructed and compared for production, consumption and biomass flows. I also calculate higher-order indices of ecosystem functioning of the Mae Klong Estuary in each of the seasons to evaluate ecosystem health. In Chapter 5, I apply the Ecosim model to explore how do the shrimp and sergestid shrimps harvests affect key commercial fish species in the Mae Klong Estaury and what is the likely mechanism of this effect. In the final chapter (6), I present a synthesis of the main findings, and critically comment on the various ways in which the results can be interpreted, as well as making an assessment of the potential and limitations of the mass-balance approach for ecosystem management.

CHAPTER 2

Mangroves and fisheries in Thailand and the Mae Klong Estuary

2.1 The ecosystems of mangrove estuaries in Thailand

2.1.1 Habitat characteristics

The word mangrove is a functional classification not a taxonomic one. Mangroves have been defined as woody plants with a canopy cover of greater than 50%, covering an area of approximately 190,000 to 240,000 km^2 of sheltered coastlines in in the tropics and subtropics between latitudes 25°N and 28°S in 117 countries, occupying about one-quarter of the world's coastal line (Lugo et al. 1990; Upadhyay et al. 2002). Mangroves usually grow in the upper part of the intertidal zone between mean sea level and mean high water spring tide (Chapman and Underwood 1995), and they comprise ~ 70 species in ~ 27 genera from 20 quite different angiosperm families worldwide (Tomlinson 1986; Duke 1995). Mangrove forests can be divided into two groups: Old World and New World (Mitsch and Grosselink 2000). The greatest number of mangrove species (~60 species) are in the Old World, concentrated in Asian countries such as Indonesia, Malaysia, Vietnam, and Thailand and in the Indo-West Pacific region, which includes Australia and East Africa. Only a small number of mangrove species (~10 species) are found in the New World, which includes the north and south coasts of America and the west coast of Africa (Taal 1994: Raffaelli and Hawkins 1996). Only two families, Pellicieraceae and Avicenniaceae, are comprised exclusively of mangroves. In the family Rhizophoraceae, for example, only four of its sixteen genera live in mangrove ecosystems (Duke 1992). *Avicennia, Rhizophora* and *Bruguiera* are the most widespread genera, and these have extensive modifications of their root systems (Raffaelli and Hawkins 1996). In Thailand, mangrove forests have been named "Pa Kongkang" after the major species (*Rhizophora* spp.), and the local name is "Pa Chai Len" (Tomlinson 1986). They occur on the muddy tidal flats at seashores, around lagoons, and river mouths along the coast of southern and eastern Thailand, covering large areas along the western and the eastern Peninsular coast, in the Chao Phraya delta and along the south-eastern coast (Figure 2.1) (Aksornkoae et al.1985; Giessen et al. 2007).

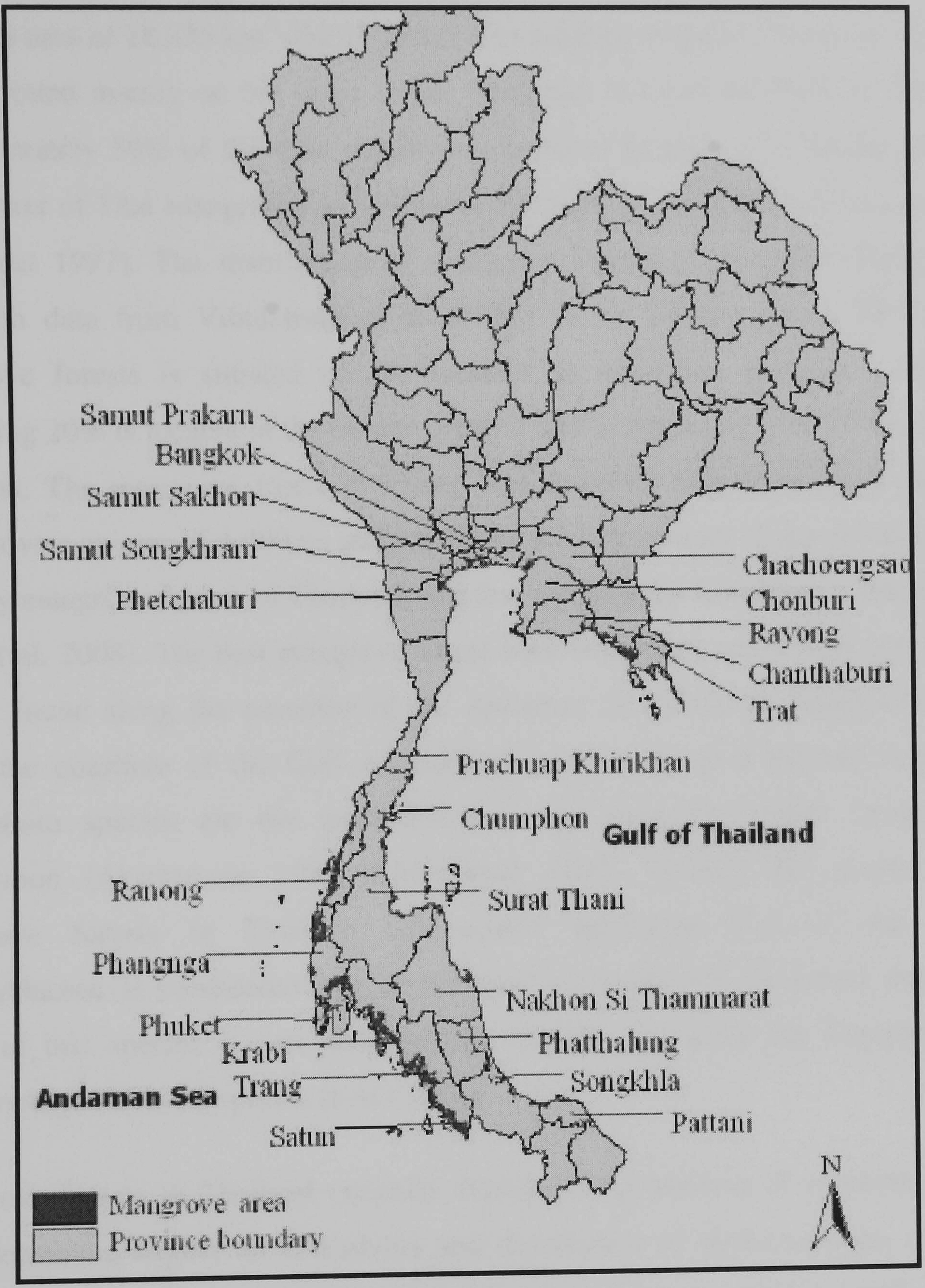


Figure 2.1 Distribution of mangrove forests in Thailand (Dulyapurak et al. 2007).

Thailand, a tropical country lying in the center of mainland Southeast Asia, has marine fisheries operated in two major fisheries area of 23 coastal provinces: the Gulf of Thailand (17 provinces) with a coastline of approximately 2,700 km (1,143 miles) and the Andaman Sea (6 provinces) 865 km (537 miles), giving a total shoreline area of 18,235 km^2 (OEPP 1998). The existing mangrove forest in Thailand can be found mainly on the coast of the Andaman Sea and the Gulf of Thailand. Approximately 50% of the total coastline is covered by mangrove forests, and the total extent of Thai mangroves in 1996 was estimated at 167,582 ha (Charuppat and Charuppat 1997). The distribution of mangrove forests in Thailand (Figure 2.1), based on data from Vibulsresth et al. (1975), is as follows: about 80% of the mangrove forests is situated on the western or Andaman coastline, while the remaining 20% is located in the eastern, central and southeastern areas of the Gulf of Thailand. The mangrove forest in Phang Nga Province (Ao Phang Nga National Park) covers an area of 4,000 ha and represents the largest tract of remaining original primary mangrove forests of Thailand, and that in Prachuap Khirikhan is the smallest (Giri et al. 2008). The best mangrove forest with large trees and a high tree density can be found along the coastline of the Andaman Sea while the mangrove forest along the coastline of the Gulf of Thailand exists only as a narrow strip, with *Rhizophora* species are the most abundant and have the widest geographical distribution (Aksornkoae 1985; Aksornkoae 2004). Among the plants of the mangrove forests in Thailand, *Excoecaria aqallocha* (L.) of the family Euphorbiaceae is considered one of the most economically important trees. The wood of this species is soft and foresters recently requested the Department of Forestry to protect this species (FAO 1980).

Mangrove forests in Thailand typically exhibit strong patterns of zonation (Figure 2.2), depending largely on availability and distribution of seeds/seedings, tolerance of species for inundation, differences in the rooting as well as soil salinity (Aksornkoae et al 1985; Amarasinghe et al. 2009), but generally the forest is two-storeyed, water-front zone and mixed species zone, with an upper layer to 20 m high (FAO 1980). FAO (1980) and Giesen et al. (2007) give a good description which is repeated have verbatim “The pioneer mangrove tree growing in the upper storey is *Rhizophora apiculata* and mixed to a lesser extent with species such as *Rhizophora* *mucronata* (both are locally named kongkang), ngon kai (*Heritiera littoralis*) and *Xylocarpus moluccensis*. Common species of the lower layer are thua khao (*Bruguiera cylindrica*), thua dam (*Brugaria parviflora*), prasak nu (*Bruguiera sexangula*) and prong (*Ceriops decandra* and *Ceriops tagal*). Prasak (*Bruguiera gymnorrhiza*) is a common emergent up to 40 m in height and 2 m in girth.

Other species are ta bun khao (*Xylocarpus obovata*), ta bun dam (*Xylocarpus moluccensis* syn *Carapa moluccensis*), samae (*Avicennia officinalis* and *Avicennia marina*), lam phu (*Sonneratia caseolaris*), lam phaen (*Sonneratia griffithii*), fat (*Lumnitzera* sp.), tatum (*Excoecaria agallocha*), tin pet (*Cerbera* spp.), ngon kai and lumpho thale (*Intsia retusa*).

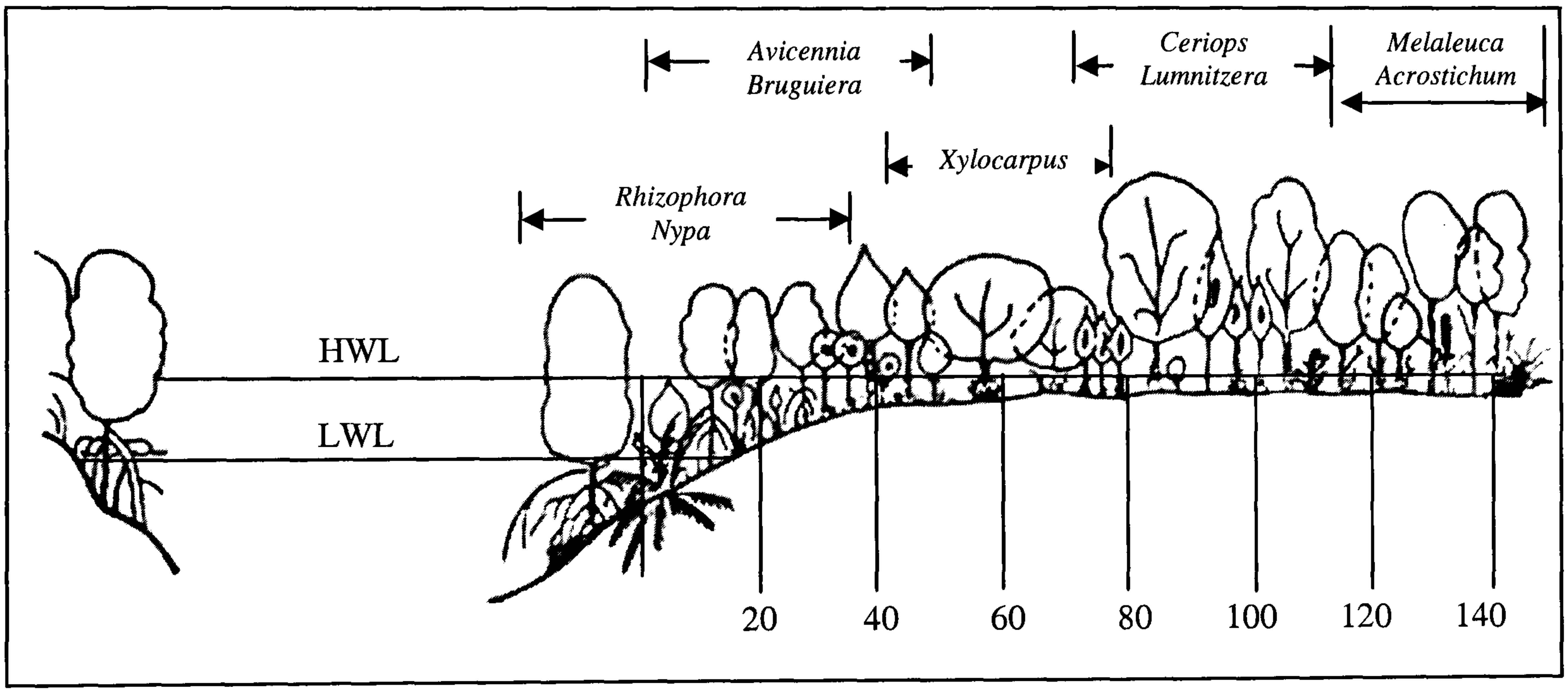


Figure 2.2 Thai mangrove zonation with distance (m) from forest margin to land (modified from Aksornkoae 1980, 1993).

Further inland, on even drier and more elevated sites that are still less subject to tidal flooding where mud has accumulated, drier soils are overgrown with ferns (*Acrostichum aureum*) and herbs and give way to evergreen forest. On the edge of creeks, the chak palm *Nypa fruiticans* is common. A major part of the mangrove is under management for charcoal production for which the species most used are *Rhizophora apiculata, Rhizophora mucronata, Avicennia marina* and *Xylocarpus* spp. Beach forests develop on sandy beaches along the coast. The main species of this narrow forest belt are son thale (*Casuarina equisetifolia*), krathing (*Calophyllum inophyllum*), Ka fak ma muang (*Dendrophthoe pentandra*) yi thale (*Pongamia pinnata*), hu kwang (*Terminalia catappa*) and pho thale (*Hibiscus tiliaceus, Thespesia populnea*)".

2.1.2 Energy flow within the mangrove system

The mangrove ecosystem can be described as an open system in terms of energy and nutrient flows (Butler et al. 1975), and is a key component in the global cycle of carbon dioxide, nitrogen and sulfur (Miles et al. 1998). It receives matter and energy from freshwater, from terrestrial habitats and also from the sea as a result of frequent tidal inundation. Odum (1969) has described the energy pathway in this system in terms of the out-welling hypothesis, where detritus and other organic material is exported to other ecosystems. Moreover, Odum claimed that mangrove and salt marsh habitats are fertile systems which export nutrients to support the productivity of coastal areas. The gross productivity of mangrove detritus is largely due to mangal leaf litter (average 10 ton/ha per year, in spite of relative low standing biomass, average 150 ton/ha) (Adeel and Pomeroy 2002), additional benthic cyanobacteria, diatoms and micro algae that live on the mangrove roots (Alongi 1998).

Several factors contribute to the potential of mangroves to act as exporters of organic matter, including tidal range, pore water concentration, the amount of rainfall, volume of water exchange, the ratio of areal extent of mangroves to that of the watershed, and the type of mangrove system (Tanaka and Choo 2000; Dittmar and Lara 2001). The extraordinary high rates of mangrove productivity, often exceeding 2 ton/ha per year, support both terrestrial and marine (both pelagic and benthic) food webs and contribute significant carbon to some offshore fisheries (Ellison 2008). Globally, carbon exported from mangrove is estimated at approximately 350-500 g c/m^2/year (Upadhyay 2002), and comparable figures from the Gulf of Thailand are >300 g c/m^2/year (Piyakarnchana 1989). The energy and nutrient pathway models developed so far are more meaningful for tropical mangrove systems where they receive substantial freshwater run-off (Odum 1971; Lugo and Snedaker 1974; Wolf et al. 2000).

2.2 Biodiversity of mangrove forests in Thailand

The mangrove vegetation in Thailand has a rich, well-developed associated flora: 87 true and associated mangrove plant species belonging to 55 genera and 41 families have been recorded (National Research Council of Thailand 2002). The mangrove tree and shrub species are differentiated ecologically into two categories; true mangrove species and mangrove associates. The best developed mangrove forests, classified as the old growth stands, remain only on the Andaman coastlines in the provinces of Ranong, Phang-nga and Trang. The mangroves along the coasts of the Gulf of Thailand are mainly classified as young growth stands due to the heavy selective cutting by humans. Other types of flora, such as epiphytic flowering plants, algae and seagrasses, are also diverse (FAO 1980; Aksornkoae 1985). Sahavacharin and Boonkerd (1976) reported 18 species of epiphytic flowering plants belonging to 13 genera and three families (*Asclepiadacae, Loranthaceae* and *Orchidaceae*). Mangrove species in Samut Songkhram was studied by Sudara et al. (1994), 29 species within 21 genera were recorded with *Avicenia alba* was the dominant species. The algal flora recorded by Lewmanomont (1976) totalled 47 species within the mangrove forest in Thailand. The study of seagrass beds in Thailand is at its fledging stage, but the ecological role of seagrass beds as nurseries and shelters for commercially important vertebrates and invertebrates has long been recognized. This ecosystem is closely connected with the mangrove ecosystem.

The fauna of the mangrove forests is rich in terms of species composition and abundance due to the diversity of food resources and microhabitats. Mangroves serve as a breeding ground and nursery habitat for marine life, which is an essential ecological support function for many coastal and offshore fisheries, but also for birds and mammals. The mangrove fauna is represented by most phyla ranging from protozoa, nematodes, nemertines, polychaetes, gastropods, bivalves, crustaceans, fish, reptiles, birds and mammals (Paphavasit 1995).

Frith et al. (1976) were the first group who reported on the zonation of macrofauna (epifauna, infauna and mangrove tree fauna) on a mangrove shore in Phuket Province. They recorded 139 species: 59 crutaceans, 42 mollusks, 22 polychaetes, 6 fish, 4 coalenterates, 1 nemertine, 2 sipunculids, 2 echinoderms, 1 platyhelminth, and 1 brachyopod. They concluded that ecological factors, notably the substratum and the exposure period at low tide, were the most important factors limiting the distribution of these animals. The macrofauna including epifauna and infauna were also studied in Don Hoi Lord, Samut Songkhram Province by Sriburi and Gajaseni (1996, cited in Worrapimphong 2005), 39 species belonging to 7 phyla of invertebrate and vertebrate were recorded.

To-on (1999) studied species composition, abundance and biomass of benthic macrofauna in the mangrove forest, Tha Chin Estuary. A total of 68 species were reported with crustaceans, gastropods, bivalves and polychaetes were the major benthic groups. Zooplankton in mangrove forest at Baan Klong Kone, Samut Songkhram Province was reported by Sikhantakasamit (2001). He reported 31 groups (11 phyla) of zooplankton with copepod dominated zooplankton populations that contributed about 40% of the total zooplankton density. Boondao (2006) studied the relationship between species composition and abundance of phytoplankton with zooplankton. A total of 342 plankton species (259 species of phytoplankton and 83 species of zooplankton) was recorded, with Babillariophyceae and protozoa the dominant groups of phytoplankton and zooplankton, respectively.

Studies on crabs in the mangrove forests in Thailand have been carried out by Naiyanetr (1979), who reported on the distribution of 11 species of *Uca*, with 8 species found in Phuket Province alone.

Insect diversity in the mangrove is also high. A survey of insects in a mangrove forest at Bangpoo was carried out by Vaivanijkul (1976), who recorded 38 species including adult moths, caterpillars, pyralids, beetles, mosquitos, biting midges and aphids. The roles of forest insects as links in the energy flow of forests and as pests in relation to mangrove management have also been investigated (UNDP/UNESCO 1991). A total of 29 species of insects found in the Ranong mangrove forest was recorded. The dipteran fauna in a mangrove forest at the mouth of Bang Pakong River in Thailand was reported by Prayoonrat (2004), with 33 species representing

32 families and 32 genera. The diversity of these insects was greatest for mosquitoes and punkies with 14 and 11 species, respectively. It should be noted that emphasis has been on the taxonomy of certain groups of mangrove-associated fauna and the population biology of those groups is far from understood completely. There are many undescribed species of benthos, both the macrobenthos and the meiofauna, plankton (where most of the identification uses been carried out to genus level) and fish. Thus, the biodiversity of the mangrove forests of Thailand is probably much higher than record.

Mangrove forests have long been recognized as nursery grounds for many marine fishes and crustaceans (Monkolprasit 1983; Well 1983; Bell et al. 1984; Aksornkoae 1993; Raffaelli and Hawkins 1996; Nagelkerken et al. 2000). Macintosh et al. (2002) studied mangrove rehabilitation and intertidal biodiversity in Ranong Province, the Andaman sea coast of southern Thailand, 30 crustacean species and 33 molluscan species were recorded. They also stated that snails of the families Ellobiidae and Neritidae were dominant at the mature forest. Community structure of prawns in the Tha Chin Mangrove Estuary was studied by Nilvanich (1999), 18 species from 9 genera and 5 families were recorded. She also stated that two planktonic shrimps, *Acetes indicus* and *A. vulgaris* were dominant in the night catches. Several studies of mangrove and fish communities in Thailand (Table 2.1) provide evidence that Thai mangrove forests are used by fish as nursery grounds, as permanent habitats or as breeding grounds in the case of some coastal species.

Two snake species, *Cerberus rhynchops* (dog-faced water snake) and *Acrochordus granulatus* (file snake) were recorded from Samut Songkhram mangrove by Sudara et al. (1994), with other amphibians and reptiles were also reported such as common water monitor (*Varanus salvator*), crab-eating frog (*Rana crancrivora*) and Asian giant softshell (*Pelochelys bibroni*).

Table 2.1 Fish diversity in the mangrove forests of Thailand.

Location	Fish diversity	References
Laem Pak Bia, Petchaburi	More than 30 families of fish larvae of economic importance such as milkfish, groupers and mullets	Vatanachai (1979)
Klong Wan, Prachaup Kiri Khan	31 species of fish larvae with tarpon, lady fish, milkfish, mullets and snappers the dominant groups	Sontirat (1982)
Phang-nga Bay, Phang-nga	44 families of fish larvae with Gobiidae, Engraulidae and Clupeidae the dominant groups	Janekarn and Booruang (1986)
Phuket to Satun	More than 60 families of fish larvae, with the major groups being Carangidae, Gobiidae, Clupeidae, Monacanthidae and Callionymidae	Janekarn and Sawangarreruk (1987)
Gulf of Thailand and Andaman coastline	72 species, 43 genera and 30 species	Monkolprasit (1983)
Ranong	111 species in 48 families with Leiognathidae, Clupeidae, Engraulidae, Centropomidae, Mugillidae and Carangidae the dominant groups	UNDP/UNESCO (1991)
Phang-nga Bay, Phang-nga	44 species in 31 families with Sciaenidae, Plathycephalidae and Ariidae the dominant groups	Janekarn (1993)
Klong Kone, Samut Songkhram	55 species in 32 families with Ambassidae, Clupeidae and Engraulidae the dominant groups	Sudara et al. (1994)
Phuket	95 species recorded, of these 71 species in seagrass beds and 69 species in mangrove creeks with Siganidae, Leiognathidae and Ambassidae the dominant groups	Poovachiranon and Satapoomin (1994)
Klong Kone, Samut Songkhram	16 families of fish larvae with Gobiidae larvae the most abundant group	Aiemsomboon et al. (1997)
Tha Chin, Samut Sakhon	42 species with Sciaenidae dominant in terms of number of species	Janekitkarn et al. (1999)
Thai estuaries	607 species in 87 families with gobies being the most diverse	Vidthayanon and Premcharoen (2002)
Trang	89 species of juvenile and small fishes with Gobiidae the most diverse	Ikejima et al. (2003)
Pattani Bay	108 species of juvenile and small fishes with most are pelagics	Hajisamae et al. (2006)

Birds and mammals utilize mangrove forests as feeding grounds, breeding areas, resting places and roosting areas, 88 species of birds, both migratory and resident, including several species of egrets, herons, kites, plovers and hawks were recorded Nabhitabhata (1982). Waterfowl in the Gulf of Thailand has been described by Round (2001), where 201 species were recorded, including 16 species which are globally threatened or near threatened. Sittilert (1985) reported a total 106 species of mangrove birds and 24 species of mangrove mammals. Two groups of mammals are true mangrove species and species found at the forest margin. The former are species found in large numbers well adapted to mangrove life, such as rats, squirrels and bats. The latter are those species that may invade the forests in their search for food and include bandicoot rats, spotted cats, civets, wild boar, crab-eating macaques and otters. 35 species of mammals have been reported by Lekagul and McNeely (1977), with macaques, otters and fishing cats very common. Nabhibhata et al. (2004) reported that there are six amphibian species known to occur to the mangroves, but only two are true residents, and 32 reptile species are known to inhabit mangroves. A survey of vertebrates (except fishes) was carried out by Sittilert et al. (1976) in Ranong, Chantaburi and Samut Prakan. A total of 7 species of mammals, 42 species of birds, 2 species of reptiles and one of amphibians were recorded. Kongsangchai and Prayoonsitti (1990) have complied a checklist of mangrove vertebrates in Thailand. They found a total of 278 species (excluding fish) from 177 genera and 68 families. These included 36 species of mammals, 204 species of birds, 32 species of reptiles and 6 amphibians. Ten species in 7 families of mammals were recorded from mangrove in Samut Songkhram Province (Sudara et al. 1994). Two species were frequently found, the crab-eating macaque (*Macaca fascicularis*) and the roof-rat (*Rattus rattus*).

2.3 Destruction of mangroves in Thailand

Mangroves of southeast Asia are spread over an area of 60,000 km^2 and account for more than 35% of the area of global mangrove vegetation. It is believed that the area under mangrove, on a global scale, is shrinking by 1,000 km^2 annually, and from 5,500 km^2 to 2,470 km^2 for Thailand during the period 1961-1986 (Tabuchi 2003). The Andaman coast experienced much less development pressure the Gulf of Thailand, approximately 80-90% of mangrove forests along the Gulf have disappeared in the last 30 years (Giri et al. 2008). According to Kongsangchai (1994), about 50% of the mangrove area in Thailand was converted to other land uses before 1991. Of the total area of Thai's mangrove forest that disappeared between 1961 and 1996, 33% was converted into shrimp ponds, 4 % to resettlement areas and 63% were used for other purposes, including agriculture, urbanization, ports and harbours (Charuppat and Charuppat 1997).

In Thailand, most of the mangrove forests are under the management of the Royal Forest Department. In recent years, the total area of mangrove in Peninsular Thailand has declined considerably, from over 367,900 ha in 1961 to 167,582 ha in 1996 (Table 2.2) (Charuppat 1998). The cause of mangrove destruction in Thailand is currently over-exploitation by traditional users and destruction resulting from activities related to the unsustainable uses of mangroves (Figure 2.3). Most of the mangrove forests around major areas of human population settlements have been lost or degraded.

2.3.1 Fluctuation in freshwater and seawater

These changes are due to the irrigation, land clearing and road construction. Land use change causes low freshwater inputs during the dry season and high input during the rainy season. Furthermore, road construction through mangrove areas obstructs tidal flow and causes changes in the forest flora and fauna. Salinity fluctuations over a wider range can cause stress to the mangrove ecosystem in enclosed bays.

Table 2.2 Existing mangrove forest of Thailand (Charuppat 1998)

Year	Area in ha	Percent of the country
1961	367,900	0.72
1975	312,700	0.61
1979	287,308	0.56
1986	196,428	0.38
1991	173,608	0.34
1993	168,683	0.33
1996	167,582	0.33

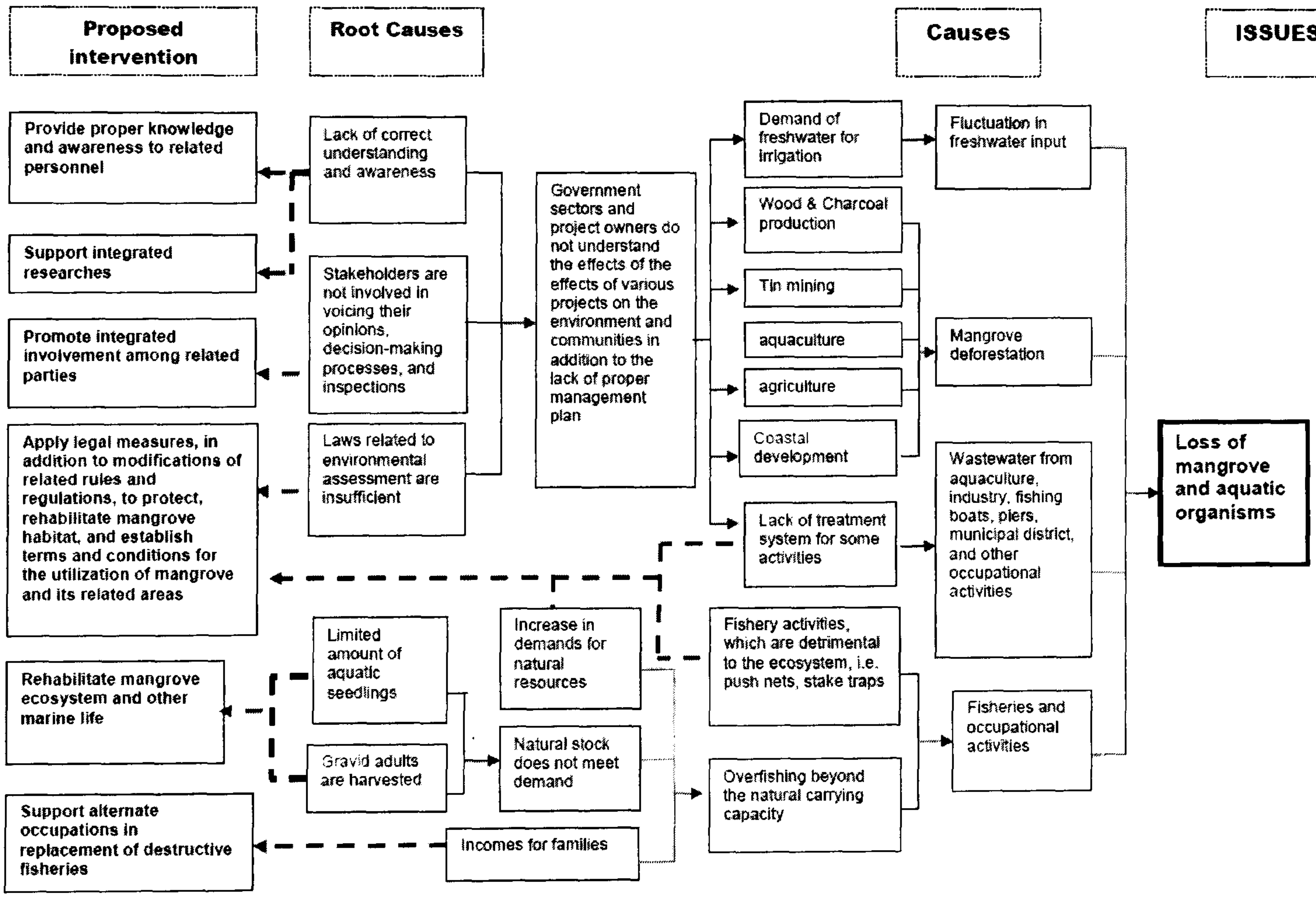


Figure 2.3 Causal chain analysis and management intervention for the loss of mangrove and aquatic organisms (Plathong and Plathong 2004).

2.3.2 Mangrove deforestation

2.3.2.1 Wood and charcoal production

2.3.2.1.1 Local uses

Mangrove forests are used directly by local inhabitants for charcoal making (90%) and the rest for fuel wood and poles (Aksornkoae et al. 1985; Choudhury 1997). Large-scale forest exploitation occurs in the form of the cutting of timber and wood for general use. Such exploitation may result in a gradual decrease in mangrove area due to unsuccessful natural regeneration. In Thailand, sustainable charcoal production, using wood from mangroves, generates an annual income of approximately US$22.4 million (Dixon and Sherman 1991).

Promotion of regeneration after a harvest of biomass for charcoal making has been a common strategy for re-establishing the mangrove forest in Thailand. The concessionaries of charcoal kilns have to establish plantations if natural regeneration after harvesting is too poor to satisfy the requirements of sustainable management. The private sector, non-government organizations and government organizations have all made significant efforts towards rehabilitation of the forest, though with minor differences in their approaches. Charcoal kiln owners have created extensive plantations in E-Sarn district, Samut Songkhram Province near Bangkok, seemingly from the need to increase the scale of charcoal production but with benefits for biodiversity and ecosystem services. The goal of rehabilitation varies from simply 're-greening', serving only environmental goals, to a mix of plantation and fish/shrimp farming serving both environmental and economic development goals (Tabuchi 2003).

2.3.2.1.2 Mangrove concession

In Thailand, the timber exploitation of mangrove forests had never been worked out for commercial purpose at the very beginning of mangrove forest management. Short-term leases had been issued for domestic consumption and the Royal forest Department did not have any control on this activities. The mangrove forests were then heavily exploited both legal and illicit practice (Havanond 1997). Before 1961, The

Royal Forest Department permitted logging in mangrove forests. A concession system allowed logging each year. In 1961-1969, a shelter wood system with minimum girth size was practiced. The rotation and felling cycle was set at 10 years with annual coupes. Each year one coupe was granted for extraction under a short-term (one year) permit. However, in 1968 the concession system was changed to long-term concessions for 15 years. In the first period (1968-1983) concessions were issued for 310 felling series with an area of 176,948 ha along the coast of Thailand, of which 154,791 ha were in Peninsular Thailand. The total production in the first period of the concessions in Peninsular Thailand was 10,068,559 m^3. The second period of concession was between 1986 and 2001, amounting to an area of about 142,250 ha, 8% less than the first period. This system was practiced until 1991, when it temporarilly ceased due to the degradation of mangrove resources throughout the country. The conflict between destructive development and conservation pressurised the government to develop a number of mitigation policies including the cancellation of mangrove forest concessions throughout the country. In 1998, the cabinet announced that the concessionaires could further their charcoal production in their concession areas until their concessions expire (Plathong and Plathong 2004).

2.3.2.2 Tin mining

Hydraulic tin mining in the mangrove area has been carried out mostly in Ranong, Phangnga, and Phuket provinces. There are 24 mining areas, approximately 926 ha in 1979 in 23 mining concessions (Aksornkoae et al. 1985; Plathong and Plathong 2004). Although mining accounts for only a small proportion of mangrove destruction, its impact on the mangrove ecosystem can be considerable. Mining requires clear-cutting of mangrove forests followed by dredging operations which disturb the mangrove soil, introducing silt into the water which is then transported to neighbouring environments. Mining sediments directly affect species composition, population and forest structure (Snidvongs 1982). The dominant impact of mining activities is the deposition of sediment. Excessive sedimentation is detrimental to mangroves through blocking exchange of water, nutrients and gases within the substrate and between the substrate and the overlying water. Partial cessation of this exchange causes stress, manifested by reduced productivity and reduced survival. Mining activities are also frequently associated with increased turbidity and increased siltation caused by dredging and overburden disposal (Chansang 1988). Furthermore, mangrove detritus-based food webs are disrupted and overall there may be a reduction in fishery yield. The reforestation in abandoned mining areas is costly: it takes a very long time for the plants to grow and for the ecosystem to recover. At present, the government has policies to stop mining concessions in mangrove areas (Plathong and Plathong 2004).

2.3.2.3 Aquaculture

Fish and shellfish farming is increasing around the world. Catches of finfish and shellfish are declining, but aquaculture production of fish and shellfish is increasing. In Thailand, the global demand for shrimp products coincides with national economic policies to promote coastal regions as sites for export processing. The Thai government subsidized export processing zones along the coast during the 1980s through the development of transportation, financial and institution infrastructures, which are directly related to the growth in shrimp farms and declines in mangrove forests (Curran and Cruz 2002). Since 1991, Thailand has become one of the world's largest producers of cultured shrimp, and this has come at a cost of reduced mangrove forest area. Between 1961 and 1996, Thailand lost around 20, 500 km^2 of mangrove forests, or about 56% of the original area, mainly of shrimp aquaculture and other coastal developments (Charuppat and Charuppat 1997). Estimates of the amount of mangrove conversion caused by shrimp farming vary, but recent studies suggest that up to 65% of Thailand's mangroves have been lost to shrimp farm conversion since 1975 (Charuppat and Charuppat 1997; Barbier 2003) and causing a loss of 65,000 ha of mangroves in Thailand (Upadhyay et al. 2002).

After 1985, large-scale intensive shrimp farming began to accelerate in Thailand with the adoption of an aquaculture promotion policy and the first area to be intensively developed was the inner Gulf of Thailand (Hossain and Lin 2001), due to its proximity to Bangkok. During the period 1975-1993 this region lost 85% of its mangroves (Figure 2.4) (Huitric et al. 2002). Figure 2.5 shows the peaking of the industry in this area, its collapse in 1989 and its subsequent spread to the western, then eastern coasts of the Gulf. Thus, there has been a sequence of exploitation; moving into one area,

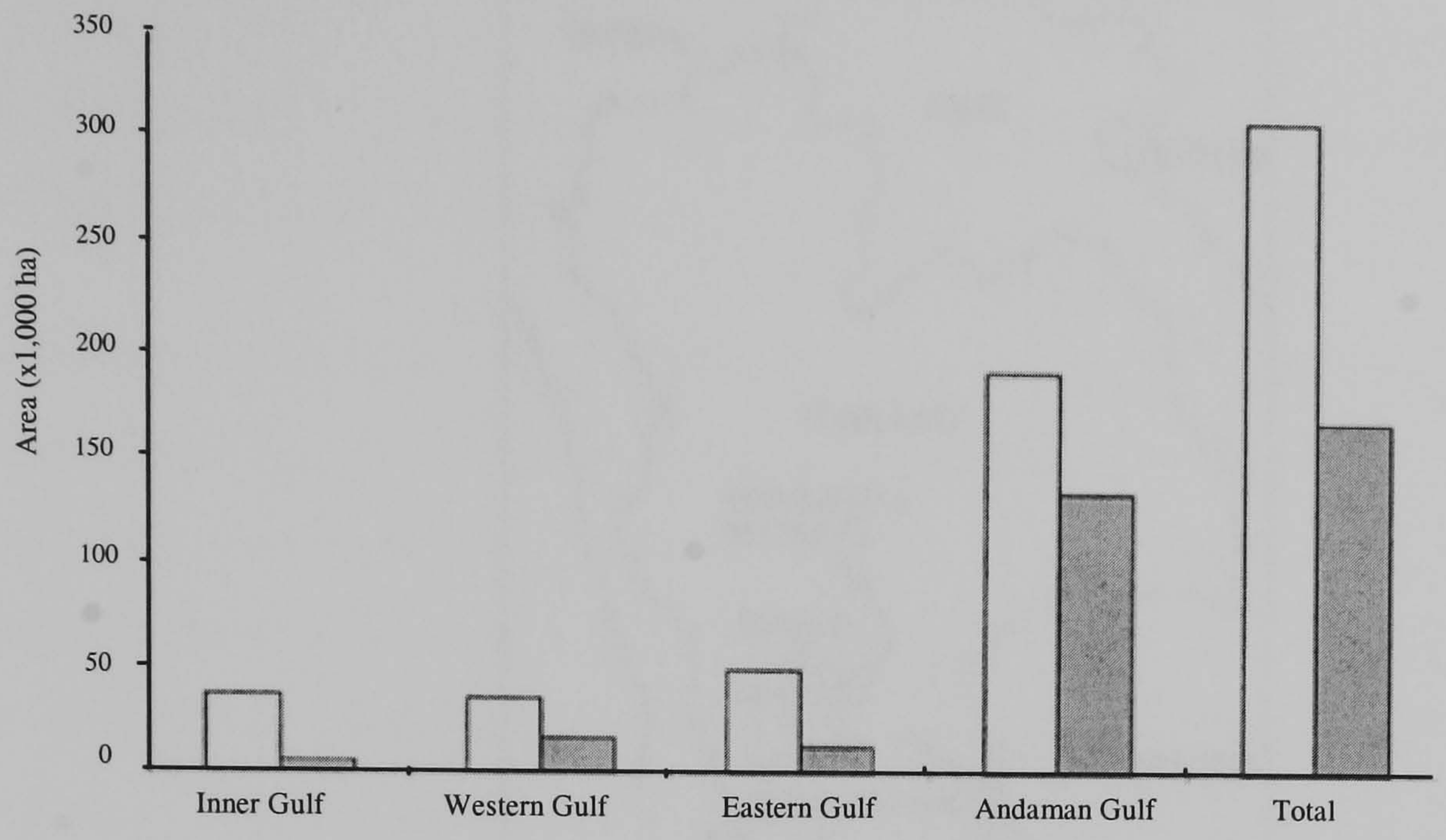


Figure 2.4 Regional spatial changes in mangrove areas in Thailand from 1975 to 1993 (Royal Forest Department 1997).

degrading it, then moving to the next and degrading it and so on (Grima and Berkes 1989). "Mangrove forests have been destroyed or surrounded with embankments to make shrimp ponds and after 2-3 years production drops. After 5 years, most ponds are completely abandoned due to effects of increasing acidity from mangrove soils on water quality, declining productivity, self- pollution and virus disease problems. Diseases spread rapidly when the ponds are sited close together" (Plathong and Plathong 2004). Stevenson (1997) provides an excellent account of these issues "By 1996, massive shrimp mortality in Thailand by the Yellow Head Baculovirus (YHDBV), was estimated as being responsible for a loss of 50-80% of production, amounting to £21 million in 1992". Macintosh (1996) stated in his papaer that "many disused ponds now lie unproductive in Thailand. However, unofficial estimates of pond disuse have suggested that the percentage of ponds left idle after a period in production can be as high as 70%, although it would appear that some disused shrimp ponds in Thailand (and probably elsewhere) are subsequently converted to other uses, such as redevelopment into factory or housing estates. In Samut Sakhon, large tracts of abandoned shrimp ponds are being converted to non-agricultural land use, such as housing estates and industrial development. Some abandoned shrimp farms have been converted to salt farms or fish culture operations".

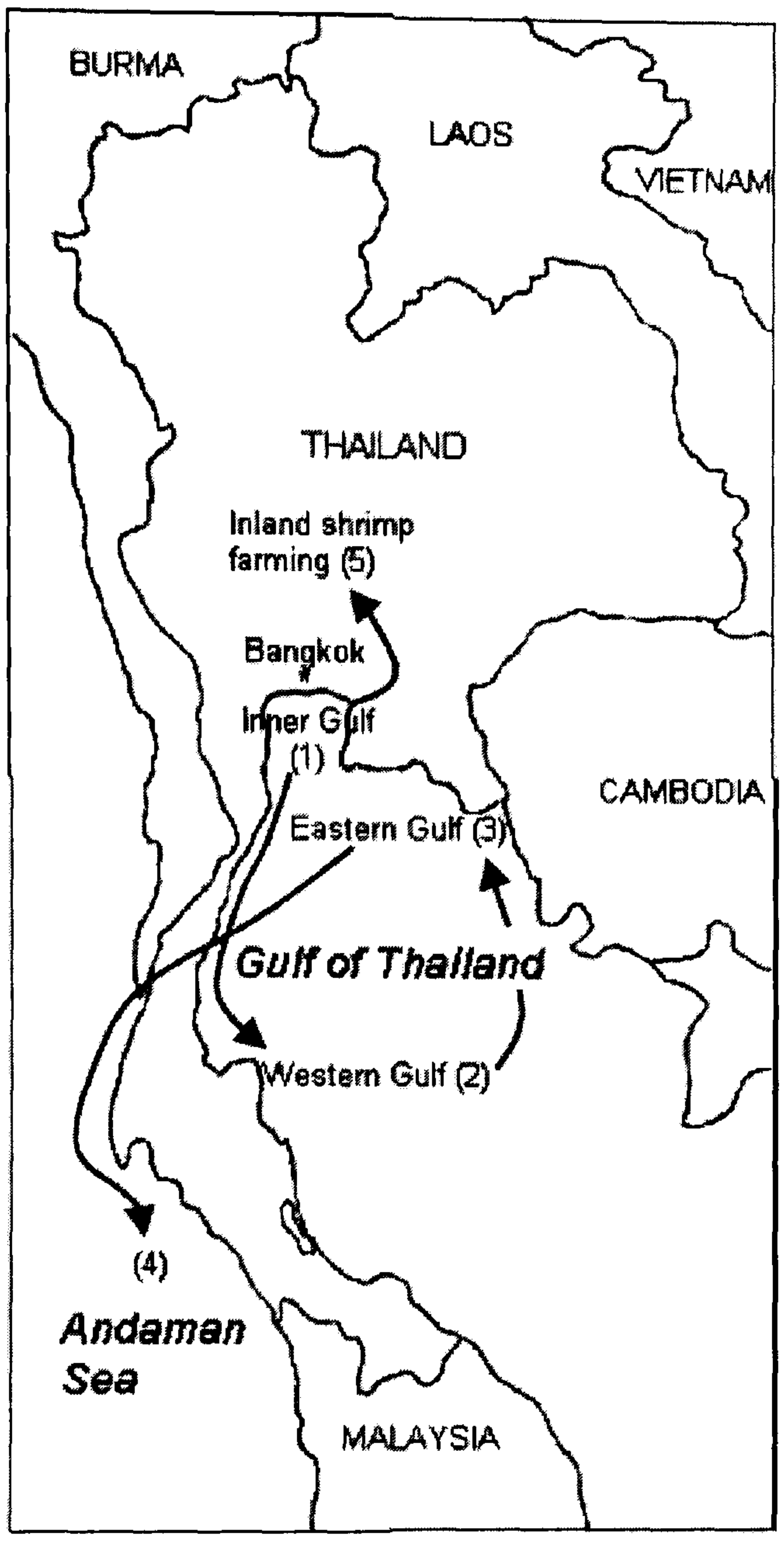


Figure 2.5 The sequential (1-5) exploitation of mangrove forests by shrimp farms (Huitric et al. 2002).

There have been several reports from Thailand that have included estimates of the scale of pond disuse and/or abandonment (both in mangrove and non mangrove) (Stevenson 1997). Briggs and Funge-Smith (1994) reported that an area of 40,000 to 45,000 ha south of Bangkok became derelict after shrimp production collapsed in 1989/1990. Pataros (1995) stated that around 19,900 ha of shrimp farms in the five provinces of the inner Gulf of Thailand were closed in 1990-91. A report produced by NACA (1996) details that in 1989 about 62% of farms were operating "under capacity" and another 22% of farms were abandoned in Samut Sakhon Province (OEPP 1994). Stevenson (1997) estimated that "currently 70-80% of ponds are abandoned in Prachuap Khiri Khan, with a similar figure for the provinces of Songkhla and Nakhon Sri Thammarat. However, it must be remembered that the situation changes rapidly from month to month and ponds are frequently converted to other uses and shrimp production can be recommenced at any time". In 1996 there were 20,800 ha of abandoned shrimp ponds in Thailand with an economic loss of about THB 5,000 million (Hossain and Lin 2001).

Intensive shrimp farming produces both direct and indirect impacts on mangrove and other coastal ecosystems (Plathong and Sitthirach 1998). Shrimp farming, including pond construction requires extensive coastal areas, and this leads to mangrove deforestation; reduction of habitats; declines in shoreline production; increased coastal erosion; misuse of chemicals and antibiotic, and coastal pollution; nutrient enrichment; depletion of wild prawn and fish stocks; land subsidence; salinisation of soils, agricultural land and ground water, activation of acid sulphate soils; loss of agricultural land; introduction of exotic species and spread of disease (Stevenson 1997).

2.3.2.4 Agriculture

If agriculture plays a dominant role in economy of the area, the main land use type will be dominated to agriculture land (Durongdej 2000). Plathong and Plathong (2004) stated that "Peninsular Thailand has very few agricultural areas located in former mangrove areas because of the acidity of the soil which results in low productivity. However, some rice fields can be found in the mangrove areas of Satun province. In the provinces of the Andaman coast, such as Phang Nga and Krabi, there are also palm tree, coconut tree and rubber tree plantations in mangrove areas. Converting mangroves into agriculture land involves the digging of narrow canal and piling up the spoil material to form bunds on one or both banks of the canals. The bunds generally prevent seawater intrusion. This may lead to extensive loss of mangrove areas and their productivity. In addition, the canals cause a change in the freshwater regimes of unreclaimed seaward mangroves and can have deleterious effects on the system".

2.3.2.5 Coastal development

Following development of coastal cities in Thailand, mangrove lands have been converted for domestic and industrial development occurs. The most common forms of conversion are housing and residential development and coastal tourist facilities, including small port development. Mangroves can be totally reclaimed by road construction which also obstructs tidal and freshwater flows. Mangrove areas have traditionally been considered as wasteland rather than as highly prized ecosystems, and much solid waste and garbage refuse has been dumped into mangrove ecosystems (Plathong and Sitthirach 1998, Plathong and Plathong 2004).

Port and channels development are generally constructed in response to the needs for passageways and docking locations, which destroy mangroves and other coastal forests. They serve as routes for transporting marine catches to consumers and for tourism. Such constructions and dredging require specific expertise in selecting appropriate sites, construction, and dredging processes. Prior to these processes, environmental impact assessments should be conducted to reduce impacts from chemicals and contamination from materials used during the construction (Plathong and Plathong 2004).

2.3.3 Waste water

2.3.3.1 Waste water from shrimp farms

Waste water from shrimp farms is discharged directly into coastal areas, increasing organic matter and nutrient concentrations including pollution caused by the chemical products used. This is turn leads to hypoxic conditions indicated by the black color of the sediment (Plathong and Plathong 2004; FAO 2007). It is therefore important that the government sector, or related organizations, provides proper guidance concerning aquaculture techniques and their environmental impacts. Technologies for individual and communal wastewater treatment before discharging waste water into natural water sources should be introduced widely. Policies related to shrimp farming should also be strictly enforced (Ruenglertpanyakul et al. 2004).

2.3.3.2 Sewage from urban and industrial areas

Increased accumulation of pollutants within the mangrove ecosystem, especially through food chains, is likely to occur due to coastal development. The coastal and marine environment of the Gulf of Thailand is degraded by pollutants from both land-based and marine sources (Plathong and Plathong 2004). Land-based sources, most of them domestic, contribute 70% of marine pollution. It is estimated that more than 200,000 tons of waste (BOD) is discharged into the Gulf annually (Thailand Environment Monitor 2000). The generation of household solid waste and industrial hazardous waste has increased significantly and poses a major threat to surface and groundwater quality. Only a handful of environmentally-safe disposal facilities are available (Ossterveer et al. 2006).

Solid waste has steadily increased from about 30,000 tons/day in 1992 to close to 40,000 tons/day in 1997. This totals about 13 million tons/year, of which about 25% comes from Bangkok, 35% from other urban areas, and the remaining 40% from rural areas (Thailand Environment Monitor 2000).

2.4 Marine fisheries in Thailand

Marine fishing grounds that fall within the Thailand's Exclusive Economic Zones (EEZs) (Figure 2.6) covers 420,280 km^2: 304,000 km^2 in the Gulf of Thailand and 116,280 km^2 in the Andaman Sea. Its maritime border is shared with Cambodia and Vietnam in the southeast, Myanmar in the west and Malaysia in the south. EEZs within the Gulf of Thailand include overlapping areas between Thailand and Cambodia (34,000 km^2), Thailand, Cambodia and Vietnam (14,000 km^2) and Thailand and Malaysia (4,000 km^2) (Nakthon 1992).

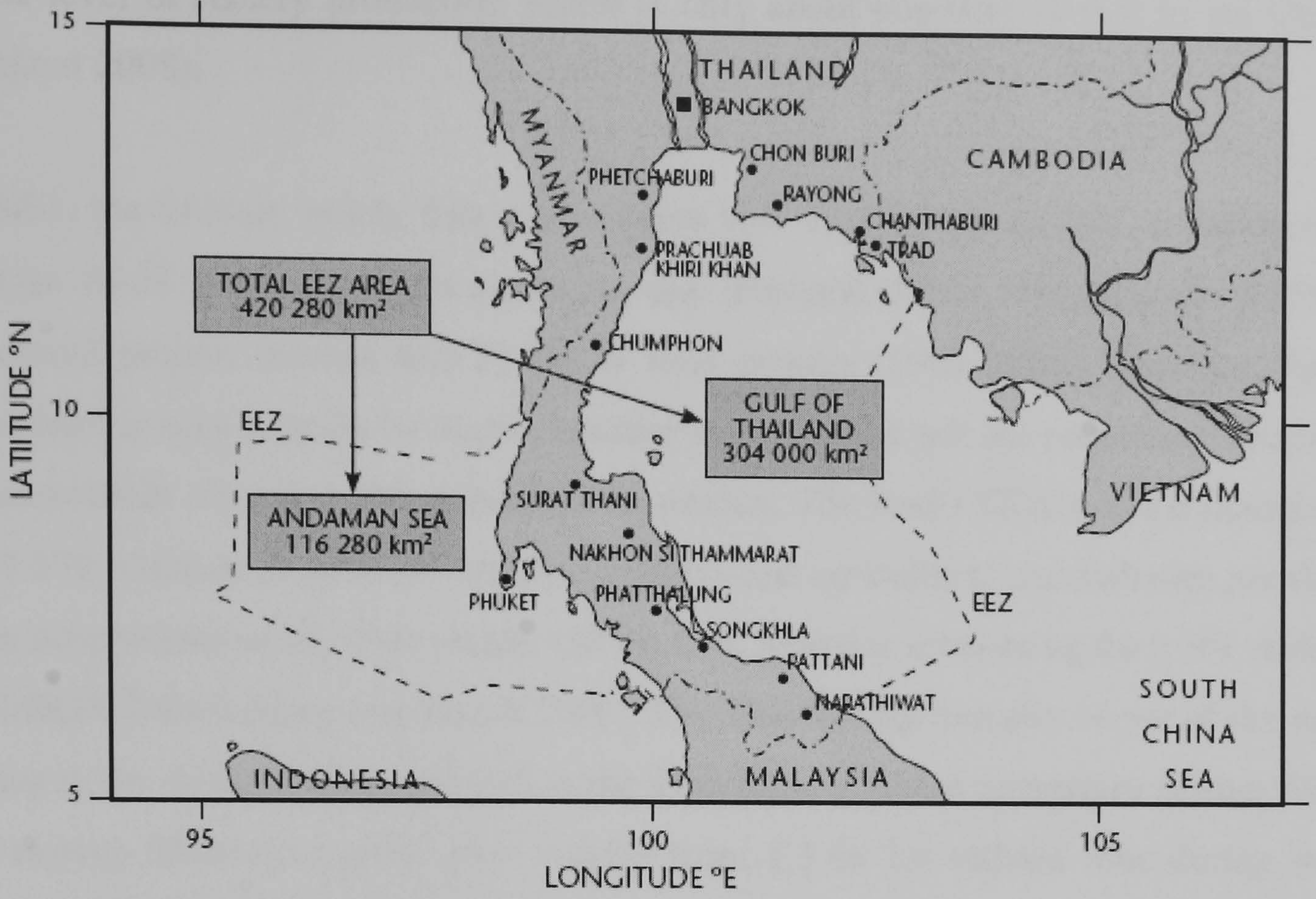


Figure 2.6 The Exclusive Economic Zone (EEZ) of Thailand (Janekitkosol et al. 2003).

The Gulf of Thailand has a coastline of 2,700 km (OEPP 1998). The waters along this coast are on the whole shallow to a good distance from the shore. The waters are rich in nutrient salts brought in by many rivers. Water from the west, the northwest mountains and the high eastern plain flow into the Gulf of Thailand through four river systems: Chao Phraya, Tha Chin, Mae Klong and Bang Pakong. The innermost part of the Gulf is a large area of intertidal mudflat around the shores of a huge, shallow sea that suitable for fishing by gill net, push net, and similar gears operated by small boats. Above all, these waters have proven to be ideal for trawling (Burnette et al. 2007; Panjarat 2008).

The Andaman sea (west coast) are very different from those in the Gulf of Thailand. The coastline is only 865 km long and the rather narrow continental shelf descends into a steep continental slope. Inshore, areas within 3 km of the coast have an average depth of about 3m. It is slightly wider in the north and narrower in the south, the latter comprising mangroves and seagrasses. The seabed for the most part is quite rough, with scattered coral rocks. The relatively unfavorable conditions are reflected in the level of fishery production which is only about one-fifth of that in the Gulf (Panjarat 2008).

In 2001, the average yearly fish consumption was 32.4 kg per capita, providing on average 10-14 g of protein per capita per day (Panjarat 2008). Fish provides 40.5% of animal protein sources and 17.6% of total protein (FAO 2001). However, fish consumption may actually be higher because many caught fish are consumed directly by households without passing through the market. Thailand's GDP was estimated at USD 176.6 billion in 2005 (World Bank 2005) and agricultural and fisheries are the main occupations of the Thai people (35%), with fisheries accounting for 2.5% of the total GDP (Flewwelling and Hosch 2006). The Thai fishing industry is one of the ten largest in the world. Ninety percent of the Thai fishery output comprises marine fish and marine fisheries capture grew rapidly from 1.3 to 2.6 million tons during the period 1970-1987 (FAO 2002).

The coastal fisheries in Thailand can be divided into two broad sectors: large-scale fisheries and small-scale fisheries (Juntaraschote 1984). Small-scale fishery (SSF) establishments are those without a boat, using a non-powered boat, outboard-powered boat or an inboard powered boat with a total gross tonnage (GT) < 10 GT. SSF mostly operated in shallow water, conducts fishing at approximately 5 km from the shoreline in one-night operation. The fish are landed at the village and sold directly to the consumers by the owner's wife. Fishing activities are for sufficiency. Large-scale fishery (LSF) establishments are those using an inboard powered boat of 10 GT and over (Tokrisna and Duangsawasdi 1992). The small-scale fisheries establishments are the largest group accounting for 92.41% of total, with the large-scale fisheries establishments comprising the remaining 7.59% (National Statistical Office and Department of Fisheries 1997). Large-scale fisheries, however, are responsible for most of the production (currently 92.7%) from the marine capture sector (Department of Fisheries 2005).

An excellent account of the development of the fisheries is provided by Department of Fisheries (2006), Thailand Environment Monitor (2006) and (Panjarat 2008) is reproduced here “Marine fisheries in Thailand have developed and expanded due to the use of new fishing gears and technologies, movement of fishing fleets into new fishing grounds, improvement of fishing vessels and the development of support facilities and infrastructure. Up to the Second World War, marine fisheries in Thailand were carried out mainly in shallow coastal waters with traditional gears such as bamboo stake traps, set bag nets, castnets and hooks. The situation changed dramatically in the early 1960s when the government promoted fisheries development, particularly deep-sea fishing, in order to increase production destined for the fast growing domestic market and for export. Among the newly introduced gears, the most far-reaching effect was created by the otter-board trawl. The Thai fisheries industry is one of the ten largest in the world. Fishery output is more than 90% marine fish. Marine fisheries capture grew rapidly from 1.3 to 2.6 million tons during the period 1970-1987. During the period 1994-1996, the total capture productivity of Thailand reached a peak of 2.8 million tons and dropped slightly to 2.6 million tons in the following year. The Gulf of Thailand contributed approximately 70% of this total catch, while the Andaman sea accounted for the remainder (Figure 2.7). Marine catch in Thailand is classed as tropical, multi species and can be categorized into five main groups of pelagic fish, demersal fish, cephalopods and crustaceans. The total catch of 2.6 million tons includes pelagic fish (33%), trash fish (30%), demersal fish (18%), cephalopods (7.5%), miscellaneous fish (7%) and crustaceans (4.5%)” (Table 2.3).

For a number of decades, fisheries development in the Gulf of Thailand has concentrated on increasing fishing effort to maintain or increase the productive volume. Increasingly, the total catch has a higher proportion of “trash” fish (consisting of by-catch and undersized juveniles of various demersal and some pelagic species, much of which goes to fish meal or duck feed or is thrown overboard), aggregated across all species and gear types (Ahmed et al. 2007). Catches from Department of Fisheries research trawl surveys comprises 30-40%

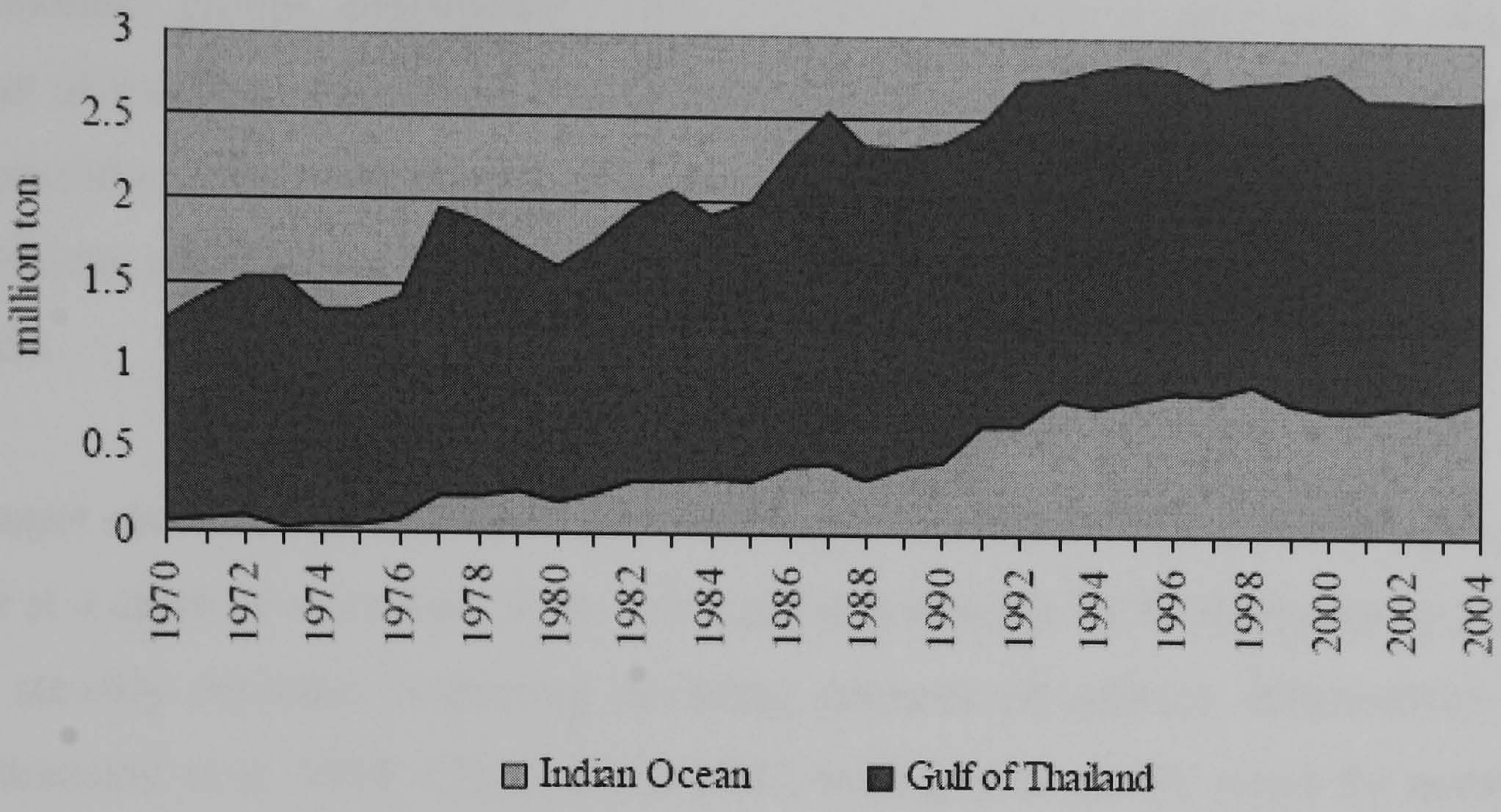


Figure 2.7 Thailand capture fisheries production (including inside and outside Thai's EEZ) (FAO 2005; DOF 2006).

Table 2.3 Thai marine capture by category in 2004 (Department of Fisheries 2004).

Category	**Catch**	
	Ton	%
Total	2,635,969	100.0
Pelagic fish	878,254	33.0
Demersal fish	482,949	18.0
Cephalopod	200,041	7.5
Crustacean	119,526	4.5
Miscellaneous fish	181,674	7.0
Trash fish	771,723	30.0

"trash" fish, of which about one-third is juvenile and undersized fish. Data from commercial fisheries also show that "trash" fish contain at least 30% juvenile fish. Pair trawl catches have the highest composition of juvenile fish, representing 70% of total "trash" fish. Otter board trawls catch juvenile fish which amounts to about 40% of total "trash" fish (Sripanpaiboon 1995; Eiamsa-Ard and Amornchairojkul 1997). Percentage of the average by weight of families in the "trash" fish (Table 2.4) showed that the most dominant family was Leiognathidae (25.06%), which is a "true trash fish" family, while Engraulidae, Mullidae and Synodontidae, which are "economic" groups, contributed 7.19%, 5.70% and 5.04%, respectively. As the main target of trawlers is demersal fish, so the main portion of trash fish is demersal fish. Engraulidae, which are pelagic fish, were also found in large amounts because pair trawls can also catch a lot of pelagic fish, especially *Stolephorus* (Khemakorn et al. 2005).

A major source of the decline in demersal fisheries since 1973 is overfishing by trawl gear at a depth of more than 50 m. catch per unit effort (CPUE) (kg/hour) by trawlers has steadily declined, indicating declining resource abundance (Meemeskul 1982; Vadhanakul et al. 1985; Chotiyaputta 1992; Intong et al. 1993), while the number of trawlers of all sizes and types has continued to increase over this period. Trawlers began to use smaller cod-end mesh sizes so that more "trash fish" could be caught to at least partly compensate for the declining production of targeted species and sizes of demersal fish (Ahmed et al. 2007).

Because of the rapid extension and development of marine capture fisheries without proper controls, Thailand has faced problems with the development of marine fisheries since 1982 (Janekitkosol et al. 2003). Marine fish resources are over-exploited, and while the catch has increased, CPUE has decreased. Most of the important pelagic fish (fish in the middle and upper parts of the water column) in the Gulf of Thailand, namely Indo-Pacific mackerel or "Pla Tu", anchovies, round scad and sardines, are fully exploited. Indian mackerel is not yet overfished (Chullasorn 1997). Almost all the demersal (bottom dwelling) resource stocks, namely fish, shrimps, squid, cuttlefish and others, are overfished (FAO 1995). At the same time

Table 2.4 Species groups composition of trash fish from marine fisheries capture in the Gulf of Thailand (Khemakorn et al. 2005).

Family	% Trash fish	Family	% Trash fish
Leiognathidae	25.06	Muraenesocidae	0.74
Misc. trash	7.74	Gobiidae	0.74
Engraulidae	7.19	Siganidae	0.34
Mullidae	5.70	Soleidae	0.29
Synodontidae	5.04	Misc. pelagic	0.29
Apogonidae	4.48	Blenniidae	0.24
Crabs trash	4.39	Octopodidae	0.22
Bothidae	3.94	Sphvraenidae	0.21
Carangidae	3.56	mollusc	0.19
Balistidae	3.22	Dasyatidae	0.17
Tetraodontidae	3.13	Synanceiidae	0.15
Nemipteridae	2.68	Misc. other	0.15
Priacanthidae	2.60	Psettodidae	0.14
Penaeidae	1.68	Lutjanidae	0.13
Platycephalidae	1.60	Terapontidae	0.07
Misc. demersal	1.44	Serraniidae	0.06
Sepiidae	1.34	Scyllaridae	0.06
Loliginidae	1.31	Gerreidae	0.05
Sciaenidae	1.30	Bregmacerotidae	0.05
Cynoglossidae	1.24	Sillaginidae	0.03
Scombridae	1.17	Menidae	0.02
Trichiuridae	1.13	Hemiscyllidae	0.02
Stomatopoda	1.10	Haemulidae	0.02
Fistulariidae	0.94	Pleuronectidae	0.01
Callionymidae	0.92	Chirocentridae	0.01
Clupeidae	0.88	Brachyura	0.01
Scorpaenidae	0.81		

the cost of fishing has increased following increases in fuel prices. Conflicts among the fishers who exploit coastal fishing grounds are increasing while the freedom to fish in more distant waters is disappearing because of Exclusive Economic Zone (EEZ) proclamations of neighboring countries. Indeed, disputes with neighboring countries have arisen because of fishing by Thai vessels.

Whilst the above fisheries issues are difficult enough, the problem has been further complicated by the 2004 Indian Ocean earthquake (Dahdouh-Guebas et al. 2005;

Stobutzki and Hall 2005). The Sumatra-Andaman earthquake occurred on December 26th 2004, with an epicentre off the west coast of Sumatra, Indonesia. The earthquake triggered a series of devastating tsunamis along the coasts of most land masses bordering the Indian Ocean, killing large numbers of people and inundating coastal communities across South and Southeast Asia, including parts of Indonesia, Sri Lanka, India, and the six provinces along the Andaman Sea coast of Thailand (UNDP 2004).

In Thailand, this disaster causes loss of life as well as major damage to property, the environment and the economy. Around 58,550 people have been affected. Nearly 500 fishing villages along the Andaman coast were seriously affected: about 30,000 households dependent on fisheries have lost their means of livelihood with over 10,000 fishing boats and 7,000 sets of fishing gear destroyed or damaged (United Nations Thailand 2008). The severe impact on the natural environment in turn had serious consequences on the fishing and tourism industries and, therefore, thousands of families' livelihoods.

In many affected areas traditional social communities were wiped out. Although there have been many recovery and rehabilitation projects undertaken by the Thai Government, international organizations and NGOs, the tsunami has created many long-term difficulties for fishers (Panjarat 2008).

2.5 Study area: The Inner Gulf of Thailand and Mae Klong Estuary

2.5.1 The Inner Gulf of Thailand

2.5.1.1 Location

The Gulf of Thailand, situated between latitudes 5°00′ and 13°00′ N and longitudes 99°00′ and 106°00′ E (Cheevaporn and Menasveta 2003), is part of the Sundra Shelf, located in the westernmost portion of the Pacific Ocean, covering an area of about 320,000 km^2, a 1,840 km coastline (Chongprasith and Praekulvanich 2003). It extends southeast from the Chao Phraya deltalic plain near Bangkok, approximately 800 kilometers to its mouth. The Gulf is a relatively flat basin with an average depth of about 45 m, and a maximum depth of 85 m, and the average width is approximately 400 km. The Gulf drains parts of Malaysia, Thailand, Cambodia and Vietnam and opens only to the South China Sea (Wattayakorn et al. 1998). It can be divided geographically into two parts: the Inner Gulf (or the Upper Gulf) and the Outer Gulf (or the Lower Gulf) (Robinson 1974).

The Inner Gulf has an inverted U-shape and is the catchment basin of four large rivers on the northern side; Chao Phraya emptying into the Bight of Bangkok, Bangpakong draining southeastern Thailand, Mae Klong and Tha Chin draining Bilauktaung mountains on the western (Burmese) border of Thailand, all discharging into the South China Sea with some influence on the lower Gulf (Brinton and Newman 1974; Burnett et al. 2007).

2.5.1.2 Bottom topography

The Inner Gulf is a relatively shallow embayment. It is shallower than the Lower Gulf with a maximum depth of about 40 m and average depth of about 15 m. The bottom topography slopes gradually downward from the shallow northern coast to a depth of 30 m at its mouth, which is between Sattahip and Hua Hin. The western side of the bay (with an average depth approximately 15 m) is shallower than the eastern side (with an average depth approximately 25 m) (Siripong 1985).

2.5.1.3 Climate

The climate of the Inner Gulf of Thailand is strongly influenced by two major Asiatic monsoons, the southwest and northeast monsooon. The southwest monsoon is usually dominant during May-September. It brings warm moist air originating from the Bengal Bay into the region, resulting in heavy rainfall. The northeast monsoon is usually dominant during November –February. Normally, the wind blows from the east. It brings cool and dry air from the Siberian anticyclone into the Gulf, resulting in cool weather and dry conditions. During February-May (the transition period), a shift from the northeast monsoon to the southwest monsoon occurs. The northeast monsoon starts shifting to east and southeast directions in the beginning of February and is seen as a southeast wind with rough sea surface conditions. This wind originates from the high pressure area in the South China Sea. In May, the southeast wind shifts to the south and southwest to become the southwest monsoon. During the monsoon transition period, wind patterns are highly variable and difficult to predict.

This monsoonal pattern gives rise to the 3 seasons in which sampling was undertaken in this present study. The rainy (wet) season, or the southwest monsoon season begins in mid-May and end in October. The winter (dry) season, or the northeast monsoon season, begins in October and ends in February. The summer (hot) season, or the transition period begins in February and ends in mid-May. (Meteorological Department 1987)

2.5.1.3.1 Rainfall

During the period November –February, the northeast monsoon normally sits over the Inner Gulf. It gradually develops during the transition period and reaches its maximum during December and January. Cool and dry weather appears within the area from the Gulf of Thailand northward. Thus, the November rainfall decreases sharply relatively to that in the previous month. The following month, December, rainfall further decreases to 20% of that in the previous month and is the driest month, coinciding with the maximum development of the Siberian high. Rainfall generally increases again in January.

The summer (hot) season starts from March to May, the transition period of shifting from the northeast monsoon to the southwest monsoon season. The wetness within the Inner Gulf during March-April begins to increase gradually, but is still less than 100 mm. By the end of this season, rainfall increases to 100-200 mm.

Patterns of rainfall in Thailand (Figure 2.8) are caused by the southwest monsoon and tropical cyclones (Weerakul and Lowanichchai 2005). The southwest monsoon normally begins to prevail over the Inner Gulf in May. The wind brings the moist air to the Inner Gulf. The monsoon significantly affects the Inner Gulf, particularly the eastern part, much more than the western part. The intensity of the monsoon during this period, together with topographic effects, means that the eastern part receives

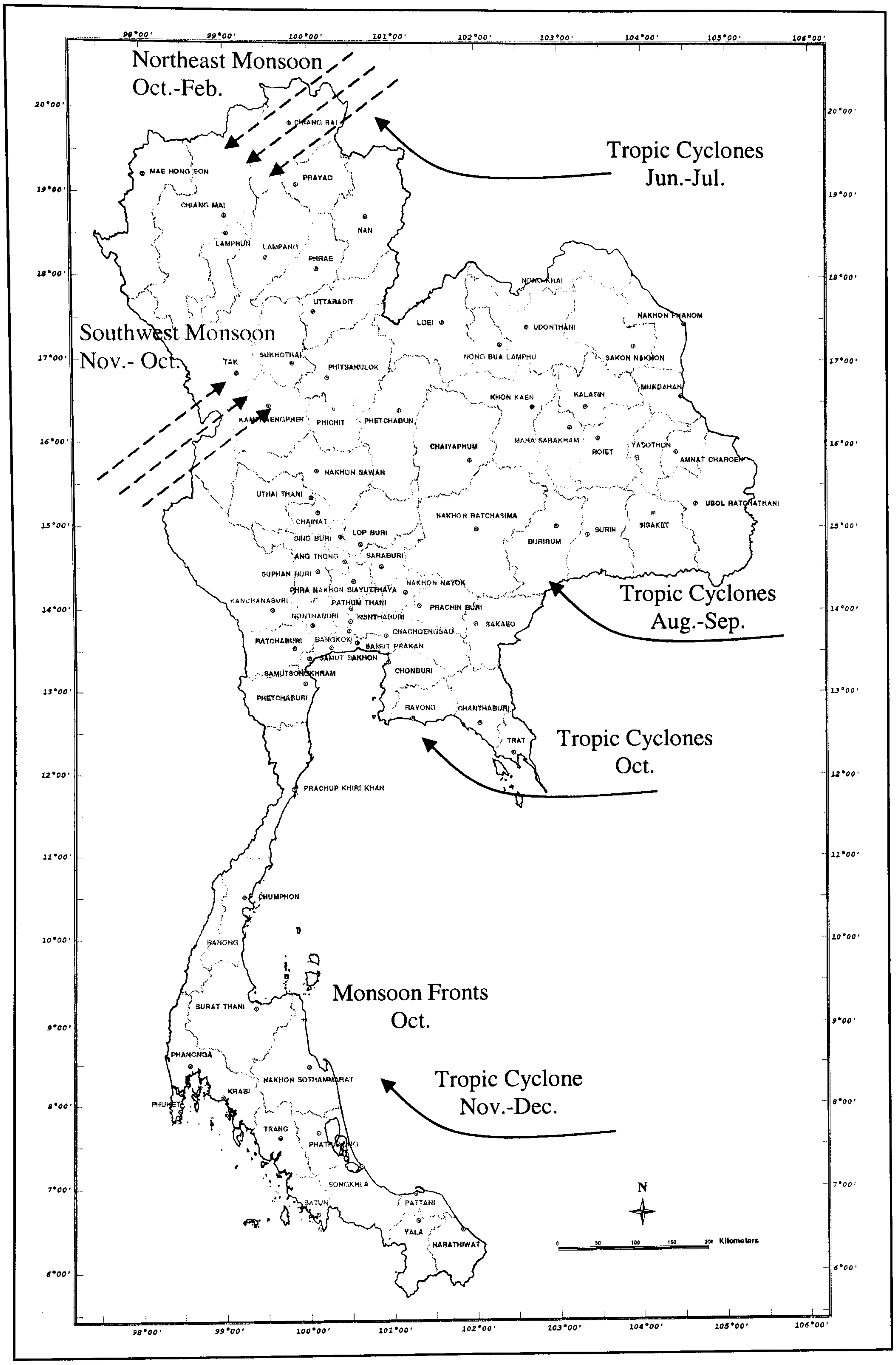


Figure 2.8 Monsoon winds and tropical cyclones influence the occurrence of rainfall in Thailand (modified from Weesakul and Lowanichchai 2005).

much more substantial rain with the maximum rainfall occuring in August and September.

October is the transition month of shifting from the southwest monsoon to the northeast monsoon. Rainfall in the Inner Gulf begins to decrease, but still remaining wet, with rainfall in excess of 200 mm (Meteorological Department 1987).

2.5.1.3.2 Air temperature

The annual mean temperature along the coastlines varies slightly between 26 and 28° C. The monthly mean temperatures in the Inner Gulf are lowest during the winter and highest in the summer (April). The difference between the hottest and coolest months is about 4° C. The mean monthly temperatures from January to April increase rapidly, on average 1° C/month, due to the prevailing southeast wind. During the rainy season, the temperatures tend to drop gradually from about 28-29° C at the beginning of season to 27-28° C at the end of the season. This is because cloudiness reduces the intense heating of the surface. The mean monthly temperatures for the rest of the year, October to December, also decrease slightly to 26° C due to the prevailing northeast monsoon (Meteorological Department 1987).

2.5.1.4 Oceanographic features

The Inner Gulf of Thailand is surrounded by land on its northern, eastern and western sides, opening to the Lower Gulf via the southern border. The Gulf is affected by the freshwater of the four major rivers along the northern boundary. Thus, the transportation of water mass is mainly controlled by the combined effect of river runoff and tidal currents.

In the dry season (low river discharge), the water body is well mixed vertically, only occasionally being slightly stratified, particularly in the beginning of the rainy season. In the rainy season (high water discharge), the water is highly stratified, with a strong halocline between the upper and deeper waters (Wiriwuttikorn 1996).

2.5.1.5 Water current and tides

The tides in the Inner Gulf are mixed and dominated by semidiurnal tides. The mean tidal range is highest at the Gulf head (about 1.5 m) and lowest near the mouth (about 1 m).

The surface circulation of the Gulf is influenced by the patterns of the monsoon wind, the direction and magnitude of which change according to the northeast wind (November to February) and the southwest wind (May to September) (Figure 2.8). During the two transition periods (March-April and October- November), the directions of currents are weak and variable. The strength of the surface current is generally stronger in the northeast monsoon season compared to the southwest monsoon season. However, water circulation in the Inner Gulf is driven by the combined effects of river discharge, wind drift and tidal currents (Siripong 1985).

At high tide in the Inner Gulf, the direction of the tidal current is northerly, while during the low tide period the direction of the current is southerly. The average velocity of the tidal driven current varies in the range of 1.5-2 knot. The effect of wind and water density on the water circulation in the Inner Gulf is less than the tidal current, with wind-driven current velocity being less than 0.5 knot.

During the transition period of the northeast monsoon to the southwest monsoon in March and April, a southerly wind blows over this region which consequently induces a wind driven current. The surface current flows in the northeast direction towards the eastern coast of the Inner Gulf. In deeper layers, the direction of flow deviates to the right of the wind direction more than at the surface until it moves in the opposite direction. The magnitude of the current decreases with depth, so that the water mass at the surface flowing into the Inner Gulf is larger than in the deeper layer flowing out. The excessive water mass piles up along the northern and eastern coast during strong southerly winds, similar to a storm surge (Neelasri 1981).

2.5.1.6 Temperature distribution

Generally, the sea water temperature of the Inner Gulf of Thailand varies little in both the horizontal and the vertical planes. The northeast and southwest monsoon play an important role in the water temperature distribution. During the northeast monsoon season, sea temperature varies in the range 27-30° C. Surface temperatures increase slightly within the range 28-32° C during the southwest monsoon (Meteorological Department 1987).

2.5.1.7 Salinity distribution

In the Inner Gulf of Thailand, salinity is the driver of water density change. Annual salinity varies between 5 to 33 psu and extreme variation only occurs near rivermouths during the rainy season (wet period from July-December) in the range of 22-32.5 psu, but there is a small fluctuation in the summer (dry period from January - May) within the range 28-32.5 psu.

The effect of freshwater runoff on the salinity of the Inner Gulf is very significant, particularly the effect on surface salinity. In the dry period (January-May), the Inner Gulf is well mixed except in the area of the rivermouths. During the wet period (June- December), surface salinity near rivermouths may drop to 1psu and in some years the large amount of river runoff can affect surface salinities as far as the middle part of the Inner Gulf (Wiriwuttikorn 1996).

2.5.2 Mae Klong Estuary

The Mae Klong River lies in the western part of the inner Gulf of Thailand (13.33° - 14.00° N, 99.50° - 100.09 ° E) (Figure 2.9). The river, which is 138 km long, starts from the confluence of the Khwai Yai and Khwai Noi rivers in Kanchanaburi Province and flows through Ratchaburi and Samut Songkhram Provinces into the Gulf of Thailand, where one of the most important areas of tin production in South East Asia is located (Censi et al. 2007). The Mae Klong river discharges from 9,000 to 16,000 x 10^6 m^3/year into the inner Gulf of Thailand at Samut Songkram (Boonyatumanond et al. 2003). The gradient of the river is about 1:5000 between Kanchanaburi and Tamaka sub-district (in Kanchanaburi Province) and 1: 7250 from

Tamaka to the river mouth. The channel cross section is a wide U-shape with a typical flow velocity of 0.3 to 0.4 m sec $^{-1}$. The river supplies water for irrigation and supports aquaculture industries such as fish ponds and shrimp farms. The Mae Klong tributaries run through agriculture areas such as rice fields, vegetable farms, fruit orchards, and also chemical industries, paper factories and storage battery factories (Peebua et al.2006).

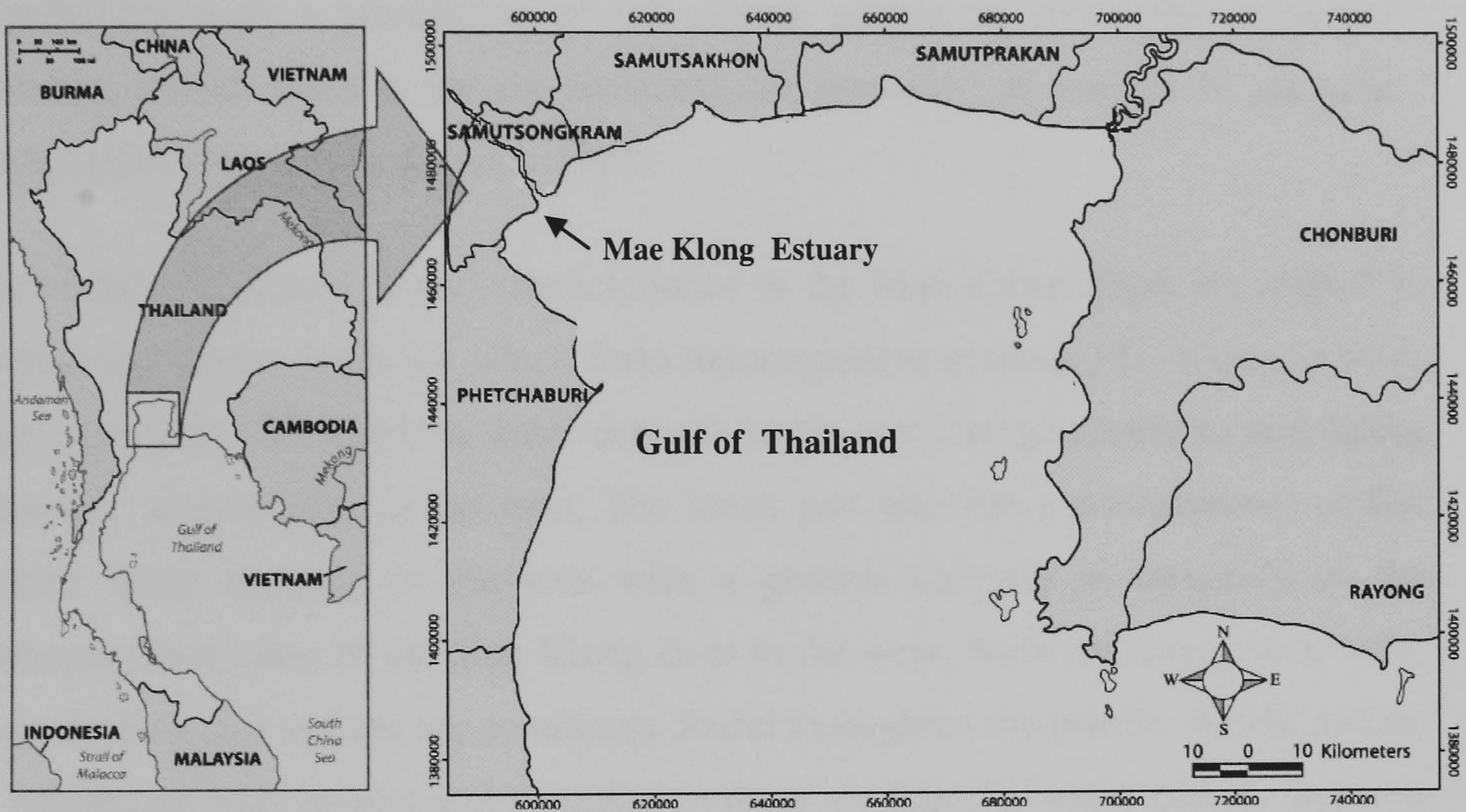


Figure 2.9 Location of Mae Klong Estuary, Thailand (modified from Marine and Coastal Resources Department 2006, http://www.dmcr.go.th/uppercenter/page08.htm).

The climate of this area can be divided into two seasons, the wet season from May to October, and the dry season from November to April. The river receives heavy freshwater loading during November to January each year due to the release of water from the Vachiralongkorn Dam upstream. There are two peaks of heavy rainfall in the area of the Mae Klong River, in May and October. Evaporation at Kanchanaburi is higher than rainfall, except between August and November (Thailand Environment Foundation 1997).

The river discharge is 35-100 m^3 sec^{-1} from January to May, discharge increasing to 150-950 m^3 sec^{-1} from the period June to December, with the peak discharge in

August or September (Thailand Environment Foundation 1997). The tidal range at the mouth of the river varies from 1-2 m on neap tides to 2-3 m on spring tides. Tidal instrusion extends 28 km upstream in the dry season, and less than 8 km from the river mouth in the wet season (Hungspreugs et al. 1987). Most recent data (Hungspreugs et al. 1987) indicate that tidal influence may extend 40 km and 69 km upstream from the river mouth for high and low stream flow conditions, respectively. The total catchment area of the river and its tributaries is about 4,200 km^2, covering six provinces with a population of 1.2 million people in 1996. The projected population growth rates in the six provinces are between 0.6 and 0.8 % per year (Thailand Environment Foundation 1997).

Most of the differences in soil characteristics in the Mae Klong basin are related to elevation and parent materials which form toposequences gradually from east to west. In the upper part, flat brackish water deposits in the east change slowly to undulating freshwater alluvial fans in the west. The lower part also has a toposequence of flat brackish water deposits in the east with a gradual increase in elevation of the freshwater flood plain of the Mae Klong river in the west. Soils are clayey in texture, poorly drained and mottles are commonly found throughout the profile. Acidic sulfate soil conditions with jarosite and a mottles horizon are found in many places. Gypsum is formed in the soil profile as the result of the reaction between calcium carbonate-charged water and sulphate materials in soil parent materials. Most of the soils are moderate in fertility and are not suited for upland crops due to the limitations in flooding and restricted drainage. Most of the soils are suited to paddy field rice but are limited by strong acidity which may be reversed by liming and proper soil management practices (Stonsoavapak 1982).

The Mae Klong basin can be divided into two sub-basins. The lower sub-basin, under the influence of sea water intrusion, extends from the Mae Klong River mouth in Samut Songkhram Province to Sirilak Bridge in Ratchaburi Province. This sub-basin is about 45 km in length, with a highly populated area near the coast. Patches of mangrove and broad mudflats occupy the coastline of Samut Songkhram Province, supporting mussel and clam cultivation. The main activities in the coastal area include aquaculture, salt ponds and fisheries, particularly razor clam harvesting. Fish, shellfish and jellyfish are important fishery products in this area. Agriculture and food industries account for only a small proportion of land use. Don Hoi Lot (Figure 2.10), which translates into English as "Razor Clam Mudflat", covers 87,500 ha, and is located in the eastern shoreline of the mouth of the Mae Klong river. It has been designated a Ramsar site, effective from 5 July 2001. This site represents a rare type of natural wetland for Thailand, comprising sandbars at the mouth of the Mae Klong River with a vast area of intertidal mudflat, an extremely productive location for the razor clam Hoi Lot (*Solen regularis*), an economically important mollusc unique in this region. Mangroves are present along the shoreline on the eastern side of the river mouth. In addition to its 10 economically important mollusc species, this site is also important for ecotourism, its local identity and its traditional fisheries, fishing technologies, sea foods and other fishery products. Development projects for this area are a potential threat, and water pollution from upriver industries, urban and agricultural runoff present major problems. The encroachment into the mangroves for aquaculture and tourist infrastructure is also a threat, to the extent that local extinction of *Solen regularis* is feared unless there is more effective management. A management plan has been approved by the National Environment Board but not yet budgeted for (Ramsar Convention Bureau 2001).

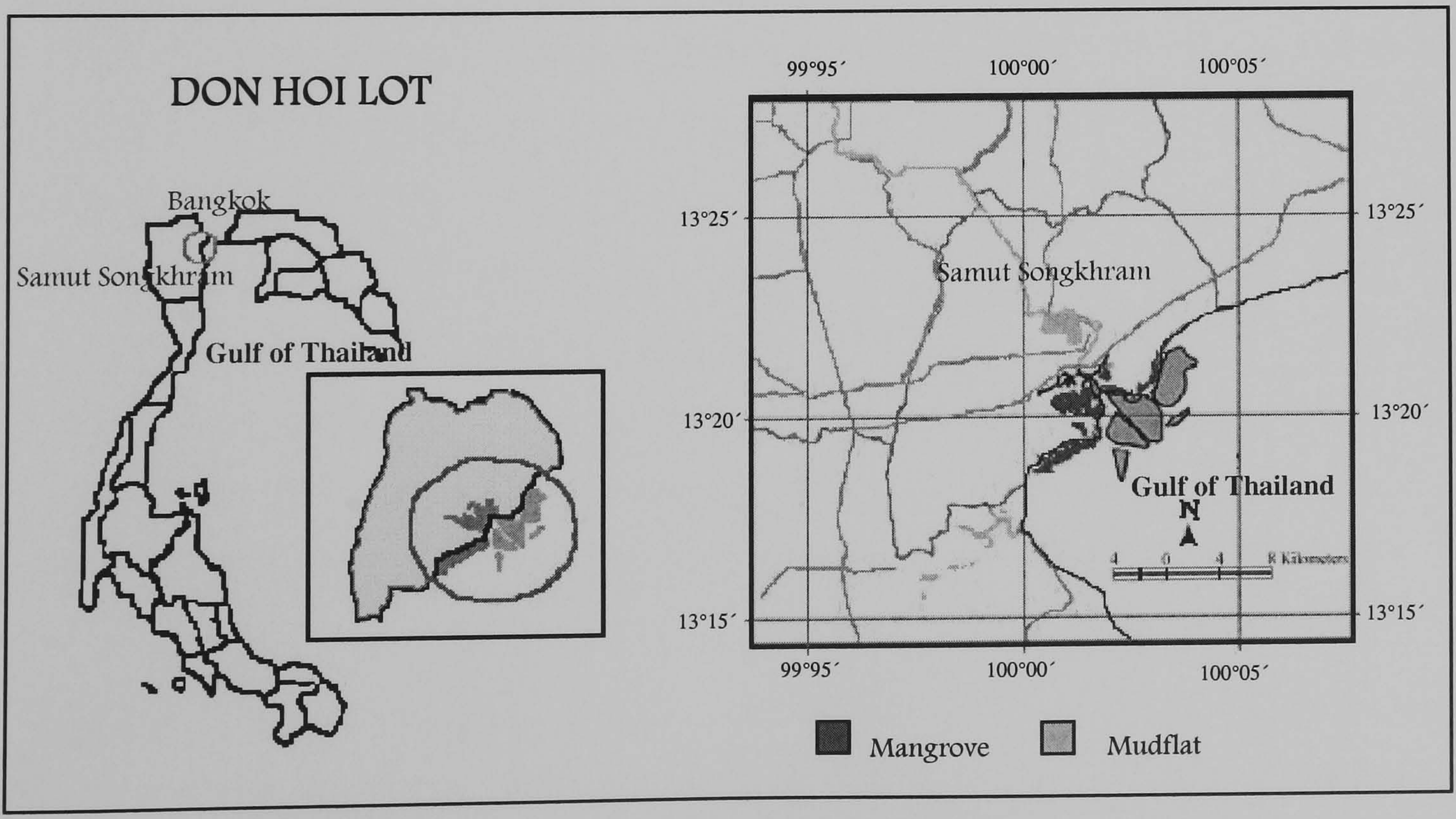


Figure 2.10 Don Hoi Lot (modified from Veeravaitaya 2007).

The upper sub-basin extends 95 km, from Photharam district, Ratchaburi Province to Maung district in Kanchanaburi Province. Most of the land use in this sub-basin consists of pig and duck farming, agriculture, pulp and paper production and sugar refining (Hungspreugs et al. 1987; Thailand Environment Foundation 1997).

CHAPTER 3

Fish assemblages of the Mae Klong Estuary: Species composition, abundance, biomass and diets

3.1 Introduction

Throughout the world there is a general acknowledgement of the ecological and economic importance of forests. Mangroves, being intertidal forests, are no exception. They are one of the most productive of all natural ecosystems in Thailand (Aksornkoae 1980, 1993, 1997) and are widely acknowledged to be important elements in estuarine and coastal ecosystems (Snedaker 1989; Tomlinson 1986; Aksornkoae 1997). Although recent research has necessitated a re-evaluation of their precise role in coastal dynamics (Wolanski and Ridd 1986), these wetlands are known to be highly productive systems that support large densities and high biomass of both fish and invertebrates. With respect to many organisms including commercially valuable finfish, molluscs and shrimp species, these habitats serve as sheltering, breeding and nursery grounds (Monkolprasit 1983; Bell et al.1984; Aksornkoae 1993; Raffaelli and Hawkins 1996; Nagelkerken et al. 2000), providing habitat for early life stages of invertebrates and fish that reside in upstream or downstream habitats as adults (Nagelkerken et al. 2001, Nagelkerken et al. 2002). These vegetated habitat types offer a structurally complex refuge that may reduce predation pressure on small nekton and enhance their growth and survival (Rozas and Minello 1998; Laegdsgaard and Johnson 2001; Minello et al. 2003). In addition, the more turbid water is considered to reduce foraging efficiency of predators (Weis and Weis 2005). Mangroves also benefit neighboring ecosystems (Nagelkerken and Faunce 2007), perhaps best known in this respect as valuable fish habitats for species that are utilized by commercial and subsistence fisheries (Weis and Wies 2005).

Several studies of mangroves associated with fish populations in Thailand provide evidence that mangroves maintain estuarine water quality and play crucial roles in the life cycles of many species of fish (Vathanachai 1979; Monkolprasit 1994; Boonruang and Satapoomin 1997; Janekitkarn et al. 1999; Vidthayanon and

Premcharoen 2001, 2002; Ikejima et al. 2003). This area is often characterized by a high diversity of fish, with 607 species recorded from Thailand (Vidthayanon and Premcharoen 2002), and by high densities of juvenile fishes (Ikejima et al. 2003).

Due to the fact that mangroves have high economic value, they have also been heavily impacted. Anthropogenic influences on these systems have led to a ~35% reduction in mangrove area over the last fifty years (Alongi 2002), with half to three-quarters of mangrove area lost in parts of Souhteast Asia (Field et al. 1998). Mangrove areas are used in many ways that result in elimination or severe degradation. The primary threats to mangrove forests are exploitation for lumber and increasingly, deforestation for agriculture, aquaculture and coastal construction (UNEP 1995; Valiela et al. 2001). Most of these threats, either directly or indirectly influence fish and fisheries of estuarine and diadromous species. Thailand lost more than half of its mangrove forest area between 1961 (372,000 ha) and 1993 (168,000 ha) (Thailand Environment 2000) and large areas of mangrove have been converted to shrimp farms in Chantaburi, Samut Sakhon, Chonburi, Samut Songkhram and Petchburi provinces (Aksornkoae 1980).

Mangrove habitat is often characterized by high densities of juvenile fishes (Robertson and Duke 1987; Ikejima et al. 2003), creating a complex food web (Lugo and Sedaker 1974; Tomlinson 1986). The extraordinary high rates of productivity of mangrove, often exceeding $2tha^{-1}y^{-1}$, supports both terrestrial and marine (both pelagic and benthic) food webs and contributes significant carbon to offshore fisheries (Manson et al. 2005a, b). Epifaunal and infaunal organisms are an abundant, high quality food resource for fishes and crustaceans in mangroves (Sasekumar et al. 1992; Ley et al. 1994). Fish are often at the top of food chains in estuarine systems. Nevertheless, they are a trophically diverse group, encompassing species of different sizes and diverse feeding strategies (Abrantes and Sheaves 2009) Although tropical coastal ecosystems in Southeast Asia are important habitats for fish (Blaber 1997; Chou 1996), there has been relatively few studies on fish ecology and trophic organization of these ecosystems (Chong et al. 1990; Hajimae et al.1999; Poovachiranon and Satapoomin1994; Sasekumar et al. 1994; Thollot et al. 1999;

Layman and Silliman 2002; Hajisamae 2003; Hajisamae et al. 2003, 2004, 2006; Bachok et al. 2004), so that mangrove utilization by fishes is poorly understood (Faunce and Serafy 2006). Analysis of the trophodynamics of an ecosystem involves a description and quantification of its food web, defined here as the macro-description of community feeding interactions that can be used to map the flow of energy, materials and nutrients in an ecosystem (Jepsen and Winemiller 2002). In recent years, several studies have emphasized the importance of trophic interactions in estuaries (Wilson and Sheaves 2001; Hajisamae 2003), and implementation of multispecies approaches to fisheries management will require an improved understanding of the community ecology of fish assemblages (Mbabazi et al. 2004).

Overfishing and habitat decline (deforestation of mangroves) have been seen as the main threats to the status of estuarine fishes and fisheries in Thailand (Monkolprasit 1983; Vidthayanon and Premcharoen 2002). The destruction of mangrove, together with aquatic pollution, also would have implications for pathways of energy flow in the coastal ecosystem, population stocks and production decreasing as trophic linkages become disrupted (Kennish 1994).

The main aim of this chapter is to investigate the fish assemblages and ultimately the trophic demand of these fish in the Mae Klong Mangrove Estuary, in order to facilitate the incorporation of the ecologically important species into ecosystem-based management. I used several quantitative measures of assemblage structure to examine species composition, abundance, biomass and diet of fish in the area and also characterized seasonal changes in the assemblage. The specific objectives were to investigate the fish assemblages present in the 3 main seasons (dry, hot and rainy), to identify the dominant food components in the diet of the main fish species and to explore the trophic structure of the fish species utilizing the habitat. The results from this chapter are used to construct a mass-balance model which will be explored in chapter 4.

3.2 Sampling of the fish fauna

All field studies were conducted at Mae Klong Estuary, intertidal mangrove-fringed located in Samut Songkhram Province, western part of the inner Gulf of Thailand (Chapter 2). The forest is dominated by trees of the genera *Avicennia, Rhizophora* and *Sonneratia.* Six sampling sites (Figure 3.1), covering these different mangrove types, were selected and located by GPS (MAGELLAN model GPS 315).

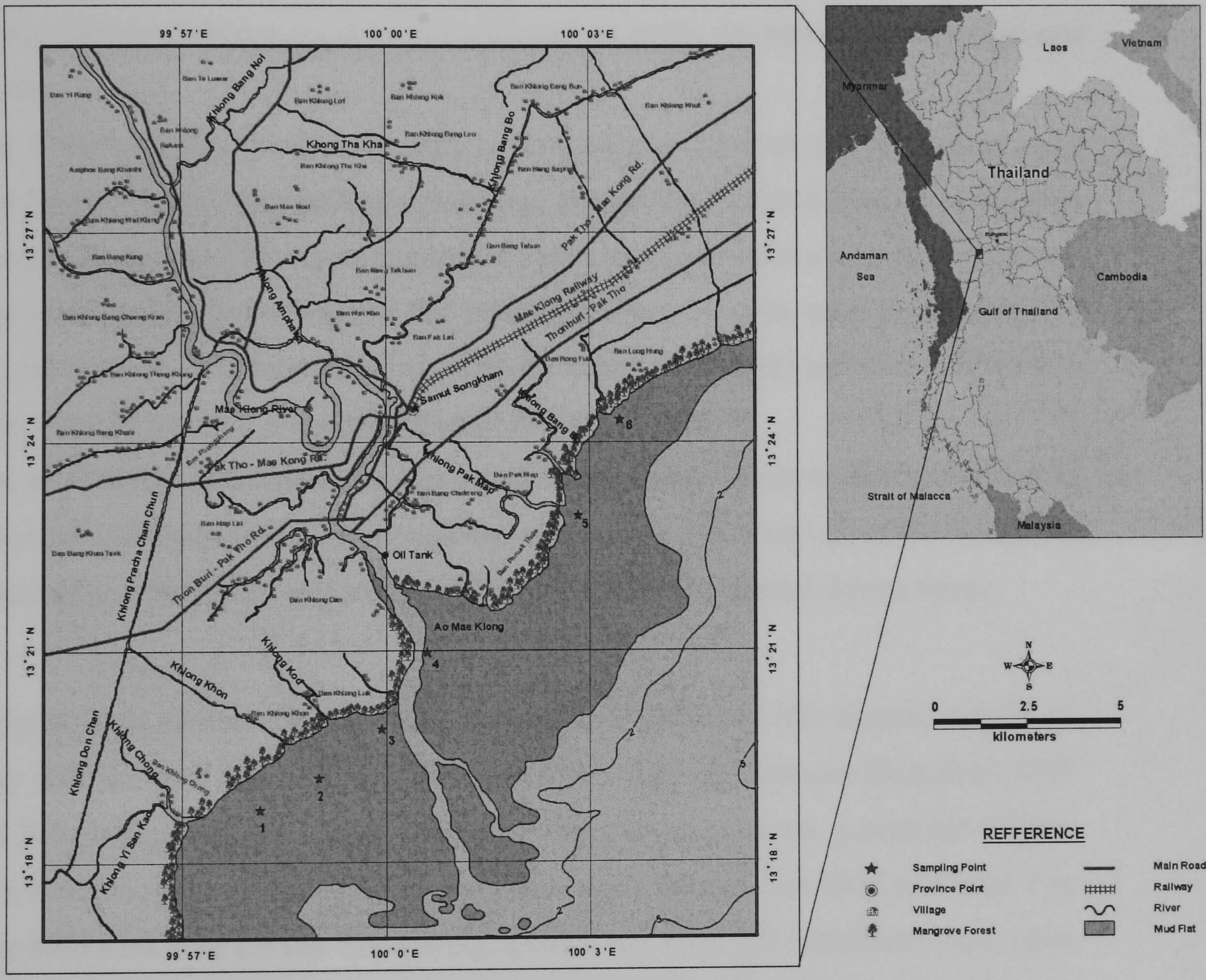


Figure 3.1 Mae Klong Estuary, Inner Gulf of Thailand, showing location of sites (⋆) from which fish samples were collected (modified from Naval Hydrographic Department 1980).

Site 1: **Prag Talay**, dominated by *Sonneratia* (covering around 8 km^2) and a Blood cockle (*Arca granulosa*) farm.

Site 2: **Khlong Khon**, dominated by *Sonneratia* (covering around 6 km^2) and mud flat. It is a public area, with no aquaculture. This area is also dominated by natural populations of the Blood cockle.

Site 3: **Khlong Kod**, is located in Laem Yai Tambon, dominated by *Avicenia marina* and some *Avicennia officinalis* and *Sonneratia*. This area is a sergestid shrimp fishery.

Site 4: **Ao Mae Klong**, dominated by *Rhizophora*, *Nypa* and *Avicennia*. Sediments are sandy (called "Khee Ped Sand" by locals) due to the water current from the Ao Mae Klong. Various fisheries occur in this area such as crabs, sea perch and clams (several species of each).

Site 5: **Khlong Pak Map**, is located in Bangjakreng Tambon and is dominated by *Rhizophora*, *Nypa*, *Avicennia officinalis* and *Sonneratia*. Oriental hard clam *Meretrix meretrix* is the main fishery in this area. There was an abundance of mangrove in this area in the past, but now little of this primary mangrove exists, most being destroyed by urbanization. This area is characterized by muddy and sandy sediment, and there is little water movement. It has an abundant fish population.

Site 6: **Khlong Muenghan**, is located in Bang Kaew Tambon and dominated by *Avicenia* and *Rhizophora*. This area has a mudflat, but away from the beach the sediment is more sandy. It is a bird reserve and has an abandoned shrimp farm.

Sampling was carried out seasonally: December to February (dry season), March to May (hot season), and June to November (rainy season), between December 2005 and November 2006. On 2 or 3 consecutive days in each season, a push net (Figure 3.2), 8 m long with bamboos 10 m long, was used. The net was 20 m wide and 2 m deep with a mesh of 2.5 cm. The towing speed was 1 knot. All collections were made in both day and night at high tide, so that the feeding habits of fish utilizing littoral habitats could be properly assessed (Hajisamae et al.2003). The distance towed was generally 1 km, and fish abundance was standardized to a 1 km tow. Two replicate 15 min tows were carried out on each sampling occasion. On average, water depths were about 1m.

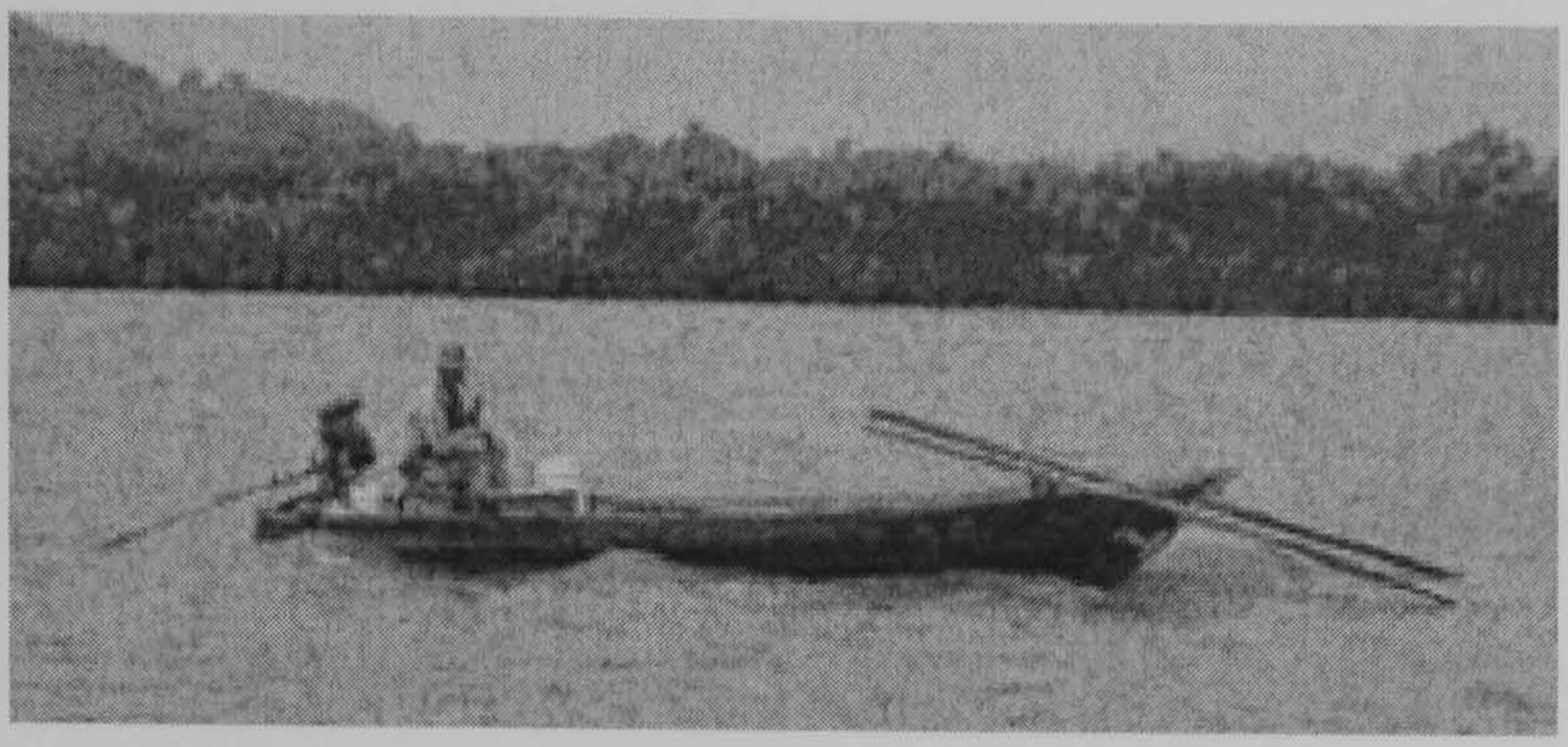


Figure 3.2 A motorized push net on the coast of the Gulf of Thailand (FAO 2008).

Depth, temperature, dissolved oxygen, salinity, and pH were recorded for each haul. Depth was measured with a weighted tape measure. Water temperature and dissolved oxygen concentration were measured with an Oakton model DO 100 meter, salinity with a Corning model salt-70, and pH with a portable pH meter. All variables were recorded at a depth of approximately 30 cm below the surface. The distance between mangrove fringe and each haul was estimated by eye, and the time of day was recorded.

Fish caught were preserved immediately in 10% buffered formaldehyde before being transferred to the laboratory where they were placed in buffered alcohol (70%), identified to species level according to the FAO Species Identification Guide for Fishery Purposes (2001), and assigned to an estuarine association according to Whitefield (1998) and Vidthayanon and Premcharoen (2002). All specimens have been lodged with the Zoology Section Laboratory, Faculty of Liberal Arts and Science, Kasetsart University, Thailand.

3.3 Stomach content analysis

Diets were derived from an analysis of the stomach contents of fishes covering a size range corresponding to the adult stage. Samples for diet analysis were taken for each species haphazardly.

Fish were identified, their total length (TL) measured to the nearest 1 mm and weighed (wet weight) to the nearest 0.1 g. Fish were dissected and the stomach (from the posterior of the oesophagus to the pylorus) was removed, weighed and preserved in 70% buffered alcohol for later identification of prey items. Gut contents were placed in a petri dish and examined under a stereomicroscope and a compound microscope (in case of phytoplankton). All items were identified to the lowest possible taxonomic level. Abundance of food items in stomach contents were estimated as volume rather the biomass because of the uniformly small size of prey species which made weighing impractical and unreliable on most occasions. The percentage volume of major gut items was estimated by using the points method of Hynes (1950). In this method, the contents of each stomach sample are taken as unity and the items expressed as a percentage of the total volume by visual inspection on a 4-point scale. The points, and the percentages they represent are as follows: 4 (75-100%), 3 (50-75%), 2 (25-50%) and 1 (up to 25%). Points for each food item were re-scaled to give the percentage composition of different food items in the diet across all individuals of that species. Empty stomachs or stomachs with almost fully digested contents were excluded. Food items unable to be identified and digested items were categorized as “unidentified” and “digested”, respectively. Due to the tendency for fish species inhabiting mangroves to exhibit patterns of ontogenetic dietary shifts (Cocheret de la Moriniere et al. 2003), diets were categorized on the basis of gut analysis of both juveniles and adults.

In the major dietary analyses which follow, food items have been grouped into 8 major categories: 1) nekton, 2) sergestid shrimps, 3) shrimps, 4) crabs, 5) benthic invertebrates, 6) zooplankton, 7) phytoplankton and other plant tissue, and 8) detritus. Fish were further assigned to residence status (permanent resident, partial resident, tidal visitor, seasonal visitor and rarely occur) and economical status (economic significance, popular aquarium fish and threatened species) according to Mongkolprasit (1983) and Vidthayanon and Premcharoen (2002).

3.4 Variation in fish assemblages

The degree of similarity in fish assemblages between sites, time of day and seasons was explored by classification and ordination using the statistical package Plymouth Routines in Multivariate Ecological Research, PRIMER Version 5.2.9 (Clarke and

Gorley 2001). Data were square-root transformed to downweight the influence of rare and extremely abundant species. Classification (cluster analysis) was performed using the Bray-Curtis coefficient of similarity by weighted clustering. Ordination was performed using non-metric multidimensional scaling (nMDS) on the Bray-Curtis similarity matrix. The extent to which ordination plots displayed the relationships between samples (i.e. goodness of fit) is determined by a stress coefficient, a value of <0.1 indicating a good representation of the data with little risk of misleading interpretation (Clarke and Warwick 1994). Analysis of similarities (ANOSIM) was used to determine whether fish assemblages separated a priori into day or night, station or season differed statistically.

3.5 Results

3.5.1 Water quality characteristics

Dissolved oxygen, pH, temperature and salinity were significant differences among seasons ($P<0.0001$, Table 3.2). Water temperature varied from 24.6 to 31.8 °C, with the highest values recorded in the hot season and lowest values in the dry season (Table 3.1). Dissolved oxygen concentrations and pH differed among the seasons, the highest values of dissolved oxygen being in the dry season and lowest in the rainy season, while the highest values of pH were in the rainy season and lowest in the hot season. Salinity was slightly lower in the dry season, unexpectedly. Water depth averaged 1.20 m, 1.08 m and 1.35 m in dry, hot and rainy seasons, respectively.

Table 3.1 Mean (±SE) environmental characteristics within Mae Klong Estuary at each season (averaged across all six sampling sites).

Parameters	Dry season	Hot season	Rainy season
DO (mg/l)	7.2 (0.2)	5.4 (0.05)	5.2 (0.25)
pH	8.0 (0.05)	7.0 (0.05)	8.4 (0.05)
T (°C)	24.6 (3.65)	31.8 (3.40)	28.0 (0.55)
Salinity (PSU)	0.8 (0.00)	0.9 (0.00)	0.9 (0.05)
Depth (m)	1.20 (0.00)	1.08 (0.00)	1.35 (0.00)

Table 3.2 Results of repeated–measures analysis of variance to test for significant differences in environmental variables among seasons.

Variables	*df*	*MS*	*F*	*P*
Dissolved oxygen				
Between group	2	15.030	20.146	<0.0001
Within group	31	0.746		
pH				
Between group	2	6.427	54.626	<0.0001
Within group	31	0.117		
Temperature				
Between group	2	148.711	14.704	<0.0001
Within group	31	10.113		
Salinity				
Between group	2	0.055	4.414	0.020
Within group	31	0.012		

3.5.2 Species composition, abundance, seasonal distribution and residence status

A total of 7,664 fish from 63 species and representing 25 families were caught in the Mae Klong Estuary over the period of the study (Table 3.3 and 3.4). A total of 4,505 fish (58.78 % of the total catch), representing 31 species from 17 families are considered to be of economic importance (Table 3.4). 61 species were osteichthyes and 2 were chondrichthyes. Of those 25 families, 84 % were resident and commonly found in the estuary in all seasons (Table 3.3). The family Clupeidae was by far the most speciose (9 species), followed by Gobiidae (6 species), Ariidae (5 species) and Sciaenidae (5 species), with 14 families represented by just one species. The total number of species (Figure 3.3a) was highest in the hot season (50 species), with 38 species recorded in dry and rainy seasons.

Of all the species caught (Table 3.5), *Ambassis gymnocephalus* numerically dominated the fish community (18.45%, 88.93 ind/km^2), followed by *Chelon tade* (11.82%, 56.98 ind/km^2) and *Arius macronotacanthus* (11.06%, 53.33 Ind/km^2). Many rare species (30 species in total) were recorded, with abundances of less than 1 ind/km^2. The ten most abundant species (Figure 3.4) accounted for 83% of the total number of individuals collected. Of the 63 species present in the area, 25 species were represented by less than 10 individuals. The total number of individuals was highest in the rainy season (Figure 3.4b), comprising 40.88 % of the total catch.

11 species were found in all seasons and at both day and night (Table 3.4). 20 species only occured in one season, usually with numbers of less than 5 individuals, except for the more numerous *Setipinna taty* and *Sadinella lemuru* which were found only in the rainy season; *Herklotsichthys dispilonotus* was recorded in the hot season and *Acentrogobius caninus* in the dry season.

Table 3.3 Fish species recorded in the Mae Klong Estuary by family and residence status: **R,** permanent resident; **PR,** partial resident; **Vt,** tidal visitor; **Vs,** seasonal visitor; **Oc,** rarely occurs.

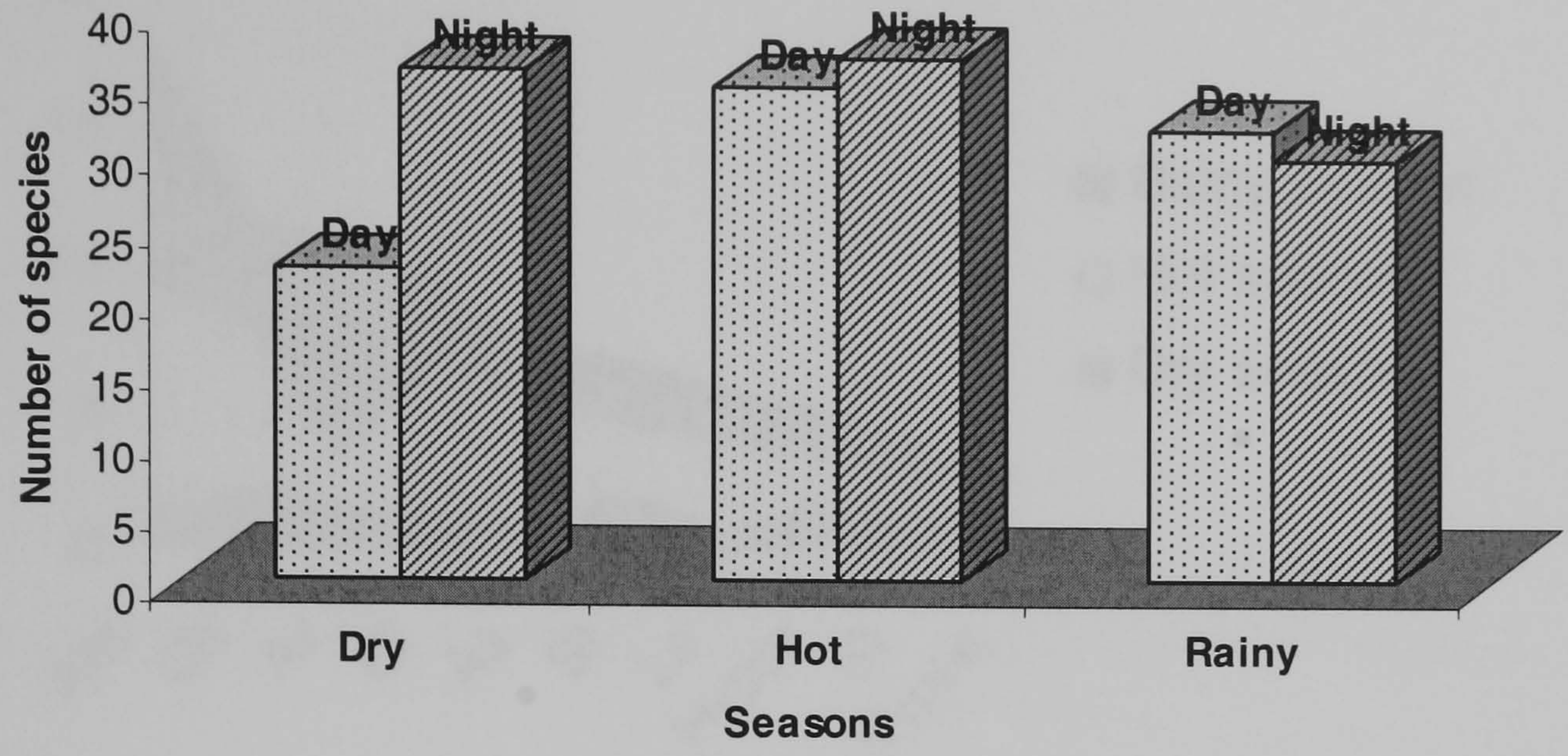
Family	Species	Commom name
Dasyatidae^{R,PR}	*Dasyatis fluviorum* Ogilby, 1908 *Himantura imbricata* (Bloch and Schneider, 1801)	Estuary stingray Scaly whipray
Elopidae^{PR}	*Elops machnata* (Forsskål, 1775)	Tenpounder
Ophichthidae^{R}	*Pisodonophis boro* (Hamilton, 1822)	Rice-paddy snake eel
Engraulidae^{R,PR,Vs,Oc}	*Setipinna taty* (Valenciennes, 1848) *Stolephorus commersonii* Lacepède,1803 *Thryssa hamiltoni* (Gray, 1835) *Thryssa setirostris* (Broussonet, 1782)	Scaly hairfin anchovy Commerson's anchovy Hamilton's thryssa Longjaw thryssa
Clupeidae^{R,PR,Vs}	*Escualosa elongata* Wongratana, 1983 *Escualosa thoracata* (Valenciennes, 1847) *Anodontostoma chacunda* (Hamilton, 1822) *Herklotsichthys dlispilonotus* (Bleeker, 1852) *Hilsa kelee* (Cuvier, 1829) *Sardinella albella* (Valenciennes, 1847) *Sardinella fimbriata* (Valenciennes, 1847) *Sardinella gibbosa* (Bleeker, 1849) *Sardinella lemuru* Bleeker, 1853	Slender white sardine White sardine Chacunda gizzard shad Black saddle herring Kelee shad White sardinella Fringescale sardinella Glodstripe sardinella Bali sardinella
Ariidae^{R}	*Arius macronotacanthus* Bleeker, 1846 *Arius sagor* (Hamilton, 1822) *Cryptarius truncatus* (Valenciennes, 1840) *Ketengus typus* Bleeker, 1847 *Osteogeneiosus militanis* (Linnaeus, 1758)	Largespined catfish Sagor catfish Spoonsnouted catfish Bigmouth sea catfish Soldier catfish
Plotosidae^{R,PR}	*Plotosus canius* Hamilton, 1822	Eel catfish
Mugilidae^{R}	*Chelon tade* (Valenciennes, 1836) *Liza subviridis* (Valenciennes, 1836) *Moolgarda seheli* (Forsskål, 1775)	Tade mullet Greenback mullet Bluespot mullet
Atherinidae^{PR,Vt}	*Hypoatherina valenciennei* (Bleeker, 1853)	Sumatran silverside

Belonidae^{R,PR,Vt}	*Strongylura strongylura* (van Hasselt, 1823)	Spottail needlefish
Hemirhamphidae^{R,PR}	*Hyporhariphus (Hyporhamphus) limbatus* (Valenciennes, 1846) *Rhynchorhamphus naga* Collette, 1976	Congaturi halfbeak Naga halfbeak
Ambassidae^{R}	*Ambassis gymnocephalus* (Lacepède 1802) *Ambassis nalua*(Hamilton, 1822)	Perchlets Scalloped perchlet
Sillaginidae^{R}	*Sillago sihama* (Forsskål, 1775)	Silver sillago
Carangidae^{R,PR}	*Alepes djedaba* (Forsskål, 1775) *Scomberoides commersonnianus* Lacepède, 1801 *Scomberoides tol* (Cuvier, 1832) *Selaroides leptolepis* (Cuvier, 1833)	Shrimp scad Talang queenfish Needlescaled queenfish Yellowstripe scad
Leiognathidae^{R,PR}	*Leiognathus decorus* (de Vis, 1884) *Secutor insidiator* (Bloch, 1787) *Secutor ruconius* (Hamilton-Buchanan, 1822)	Yellowfinned ponyfish Pugnose ponyfish Deep pugnose ponyfish
Gerreidae^{R,PR}	*Gerres erythrourus* (Bloch, 1971)	Deepbody silverbiddy
Polynemidae^{R}	*Eleuthronema tetradactylum* (Shaw, 1804)	Fourfinger threadfin
Sciaenidae^{R}	*Aspericorvina jubata* (Bleeker, 1855) *Chrysochir aureus (*Richardson, 1846*)* *Dendrophysa russelli* (Cuvier, 1830) *Nibea albiflora* (Richardson, 1846) *Panna microdon* (Bleeker, 1849)	Prickly croaker Reeve's croaker Goatee croaker White flower croaker Panna croaker
Drepanidae^{PR}	*Drepane punctata* (Linnaeus, 1758)	Spotted sicklefish
Teraponidae^{PR}	*Terapon theraps* (Cuvier, 1829)	Largescaled terapon
Eleotridae^{R}	*Butis butis* Hamilton, 1822 *Ophiocara porocephala* (Valenciennes, 1837)	Sleepers Northern mud gudgeon
Gobiidae^{R,PR}	*Acentrogobius caninus* (Valenciennes,1837) *Acentrogobius chlorostigmatoides* (Bleeker, 1849) *Acentrogobius cyanomos* (Bleeker, 1849) *Bathygobius fuscus* (Rüppell, 1830) *Favonigobius aliciae* (Herre, 1936) *Pseudapocryptes lanceolatus* (Bloch & Schneider, 1801)	Tropical sand goby Greenspot goby Dusky frillgoby Pointed-tailed goby
Scatophagidae^{R}	*Scatophagus argus* (Bloch, 1788)	Spotted scat
Uranoscopidae^{Vs}	*Uranoscopus bicinctus* Temminck & Schlegel, 1843	Marbled stargazer
Triacanthidae^{PR}	*Triacanthus biaculeatus* (Bloch, 1786)	Short- nosed tripodfish
Cynoglossidae^{R,PR}	*Cynoglossus lingua* Hamilton-Buchanan, 1822 *Cynoglossus puncticeps* (Richardson, 1846)	Long tonguesole Speckled tonguesole
Tetraodontidae^{R,PR}	*Lagocephalus lunaris* (Bloch & Schneider, 1801)	Green rough-backed puffer

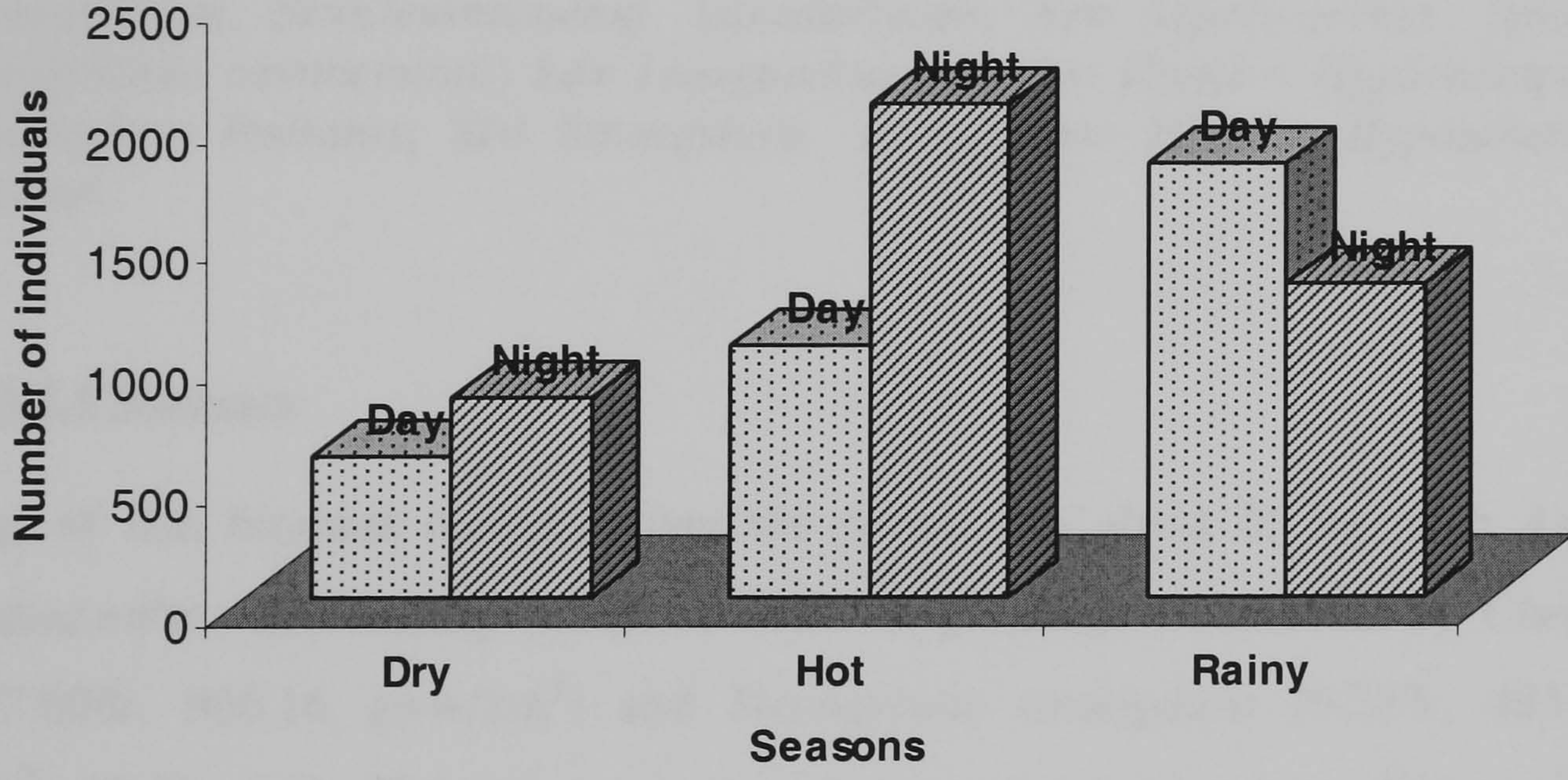
Table 3.4 Total abundance of each fish species (listed in phylogenetic order) by season and time of day (D=Day, N=Night) for 6 stations (total area fished= 15.9 km) in the Mae Klong Estuary between December 2005 and August 2006. * Denotes species of economic significance, **A** popular aquarium fish, **T** threatened species.

Species	**Dry season**		**Hot season**		**Rainy season**	
	D	**N**	**D**	**N**	**D**	**N**
Dasyatidae						
Dasyatis fluviorum *	1	4	-	4	1	8
*Himantura imbricata**	1	2	3	-	-	-
Elopidae						
Elops machnata **T**	-	-	1	-	-	-
Ophichthidae						
*Pisodonophis boro**	-	2	-	1	-	-
Engraulidae						
Setipinna taty **T**	-	-	-	-	-	48
*Stolephorus commersonii**	132	25	164	33	168	4
Thryssa hamiltoni	-	-	1	-	-	20
Thryssa setirostris	-	1	1	-	-	-
Clupeidae						
Escualosa elongata	1	-	10	31	31	21
*Escualosa thoracata**	16	10	32	19	4	-
*Anodontostoma chacunda** **T**	-	-	1	-	-	-
*Herklotsichthys dispilonotus**	-	-	1	13	-	-
*Hilsa kelee**	-	-	-	2	-	-
Sardinella albella	2	2	9	-	6	1
*Sardinella fimbriata**	-	1	-	-	6	7
*Sardinella gibbosa**	-	-	1	1	-	-
Sardinella lemuru	-	-	-	-	10	23
Ariidae						
*Arius macronotacanthus**	2	103	37	113	475	118
*Arius sagor**	-	-	9	23	69	-
*Cryptarius truncatus**	-	-	-	-	2	-
Ketengus typus **T**	-	-	-	5	18	9
*Osteogeneiosus militaris**	-	1	-	-	-	-
Plotosidae						
*Plotosus canius**	2	13	5	2	1	57
Mugilidae						
*Chelon tade**	60	94	179	298	201	74
*Liza subviridis**	27	72	10	6	1	16
*Moolgarda seheli**	-	-	1	-	-	-
Atherinidae						
Hypoatherina valenciennei	93	28	3	66	41	4
Belonidae						
*Strongylura strongylura**	31	52	13	83	31	105
Hemirhamphidae						
*Hyporhariphus (Hyporhamphus) limbatus**	129	17	74	37	21	115
Rhynchorhamphus naga	11	26	12	3	40	-

Ambassidae						
Ambassis gymnocephalus	42	45	157	474	292	404
Ambassis nalua	-	-	-	-	-	1
Sillaginidae						
*Sillago sihama**	-	34	1	16	20	4
Carangidae						
Alepes djedaba	-	-	9	-	-	1
Scomberoides commersonnianus	-	-	1	-	-	-
Scomberoides tol	-	-	1	-	-	-
*Selaroides leptolepis**	-	-	-	1	-	-
Leiognathidae						
Leiognathus decorus	1	51	42	208	96	44
Secutor insidiator	1	-	2	26	-	-
Secutor ruconius	1	1	15	-	-	-
Gerreidae						
Gerres erythrourus	-	5	-	15	5	3
Polynemidae						
*Eleuthronema tetradactylum**	24	71	241	175	115	106
Sciaenidae						
Aspericorvina jubata	-	69	7	326	80	68
*Chrysochir aureus**	-	-	-	-	-	5
*Dendrophysa russelli**	-	38	-	45	33	-
*Nibea albiflora**	-	-	-	4	1	-
Panna microdon	-	6	-	-	15	6
Drepanidae						
Drepane punctata	-	1	-	1	1	1
Teraponidae						
*Terapon theraps**	1	3	1	10	3	3
Eleotridae						
Butit butis A	2	27	-	-	1	4
Ophiocara porocephala	-	13	-	1	2	-
Gobiidae						
*Acentrogobius caninus**	4	12	-	-	-	-
Acentrogobius chlorostigmatoides	-	-	-	1	-	-
Acentrogobius cyanomos	-	3	-	-	-	-
Bathygobius fuscus	-	-	1	-	-	-
Favonigobius aliciae	-	3	-	-	-	-
Pseudapocryptes lanceolatus	-	1	-	-	-	-
Scatophagidae						
*Scatophagus argus**A	-	1	5	4	28	32
Uranoscopidae						
Uranoscopus bicinctus	-	-	-	1	-	-
Triacanthidae						
Triacanthus biaculeatus	-	-	1	-	-	-
Cynoglossidae						
*Cynoglossus lingua**	-	4	-	1	-	-
*Cynoglossus puncticeps**	-	-	-	4	9	-
Tetraodontidae						
Lagocephalus lunaris	-	-	-	2	-	-



(a)



(b)

Figure 3.3 Total number of species (a), and individuals (b) recorded at different times of day and in different seasons.

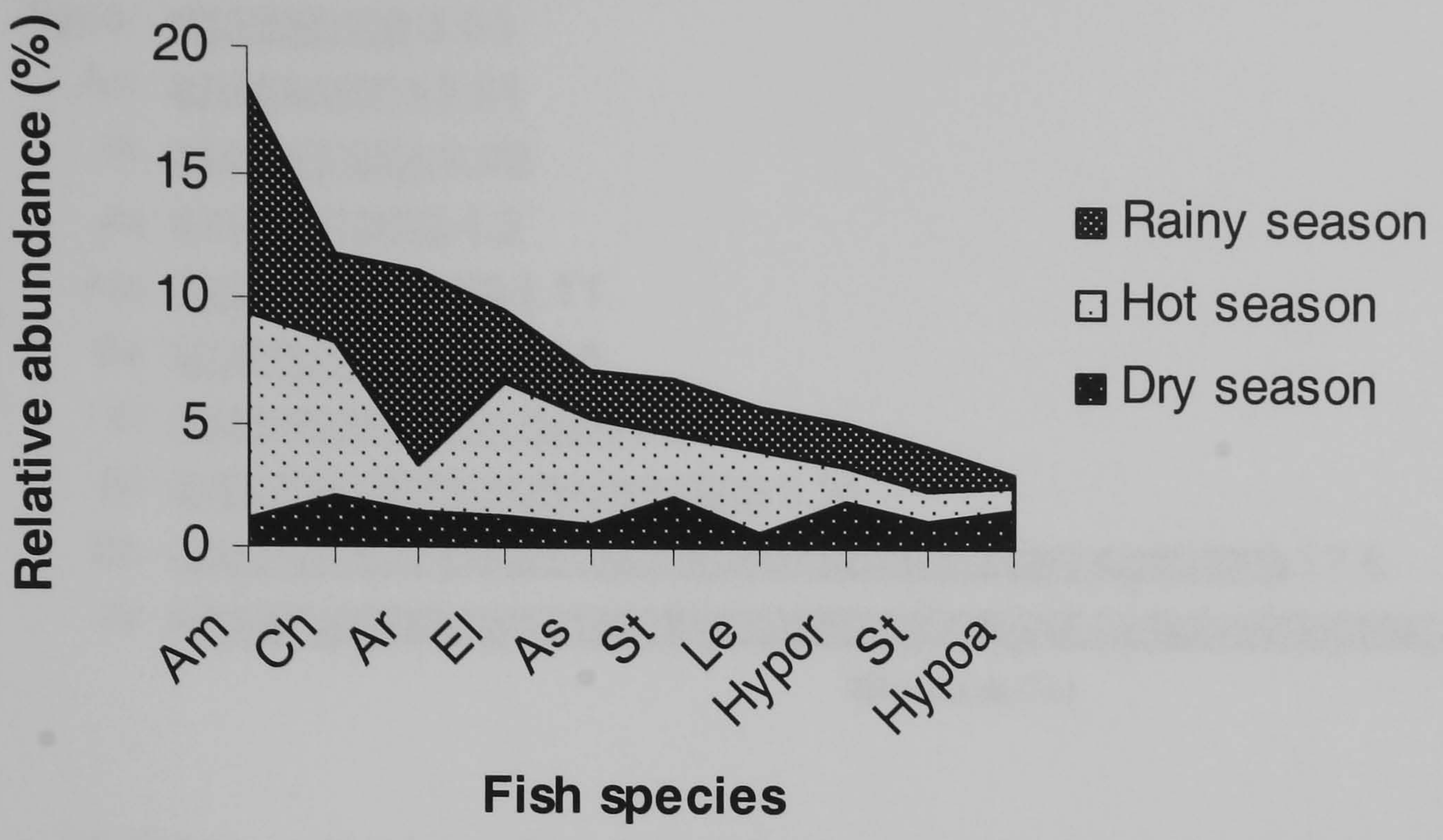


Figure 3.4 Relative abundance (percentage of number caught) of ten most abundant species in each season. ***Am***= *Ambassis gymnocephalus;* ***Ch***= *Chelon tade;* ***Ar***= *Arius macronotacanthus*; ***El***=*Eleuthronema tetradactylum;* ***As***= *Aspericorvina jubata*; ***St***= *Stolephorus commersonii*; ***Le***= *Leiognathus decorus;* ***Hypor***= *Hyporhamphus (Hyporhamphus) limbatus;* ***St***= *Strongylura strongylura*; ***Hypoa***= *Hypoatherina valenciennei.*

3.5.3 Biomass

The total of fish biomass recorded over the period was about 85 kg, with *Arius macronotacanthus* dominating (22.21%, 1181.12 gww/km^2), followed by *Chelon tade* (17.60%, 936.16 gww/km^2) and *Strongylura strongylura* (9.28%, 493.74 gww/km^2) (Table 3.5). The ten most abundant species by biomass (Figure 3.4) accounted for 82.24% of the total fish biomass. Of the ten most numerically abundant species, seven were also dominant in biomass.

Maximum biomass was recorded during the rainy season (50.95% of the total biomass) with the hot season (28.93%) and dry season (20.12%) biomasses being relatively smaller (Table 3.6).

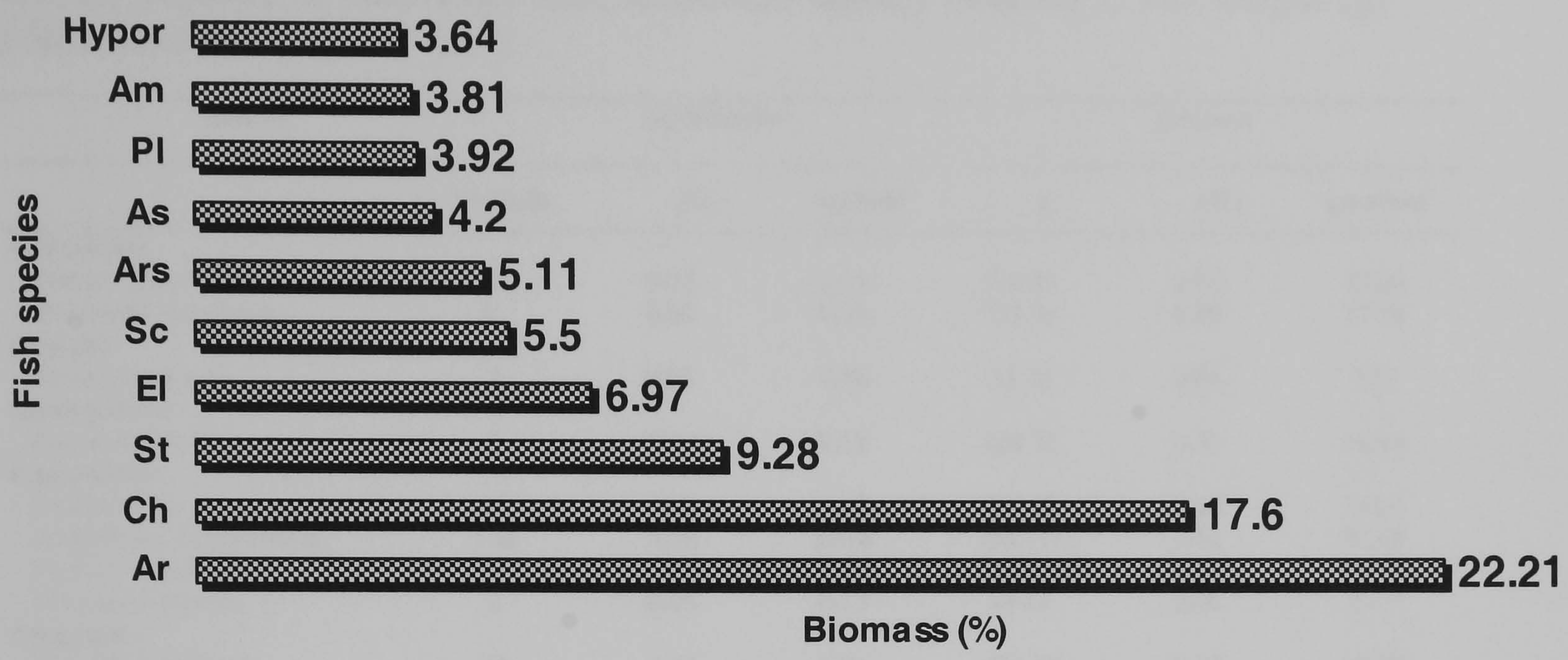


Figure 3.5 Relative abundance (percentage biomass) of the ten most fish biomass. **Ar=**; *Arius macronotacanthus* **Ch** = *Chelon tade;* **St=** *Strongylura strongylura*; **El=***Eleuthronema tetradactylum;* **Sc=** *Scatophagus argus;* **Ars=** *Arius sagor;* **As=** *Aspericorvina jubata* ; **Pl=***Plotosus canius;* **Am=** *Ambassis gymnocephalus;* **Hypor=** *Hyporhamphus (Hyporhamphus) limbatus.*

Table 3.5 Number of individual fish, numerical density (ind/km^2), wet weight (g) and biomass density (gww/ km^2).

Species	Abundance			Biomass		
	No./Inds.	(%)	ind/km^2	g	(%)	gww/km^2
Dasyatidae						
Dasyatis fluviorum	18	0.23	1.13	716.52	0.85	45.06
Himantura imbricata	6	0.08	0.38	753.39	0.89	47.38
Elopidae						
Elops machnata	1	0.01	0.06	35.28	0.04	2.22
Ophichthidae						
Pisodonophis boro	3	0.04	0.19	269.73	0.32	16.96
Engraulidae						
Setipinna taty	48	0.63	3.02	382.03	0.45	24.03
Stolephorus commersonii	526	6.86	33.08	1553.16	1.84	97.68
Thryssa hamiltoni	21	0.27	1.32	165.43	0.20	10.40
Thryssa setirostris	2	0.03	0.13	44.11	0.05	2.77
Clupeidae						
Escualosa elongata	94	1.23	5.91	271.28	0.32	17.06
Escualosa thoracata	81	1.06	5.09	188.73	0.22	11.87
Anodontostoma chacunda	1	0.01	0.06	9.0	0.01	0.57
Herklotsichthys dispilonotus	14	0.18	0.88	15.21	0.02	0.96
Hilsa kelee	2	0.03	0.13	6.33	0.01	0.40
Sardinella albella	14	0.18	0.88	72.53	0.09	4.56
Sardinella fimbriata	14	0.18	0.88	142.67	0.17	8.97
Sardinella gibbosa	2	0.03	0.13	6.23	0.01	0.39
Sardinella lemuru	33	0.43	2.08	338.48	0.40	21.29
Ariidae						
Arius macronotacanthus	848	11.06	53.33	18779.79	22.21	1181.12
Arius sagor	101	1.32	6.35	4320.92	5.11	271.76
Cryptarius truncatus	2	0.03	0.13	13.81	0.02	0.87
Ketengus typus	32	0.42	2.01	372.93	0.44	23.45
Osteogeneiosus militaris	1	0.01	0.06	57.1	0.07	3.59
Plotosidae						
Plotosus canius	80	1.04	5.03	3313.65	3.92	208.40
Mugilidae						
Chelon tade	906	11.82	56.98	14884.89	17.60	936.16
Liza subviridis	132	1.72	8.30	2803.65	3.32	176.33
Moolgarda seheli	1	0.01	0.06	43.66	0.05	2.75
Atherinidae						
Hypoatherina valenciennei	235	3.07	14.78	564.85	0.67	35.53
Belonidae						
Strongylura strongylura	315	4.11	19.82	7850.52	9.28	493.74
Hemirhamphidae						
Hyporhamphus (Hyporhamphus) limbatus	393	5.13	24.72	3077	3.64	193.52
Rhynchorhamphus naga	92	1.20	5.79	343.95	0.41	21.63
Ambassidae						
Ambassis gymnocephalus	1414	18.45	88.93	3224.64	3.81	202.81
Ambassis nalua	1	0.01	0.06	10.97	0.01	0.69
Sillaginidae						
Sillago sihama	75	0.98	4.72	492.84	0.58	31.0
Carangidae						
Alepes djedaba	10	0.13	0.63	32.40	0.04	2.04
Scomberoides commersonnianus	1	0.01	0.06	5.60	0.01	0.35
Scomberoides tol	1	0.01	0.06	3.10	0.004	0.19
Selaroides leptolepis	1	0.01	0.06	0.54	0.001	0.03
Leiognathidae						
Leiognathus decorus	442	5.77	27.80	1799.89	2.13	112.51
Secutor insidiator	29	0.38	1.82	7.70	0.01	0.48
Secutor ruconius	17	0.22	1.07	21.55	0.03	1.36
Gerreidae						
Gerres erythrourus	28	0.37	1.76	54.66	0.06	3.44
Polynemidae						
Eleuthronema tetradactylum	732	9.55	46.04	5894.77	6.97	370.74
Sciaenidae						
Aspericorvina jubata	550	7.18	34.59	3555.47	4.20	223.61
Chrysochir aureus	5	0.07	0.31	47.94	0.06	3.02

Dendrophysa russelli	116	1.51	7.30	887.19	1.05	55.80
Nibea albiflora	5	0.07	0.31	282.11	0.33	17.74
Panna microdon	27	0.35	1.70	481.72	0.57	30.30
Drepanidae						
Drepane punctata	4	0.05	0.25	384.31	0.45	24.17
Teraponidae						
Terapon theraps	21	0.27	1.32	331.68	0.39	20.86
Eleotridae						
Butit butis	34	0.44	2.14	537.48	0.64	33.80
Ophiocara porocephala	16	0.21	1.01	76.56	0.09	4.82
Gobiidae						
Acentrogobius caninus	16	0.21	1.01	92.95	0.11	5.85
Acentrogobius chlorostigmatoides	1	0.01	0.06	4.46	0.01	0.28
Aulopareia cyanomos	3	0.04	0.19	7.52	0.01	0.47
Bathygobius fuscus	1	0.01	0.06	1.26	0.001	0.08
Favonigobius aliciae	3	0.04	0.19	52.17	0.06	3.28
Pseudapocryptes lanceolatus	1	0.01	0.06	20.24	0.02	1.27
Scatophagidae						
Scatophagus argus	70	0.91	4.40	4651.41	5.50	292.54
Uranoscopidae						
Uranoscopus bicinctus	1	0.01	0.06	33.24	0.04	2.09
Triacanthidae						
Triacanthus biaculeatus	1	0.01	0.06	12.90	0.02	0.81
Cynoglossidae						
Cynoglossus lingua	5	0.07	0.31	43.98	0.05	2.77
Cynoglossus puncticeps	13	0.17	0.82	72.9	0.09	4.58
Tetraodontidae						
Lagocephalus lunaris	2	0.03	0.13	43.34	0.05	2.73

Table 3.6 Total biomass of fish in each season.

Family	Species	Biomass (g)		
		Dry	**Hot**	**Rainy**
Dasyatidae	*Dasyatis fluviorum* Ogilby, 1908	150.83	282.72	282.97
	Himantura imbricata (Bloch and Schneider, 1801)	622.7	130.69	-
Elopidae	*Elops machnata* (Forsskal, 1775)	-	35.28	-
Ophichthidae	*Pisodonophis boro* (Hamilton, 1822)	132.18	137.55	-
Engraulidae	*Setipinna taty* (Valenciennes, 1848)	-	-	382.03
	Stolephorus commersonii Lacepede,1803	487.73	642.27	423.16
	Thryssa hamiltoni (Gray, 1835)	-	29.02	136.41
	Thryssa setirostris (Broussonet, 1782)	11.78	32.33	-
Clupeidae	*Escualosa elongata* Wongratana, 1983	5.37	77.93	187.98
	Escualosa thoracata (Valenciennes, 1847)	87.98	92.74	8.01
	Anodontostoma chacunda (Hamilton, 1822)	-	9.0	-
	Herklotsichthys dlispilonotus (Bleeker, 1852)	-	15.21	-
	Hilsa kelee (Cuvier, 1829)	-	6.33	-
	Sardinella albella (Valenciennes, 1847)	22.41	45.55	4.57
	Sardinella fimbriata (Valenciennes, 1847)	8.78	-	133.89
	Sardinella gibbosa (Bleeker, 1849)	-	6.23	-
	Sardinella lemuru Bleeker, 1853	-	-	338.48
Ariidae	*Arius macronotacanthus* Bleeker, 1846	3250.56	3991.23	11500.38
	Arius sagor (Hamilton, 1822)	-	2441.42	1879.5
	Cryptarius truncatus (Valenciennes, 1840)	-	-	13.81
	Ketengus typus Bleeker, 1847	-	132.73	240.2
	Osteogeneiosus militanis (Linnaeus, 1758)	57.1	-	-
Plotosidae	*Plotosus canius* Hamilton, 1822	1433.72	753.14	1126.79
Mugilidae	*Chelon tade* (Valenciennes, 1836)	2093.74	3900.06	8891.09
	Liza subviridis (Valenciennes, 1836)	1611.86	496.94	694.85
	Moolgarda seheli (Forsskal, 1775)	-	43.66	-
Atherinidae	*Hypoatherina valenciennei* (Bleeker, 1853)	326.09	164.9	73.82
Belonidae	*Strongylura strongylura* (van Hasselt, 1823)	1555.14	2778.28	3517.1
Hemirhamphidae	*Hyporhariphus (Hyporhamphus) limbatus* (Valenciennes, 1846)	797.76	796.41	1482.83
	Rhynchorhamphus naga Collette, 1976	102.2	24.75	217
Ambassidae	*Ambassis gymnocephalus* (Lacepède 1802)	141.95	1016.88	2065.81
	Ambassis nalua(Hamilton, 1822)	-	-	-
Sillaginidae	*Sillago sihama* (Forsskal, 1775)	152.11	230.81	109.92
Carangidae	*Alepes djedaba* (Forsskal, 1775)	-	9.14	23.26
	Scomberoides commersonnianus Lacepède, 1801	-	5.6	-
	Scomberoides tol (Cuvier, 1832)	-	3.1	-
	Selaroides leptolepis (Cuvier, 1833)	-	0.54	-
Leiognathidae	*Leiognathus decorus* (de Vis, 1884)	241.57	978.9	579.42
	Secutor insidiator (Bloch, 1787)	0.63	7.07	-
	Secutor ruconius (Hamilton-Buchanan, 1822)	1.14	20.41	-
Gerreidae	*Gerres erythrourus* (Bloch, 1971)	12.39	14.71	27.56
Polynemidae	*Eleuthronema tetradactylum* (Shaw, 1804)	834.43	2634.84	2425.5
Sciaenidae	*Aspericorvina jubata* (Bleeker, 1855)	1231.18	1118.89	1205.4
	Chrysochir aureus (Richardson, 1846)	-	-	47.94
	Dendrophysa russelli (Cuvier, 1830)	466.98	180.5	239.71
	Nibea albiflora (Richardson, 1846)	-	219.39	62.72
	Panna microdon (Bleeker, 1849)	230.39	-	251.31

Drepanidae	*Drepane punctata* (Linnaeus, 1758)	29.98	7.74	346.59
Teraponidae	*Terapon theraps* (Cuvier, 1829)	70.22	207.55	53.91
Eleotridae	*Butis butis* Hamilton, 1822	451.08	-	86.4
	Ophiocara porocephala (Valenciennes, 1837)	66.18	1.17	9.21
Gobiidae	*Acentrogobius caninus* (Valenciennes,1837)	92.95	-	-
	Acentrogobius chlorostigmatoides (Bleeker, 1849)	-	4.46	-
	Acentrogobius cyanomos (Bleeker, 1849)	7.52	-	-
	Bathygobius fuscus (Rüppell, 1830)	-	1.26	-
	Favonigobius aliciae (Herre, 1936)	52.17	-	-
	Pseudapocryptes lanceolatus (Bloch & Schneider, 1801)	20.24	-	-
Scatophagidae	*Scatophagus argus* (Bloch, 1788)	123.43	596.89	3968
Uranoscopidae	*Uranoscopus bicinctus* Temminck & Schlegel, 1843	-	33.24	-
Triacanthidae	*Triacanthus biaculeatus* (Bloch, 1786)	-	12.9	-
Cynoglossidae	*Cynoglossus lingua* Hamilton-Buchanan, 1822	33.47	10.51	-
	Cynoglossus puncticeps (Richardson, 1846)	-	34.85	38.05
Tetraodontidae	*Lagocephalus lunaris* (Bloch & Schneider, 1801)	-	43.34	-
	Total	**17017.94**	**24461.81**	**43086.57**

3.5.4 Assemblage structure

Cluster analysis based on the numerical abundance of each species indicated some separation of assemblages by seasons (Figure 3.6), an impression largely confirmed by the nMDS ordination (Figure 3.7). Analysis of similarities (ANOSIM) was performed to establish whether statistical differences existed between the different sampling stations (1-6), night and day catches and season (Table 3.6). No difference was found for stations (R=0.043, P=0.176), and there was a marginal difference between day and night samples (R=0.049, P=0.045), but this is not apparent in the ordination plot (Figure 3.6). However, there was a clearly significant difference between seasons (R=0.154, P=0.001) and this is also evident in the ordination plot (Figure 3.7). Between-season comparisons indicated that the greatest difference in assemblage structure is between the dry and rainy seasons (P=0.003; Table 3.6), also clear in the ordination plot.

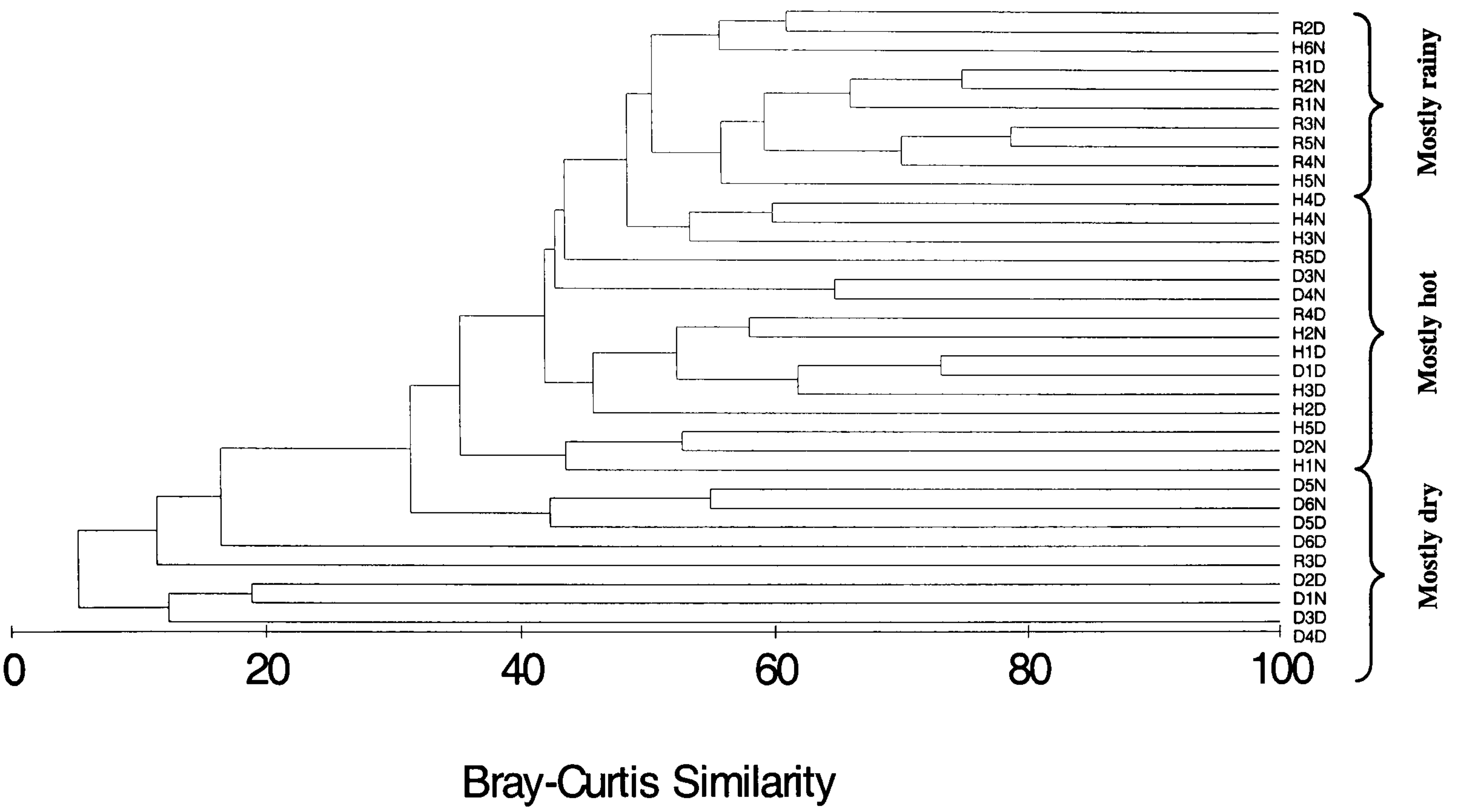


Figure 3.6 Cluster analysis of sampling stations based on species numerical abundance. DD= dry season/day; DN=dry season/night; HD=hot season/day; HN=hot season/night; RD=rainy season/day; RN=rainy season/night. 1-6 are the sampling stations.

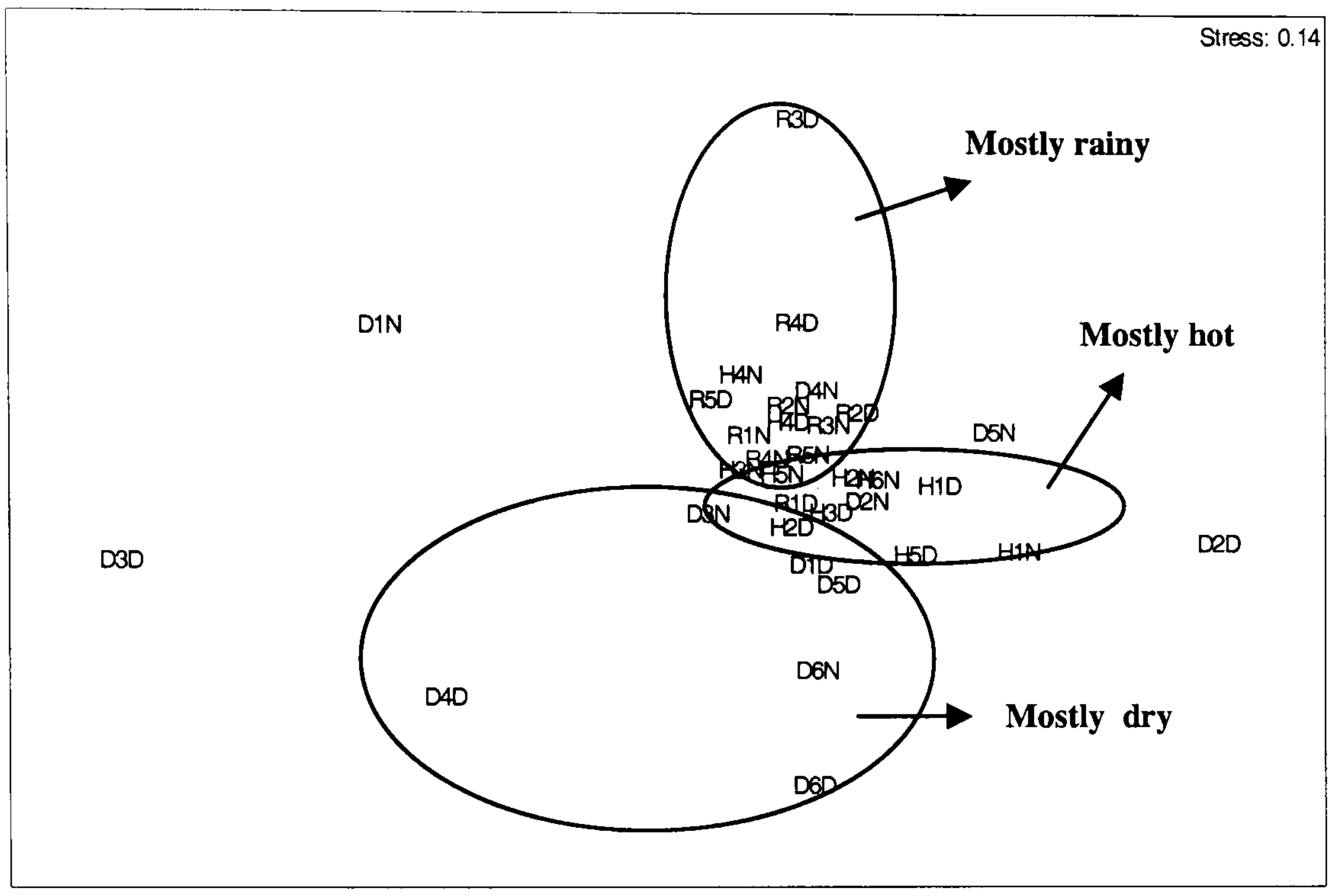


Figure 3.7 Ordination (nMDS) of sampling stations (1-6) based on species numerical abundance. DD=dry season/day; DN=dry season/night; HD=hot season/day; HN=hot season/night; RD=rainy season/day; RN=rainy season/night.

Table 3.7 ANOSIM statistics for comparison of assemblages, night vs day, and between seasons and between stations.

	Seasons*	**Day vs Night**	**Stations**
Global R	0.154	0.049	0.043
Test statistics	0.1%	4.5%	17.6%
Pairwise tests	Significant	Marginally significant	Not significant

* H vs R= 2.2 %
H vs D= 0.6 %
R vs D= 0.3 %

3.5.5 Diet compositions

The diet of each fish species as assessed by composition analyses was shown in Appendix 1. Prey items in the fish stomachs were usually digested and could not be identified to species, hence the grouping of major preys into the food types were categorized into ecological groupings (see Appendix 2) and percentage proportion of food items was shown in 8 highest contribution (Table 3.8). Of the 7,664 stomachs examined, 5,514 contained food and 2,150 stomachs were empty.

Of the 63 fish species, benthic invertebrates formed the most abundance food (31.98%) in diet compositions, followed by sergestid shrimp (*Acetes* spp.) (20.65%). Of the seven most abundance and biomass fish species in the Mae Klong Estuary, sergestid shrimp also found the most abundant diet (52.43%) of *Eleutronema tetradactylum* and occurred in all seasons.

The diets of *Chelon tade* showed the clear dominance of phytoplankton (39.88%) and detritus (42.63%) in compositions and these persisted in all seasons. Benthic invertebrate was also found, 14.36% were recorded. Sergestid shrimp and zooplankton formed a minor part, <2% found in diets.

A wide variety of foods was taken by *Arius macronotacanthus* which consumed large numbers of benthic invertebrates (52.40%), sergestid shrimp (11.07%) and nekton (10.64%). Crab, shrimp, zooplankton and detritus were also found in the diets but less than 10% of each. No phytoplankton and other plant tissue found in the diets.

A diet of *Aspericorvina jubata* consisted mainly of benthic invertebrates (42.32%) and sergestid shrimp (39.72%). Crab was also consisted (7.87%). Many prey item were also found but <4% of each.

Nekton (dominated by juvenile fish) formed the major diet in *Strongylura strongylura* (66.84%). Sergestid shrimp, benthic invertebrate and shrimp were also presented. Crab was also found in very small numbers (around 0.1%) in the diets and occurred only in dry season. Whilst nekton (which included juvenile fish and insect)

Table 3.8 Percentage proportion of food items in 8 highest contribution with % proportion ≥ 0.1; Denote (*) no item in gut.

Fish species	Proportion (%) of items in fish guts																							
	Nekton			Sergestid shrimp			Shrimp			Crab			Benthic invertebrate			Zooplamkton			Diatom&Plant tissue			Detritus		
	D	H	R	D	H	R	D	H	R	D	H	R	D	H	R	D	H	R	D	H	R	D	H	R
Dasyatis fluviorum	-	-	-	63.6	33.3	1.7	-	-	48.3	13.6	33.3	8.3	19.7	33.3	33.3	3.03	-	8.3	-	-	-	-	-	-
Himantura imbricata	-	-	-	23.3	11.7	-	10.0	-	-	43.3	45.0	-	23.3	43.3	-	-	-	-	-	-	-	-	-	-
Elops machnata	-	100	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Pisodonophis boro	-	100	-	-	-	-	-	-	-	-	-	-	-	-	6.2	-	-	-	-	-	-	93.7	-	-
Setipinna taty	-	-	-	-	-	-	-	-	6.0	-	-		-	-	35.1	-	-	-	-	-	-	-	-	-
Stolephorus commersonii	0.2	3.1	1.0	75.1	46.1	23.1	4.8	4.3	13.3	-	-	1.2	19.4	10.8	46.7	0.2	33.2	14.6	-	-	-	0.3	-	-
Thryssa hamiltoni	-	-	-	-	-	97.1	-	-	1.2	-	-	-	-	-	1.8	-	-	-	-	-	-	-	-	-
Thryssa setirostris	-	-	-	100	100	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
Escualosa elongata	-	-	-	-	-	27.0	-	77.8	-	-	-	-	100	22.2	39.7	-	-	6.2	-	-	-	-	-	25.7
Escualosa thoracata	-	-	100	-	-	-	95.6	20.5	-	-	-	-	4.4	59.2	-	-	25.2	-	-	-	-	-	-	-
Anodontostoma chacunda	-	-	-	-	-	-	-	-	-	-	-	-	-	30.0	-	-	-	-	-	70.0	-	-	-	-
Herklotsichthys dispilonotus	-	-	-	-	85.9	-	-	-	-	-	-	-	-	3.3	-	-	-	-	-	-	-	-	10.9	-
Hilsa kelee	-	-	-	-	100	-	-	-	-	-	-	-	-	-	-	-		-	-	-	-		-	-
Sardinella albella	-	-	-	-	92.8	-	-	-	-	-	-	-	40.0	-	100	-	7.1	-	-	-	-	20.0	-	-
Sardinella fimbriata	-	-	-	-	-	-	-	-	-	-	-	-	-	-	73.6	100		-	-	-	-	-	-	15.8
Sardinella gibbosa	-	-	-	-	100	-	-	-	-	-	-	-	-	-	-	-		-	-	-	-	-	-	-
Sardinella lemuru	-	-	-	-	-	18.5	-	-	2.1	-	-	-	-	-	79.3	-		-	-	-	-	-	-	-
Arius macronotacanthus	6.2	23.4	2.9	-	24.3	9.5	13.0	1.1	9.8	9.3	9.3	7.4	43.4	40.6	76.0	17.1	0.5	-	-	-	-	10.9	0.6	-
Arius sagor	-	31.5	2.9	-	15.5	-	-	15.3	70.9	-	26.8	14.3	-	5.4	11.9	-	9.3	-	-	-	-	-	-	-
Cryptarius truncatus	-	-	60.0	-	-	-	-	-		-	-	-	-	-	40.0	-		-	-	-	-	-	-	-
Ketengus typus	-	81.2	64.3	-	-	-	6.2	-	4.5	-	-	-	12.5	-	31.5	-	6.2	-	-	-	-	-	-	-
Osteogeneiosus militaris	-	-	-	-	-	-	100	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

Fish species	Proportion (%) of items in fish guts																							
	Nekton			Sergestid shrimp			Shrimp			Crab			Benthic invertebrate			Zooplamkton			Diatom&Plant tissue			Detritus		
	D	H	R	D	H	R	D	H	R	D	H	R	D	H	R	D	H	R	D	H	R	D	H	R
Plotosus canius	5.2	-	0.1	-	-	-	27.6	7.7	1.0	41.4	31.5	1.5	19.0	60.8	97.4	-	-	-	-	-	-	6.90	-	-
Chelon tade	-	-	-	1.3	3.2	-	-	-	-	-	-	-	28.4	9.5	6.3	1.0	0.6	-	34.3	20.1	60.3	32.1	57.6	32.9
Liza subviridis	-	4.7	-	1.3	5.6	-	-	-	-	-	-	-	2.7	15.2	30.9	0.1	-	-	70.4	45.7	39.9	24.5	28.8	29.0
Moolgarda seheli	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	100	-
Hypoatherina valenciennei	0.5	-	62.5	36.5	38.2	8.3	15.6	28.1	-	-	-	-	11.2	10.0	29.2	13.8	19.2	-	-		-	22.3	4.5	-
Strongylura strongylura	77.8	66.6	56.0	15.6	33.3	4.4	2.4	-	5.8	0.6	-	-	3.6	33.7	-	-	-	-	-	-	-	-	-	-
Hyporhamphus limbatus	89.0	74.1	5.2	2.0	-	-	6.0	17.7	-	-	0.2	-	1.0	1.3	47.9	-	0.1	-	-	-	-	2.0	5.2	0.8
Rhynchorhamphus naga	32.4	71.4	96.0	50.0	-	4.0	9.3	-	-	-	-	-	1.85	14.3	-	6.5	14.3	-	-	-	-	-	-	-
Ambassis gymnocephalus	-	17.6	-	10.9	27.2	24.1	25.0	15.6	1.5	-	-	0.3	19.1	28.9	73.5	18.2	13.8	-	-	-	-	26.8	-	-
Ambassis nalua	-	-	-	-	-	-	-	-	-	-	-	-	-	-	100	-	-	-	-	-	-	-	-	-
Sillago sihama	1.1	-	3.3	84.6	-	-	11.6	50.0	39.3	-	-	13.3	2.08	50.0	42.8	0.5	-	1.2	-	-	-	-	-	-
Alepes djedaba	-	-	100	-	63.6	-	-	-	-	-	-	-	-	36.4	-	-	-	-	-	-	-	-	-	-
Scomberoides commersonnianus	-	60.0	-	-	40.0	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Scomberoides tol	-	-	-	-	100	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
*Selaroides leptolepis**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Leiognathus decorus	-	-	0.3	-	2.1	-	18.0	0.3	5.4	-	2.0		81.9	89.2	93.3	-	6.4	-	-	-	-	-	-	0.9
Secutor insidiator	-	-	-	-	-	-	-	-	-	-	-	-	100	100	-	-	-	-	-	-	-	-	-	-
Secutor ruconius	-	-		-	-	-	-	-	-	-	-	-	100	100	-	-	-		-	-	-	-	-	-
Gerres erythrourus	-	-	25.0	-	66.7	-	-	-	-	-	-	-	-	-	50.0	100	-	-	-	-	-	-	33.3	25.0
Eleuthronema tetradactylum	-	6.3	-	77.6	32.7	46.0	15.9	29.3	39.0	-	-	1.2	1.1	0.9	7.5	2.8	29.1	6.2	-	-	-	2.5	-	-
Aspericorvina jubata	4.8	-	1.5	61.4	0.4	57.3	4.37	0.4	2.3	19.9	0.7	3.0	4.6	98.4	33.7	2.6	0.1	1.5	2.2	-	0.7	-	-	-

Fish species	Proportion (%) of items in fish guts																							
	Nekton			Sergestid shrimp			Shrimp			Crab			Benthic invertebrate			Zooplamkton			Diatom&Plant tissue			Detritus		
	D	H	R	D	H	R	D	H	R	D	H	R	D	H	R	D	H	R	D	H	R	D	H	R
Chrysochir aureus	-	-	-	-	-	-	-	-	-	-	-	-	-	-	100	-	-	-	-	-	-	-	-	-
Dendrophysa russelli	-	-	5.4	61.5	55.0	30.3	6.2	-	4.1	6.2	-	12.0	25.0	38.2	42.7	1.0	6.8	5.4	-	-	-	-	-	-
Nibea albiflora	-	17.5	-	-	70.0	-	-	10.0	100	-	-	-	-	2.5	-	-	-	-	-	-	-	-	-	-
Panna microdon	-	-	-	43.2	-	36.0	2.7	-	30.0	36.0	-	-	18.0	-	20.0	-	-	-	-	-	14.0	-	-	-
Drepane punctata	-	-	-	-	-	-	-	-	-	-	-	-	100	100	100	-	-	-	-	-	-	-	-	-
Terapon theraps	62.5	100	5.0	-	-	-	-	-	-	-	-	25.0	-	-	45.0	37.5		25.0	-	-	-	-	-	-
Butit butis	19.35	-	-	3.2	-	-	38.7	-	-	-	-	-	4.8	-	100	-	-	-	-	-		33.9	-	-
Ophiocara porocephala	-	-	-	76.2	-	-	-	-	-	-	-	-	23.8	-	100	-	-	-	-	-	-	-	-	-
Acentrogobius caninus	-	-	-	-	-	-	-	-	-	20.0	-	-	-	-	-	3.3	-	-	-	-	-	76.7	-	-
*Acentrogobius chlorostigmatoides**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
*Acentrogobius cyanomos**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
*Bathygobius fuscus**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Favonigobius aliciae	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	100	-	-
Pseudapocryptes lanceolatus	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	100	-	-
Scatophagus argus	-	-	-	-	-	-	100	56.2	-	-	-	-	-	24.9	59.3	-	-	-	-	-	40.7	-	-	-
*Uranoscopus bicinctus**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
*Triacanthus biaculeatus**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cynoglossus lingua	20.0	-	-	-	-		60.0	100	-	-	-	-	-	-	-	-	-	-	-	-	-	20.0	-	-
Cynoglossus puncticeps	-	-	-	-	-	14.3	-	-	-	-	-	-	-	-	85.7	-	-	-	-	-	-	-	-	-
Lagocephalus lunaris	-	100	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

formed an important component diet in *Hyporhamphus (Hyporhamphus) limbatus* (66.65%).

3.6 Discussion

3.6.1 Fish fauna

The Mae Klong Mangrove Estuary supports a rich fish fauna in terms of number of species, abundance and biomass. On the 63 fish species, clupeid, gobiids (Gobiidae and Eleotridae), ariid catfish and croaker were the most speciose families among estuarine fishes, and the seven most abundant families (Ariidae, Plotosidae, Mugilidae, Ambassidae, Sciaenidae and Polynemidae) are known to inhabit mainly mangrove estuaries. This is consistent with the findings of others researchers in Thailand (Poovachiranon and Satapoomin, 1994; Sudara et al.1994; Vidthayanon and Premcharoen, 2002; Hajisamae et al. 2006), Malaysia (Chong et al. 1990; Sasekumar et al. 1994; Singh and Sasekumar 1994), Kuwait (Abou-Seedo et al. 1990), Taiwan (Lin and Shao 1999), Africa (Little et al. 1988; Kimani et al. 1996; Whitefield 1999; Albert et al. 2004; Crona and Rönnbäck 2007) and Australia (Roberson and Duke 1990a; Halliday and Young 1996). Also, these diverse fish assemblages have a high value for the local communities as a large proportion of the species (>30 species) are utilized by subsistence fisheries (May 2005, cited in Unsworth et al. 2007). Fish assemblages showed a common characteristic for tropical estuarine fish populations (Robertson and Duke 1990a; Sasekumar et al. 1994; Vidthayanon and Premcharoen, 2002; Ikejima et al. 2003).

Around half of the species collected were juveniles, being dominated by engraulids, clupeids, ambassids, leiognathids and sciaenids. These and other less dominant juveniles, such as ariids and mugilids, are known to use estuaries during their juvenile stages (Robertson and Blaber 1992; Blaber 1997). Thus the present results support the view of mangrove estuaries serving as nursery grounds.

Gobies and anchovies are often the numerically dominant taxa in estuarine and coastal ichthyoplankton communities worldwide (Newton 1996; Sanvicente-Anorve et al. 2002). Their dominance tends to be most conspicuous in low-salinity areas, contributing to low diversity-index values that are typical of oligohaline larval and nursery areas in many estuarine systems (Newton 1996). In the present study, gobiids might therefore be expected to the most diverse family and should form a large proportion of the fish community in the mangrove. The lower numbers of gobiids recorded in this study was likely an effect of the sampling gear, which was the same as by locals used, not the small seine that can be dragged along the mangrove creek where high goby numbers occur on the muddy sediments. This is consistent with the findings of Blaber and Milton (1990) who found that gobies dominated both in diversity and abundance on muddy-bottoms, but less so on hard-bottom mangrove estuaries. Another reason for the fewer than expected gobies may be the relatively large mesh size of the push net used. It is noteworthy, that gobies were the most dominant teleosts in epibenthos samples collected using a trawl net with smaller mesh size in mangrove embayments in Australia (Daniel and Robertson 1990) and that gobiid larvae and juveniles are often particularly abundant in ichthyoplankton in tropical mangrove estuaries, including Thailand (Blaber et al. 1997; Janekarn and Boonruang 1986; Little et al. 1988). In their reviewed of fish density and biomass in tropical and subtropical mangrove systems, Roberson and Blaber (1992) pointed out the difficulties of comparing studies which use different gears and sampled different microhabitats.

Species likely to remain in the mangrove habitat throughout their lives are regarded as permanent residents, whereas partial/temporary residents include fishes that regularly use the mangrove but normally remain there for a part of their life history (usually only as juveniles) (Bell et al. 1984). The results presented here tend to support the concept that resident fishes predominate in mangrove estuaries. 84% of the total fish recorded most of which were dominated in terms of species composition, abundance and biomass, could be classified as resident.

Ariid catfish dominated the assemblage in terms of biomass and abundance, consistent with the findings of Singh and Sasekumar (1994) in the Matang mangrove, Malaysia and of Wright (1986) in Nigeria. They also found that this taxon was widely distributed and found in the inshore waters, mudflats and mangrove channels.

Sciaenids dominated the assemblages in terms of species composition and biomass. Like ariids, these euryhaline fishes are able to survive even at low salinity (Yap et al. 1994). *Ambassis gymnocephalus*, which dominated most abundant family, has postlarval, juvenile and adult phases restricted to the mangrove habitat (Robertson and Duke 1990b; Sasekumar et al. 1994). This species is known to have a very low tolerance to lowered salinities, and will move out of estuaries into marine embayments after heavy rains which lower estuarine salinities (Robertson and Duke 1990a). Gerreid is typically associated with mudflats and not with mangroves (Nagelkerken and Faunce 2007). Some reef-associated fish species (e.g. carangids) were colleted from this study, but occurred in low number. The presence of these fishes is indicative of the dependence of some reefal fish on mangroves as a nursery area and these fish undergo ontogenetic habitat shifts to coral reefs as they grow (Weis and Weis 2005).

3.6.2 Seasonal variations in the assemblages

Mangrove fish communities are highly variable in both short (tidal) and longer time scales (seasonal) because of pronounced environmental fluctuations (Rehage and Loftus 2007). Seasonal changes in the abundance and composition of tropical and subtropical of fish communities have been reported in mangrove systems throughout the world, including Madagascar (Laroche et al. 1997), Brazil (Barletta et al. 2005), Australia (Loneragan et al. 1986, the Solomon Islands (Blaber and Milton 1990), Taiwan (Lin and Shao 1999), and Mexico (Yanez-Arancibia et al. 1988). In the Mae Klong Estuary, species composition was highly seasonal with peak diversity in the hot season and lower value in the dry season, consistent with the findings of previous studies in other tropical regions (Spash et al. 2004). Abundance was highest in the rainy season, also in general agreement with other studies (Thayer et al. 1987; Robertson and Duke 1990a; Ikejima et al. 2003). Such seasonal changes in species compositions and abundance may be a reflection of the breeding patterns of fish and changes in food availability (Robertson and Duke 1990a). The rainy season coincided with the period of greatest recruitment of juvenile fishes and greatest zooplankton abundance in a mangrove estuary in tropical Australia (Robertson et al. 1988; Robertson and Duke 1990b). The availability of prey for juvenile fishes in the

Mae Klong Estuary would also increase in the rainy season, since crustacean larvae were most abundant at this time in other mangrove areas of Thailand (Boonruang and Janekarn 1985). Many fish species spawn during early summer, which coinsides with the influx of postlarvae and juveniles into estuarine areas in late summer after their planktonic phase (Laegdsgaard and Johnson 1995).

It is widely acknowledged that many interacting physical and biological factors influence the occurrence, distribution, abundance and diversity of tropical estuarine fishes (Whitefield 1998; Blaber 2000). Local population abundance is a response to changes in local environmental conditions as well as to large-scale seasonal migrations, particularly of immature life stages. While these changes have frequently been related to salinity and /or temperature, Blaber and Blaber (1980) consider that these variables probably do not affect the distribution of the juveniles of estuarine-dependent species. In the Mae Klong Estuary, salinity values were not very different among seasons, can sequently had little influence on the assemblage structure observed. In fact, most of the species recorded probably have wide tolerance limits to the fluctuating conditions found in this system. The salinity values recorded in the rainy season might be expected to those in other seasons, but the samples in rainy season were taken in August and this was not the highest rainfall month (see Table 3.9).

Table 3.9 Monthly rainfall (mm) at Samut Songkhram during 2004-2006 (Royal Irrigation Department, Thailand, pers. comm.).

Water days	Year	A	M	J	J	A	S	O	N	D	J	F	M	Annual
62	2004	30.0	218.0	123.0	164.4	128.8	253.7	15.0	0.0	0.0	9.0	11.0	55.0	1007.9
62	2005	42.5	164.3	170.8	164.7	46.5	144.1	393.3	58.0	64.0	0.0	9.0	5.5	1262.7
42	2006	22.4	35.5	135.5	32.9	71.1	103.3	151.3	0.0	0.0	0.0	0.0	0.0	552.0

From my study, salinity alone therefore, could not explain the differences in fish composition and abundance among seasons. Although seasonal differences were apparent in the Mae Klong estuary (Figure 3.7), the fluctuations were relatively small. Thus, the rainy group in the ordination is characterized by many more individuals, compared to the dry season (Figure 3.3b), an influx of the engraulids *Setipinna taty* and *Thryssa hamiltoni* as well as *Sardinella lemuru* (Table 3.6). In any case, because most fishes living in tropical estuaries are broadly euryhaline (Blaber 1997), salinity may not play an important role in structuring such fish assemblages. Most estuarine fish are able to cope with salinity fluctuations but their ability to do so varies from species to species and hence influences their distribution (Albaret et al. 2004). Due to the high degree of dominance in the assemblages, seasonal variation was mainly the result of changing distributions and abundances of dominant species at various scales, while changes in species composition were largely driven by the presence and absence of additional rare species (Gibson et al. 1996).

3.6.3 Diet compositions and trophic structure

In this study, the diet data of fish is largely in agreement with previous studies in the estuarine ecosystems (Boonruang and Janekarn 1994; Boonruang and Satapoomin 1997; Hajisamae et al.2003). All species examined in my study displayed characteristics typical of estuarine fishes, including omnivory and broad dietary overlap. Fish had a varied diet, which in part was influenced by their morphology and size, but in general displayed a lack of specialization (Mbabazi et al. 2004). Many fish species are likely to be opportunistic feeders, exhibiting no selectivity in their choice of prey species and consuming prey items in similar proportions to their occurrence. My study was not designed to obtain quantitative data by the occurrence method so there is no evidence to determine whether species were selective or opportunistic feeders.

Omnivory is common in the Mae Klong Estuary, supporting Ley et al. (1994) who stated that estuarine fish are generally omnivorous, sharing common resources and being flexible in their exploitation of temporary peaks in prey populations. This means that, although fish species composition may differ considerably between estuaries, the basic trophic structure within estuaries is generally very similar (Elliott et al. 2002). Futhermore, since most estuarine fishes are either generalist feeders or opportunists (Baldó and Drake 2002; Elliott et al. 2002), differences in estuarine small-fish assemblages based on their feeding guilds may reflect estuarine differences in prey communities. I have assumed that most fish inhabiting the estuary are juveniles that use it as a nursery area and do not exhibit major trophic changes. This assumption is consistent with the view that herbivorous fish species do not change trophic status during ontogeny, although carnivorous fish may feed on gradually larger prey (at correspondingly higher trophic levels), before they migrate out to the open sea (Sosa-López et al. 2005).

In the Mae Klong Estuary, sergestid shrimp formed the dominant group in several fish diets. This agrees with the study of Sudara et al. (1994) in the mangrove area of Samut Songkharm, Thailand. They showed that this area is famous for shrimp paste production from those of the mysid groups, especially the sergestid shrimp (*Acetes erytraeus)*, and could be collected all year round. Other shrimp and prawn species were present in my study area and should had been more abundant than recorded. Since both postlarvae and small juvenile prawns are digested very rapidly in fish stomachs (reduced to ~30% of original dry weight 1h after ingestion) (Haywood et al. 1998), this could partly explain the low numbers of small prawns found in fish stomachs. However, I attempted to minimize this effect by removing fish from the nets frequently and fixing in the buffered formalin very soon after capture.

Copepods, amphipods and mysids were the preferred prey of the zooplankton-feeding guild of the Mae Klong Estuary, as previous studies in other estuarine environments have also shown (Boonruang et al. 1994; Sudara et al. 1994; Baldó and Drake 2002; Boondao 2006). Boondao (2006) also stated that the most abundance of zooplankton group in the Mae Klong Estuary was Arthopoda, especially copepod nauplii. Zooplankton densities in estuaries are strongly associated with river flow through the introduction of nutrients and the stimulation of phytoplankton growth (Wooldridge 1999). For several of the zooplankton-feeding species, feeding preferences changed; mysids replaced copepods progressively in the diet of postlarva and juvenile fish as they grew (Baldó and Drake 2002). Therefore, size and availability of prey seem to be the principal factors in determining the trophic guild structure of the small-sized fish assemblage studied in the Mae Klong Estuary. Trophic relationships between two species can also change through ontogeny and the degree of niche overlap between two species may also vary ontogenetically (Piet et al. 1999).

Benthic invertebrates were the main component in the diet of ariid catfish. This supports the study of Wichitwarakhun (2001) who found that major benthic groups in the Mae Klong Estuary were polychaetes, crustaceans and gastropods. She also revealed that sediment characteristics, topography, tidal period, organic content, plant biomass, and mangrove forest structure were major factors determining species composition and distribution of the benthic community in the area.

The diet of mugilids was similar to that found in previous studies (Odum and Heald 1972; Boonruang et al. 1994; Vitheesawat et al. 1997). They were the only fish with relatively high percentages of phytoplankton and detritus. Blaber (1985) stated that in all southeast African estuaries, the most numerous fishes are the iliophagous species (mainly mullet) and that detritus, together with epipsammic algae and periphyton, provide a major energy input into the fish community. Organic detritus is a key food item for most fishes and has an important role in estuarine food webs (Darnell 1961).

Benthic invertebrates and sergestid shrimps were the main component of the diet of *Aspericorvina jubata*, and this agrees with the study of Yap et al. (1994). Prey items of sciaenids vary among groups, depending on their mouth characteristics (Yap et al. 1994). Sciaenids with terminal mouths usually feed in mid water, whereas those with subterminal mouths feed at the benthic surface. These feeding habits are likely to reduce food competition between the two groups.

Eleutronema tetradactylum feed mainly on sergestid shrimp and prawns, supporting the finding of Haywood et al. (1998). Salini et al. (1998) stated that this species was one of the three main predators (with *Polydactylus sheridani* and *Lates calcarifer*) in the Norman River Estuary, Australia.

In conclusion, the diet diversity of most fish species in the Mae Klong Estuary seems to reflect a lower or higher availability of prey: during the warm period, an abundant food supply in the estuary reduces competition; in winter, the low densities of the main prey make a certain diversification of diet necessary (Baldó and Drake 2002). This study is a 'snapshot' view of the diets of the fish species. It aims to provide information on the general trophic structure of the estuarine fish assemblage in the Mae Klong Estuary for input into a mass balance model described in chapter 4.

CHAPTER 4

A trophic model of Mae Klong Estuary, Inner Gulf of Thailand, with reference to the fish community

4.1 Introduction

It has been widely recognized that ecosystem structure and function are important for the sustainability of living aquatic resources, particularly trophic structure and flows of biomass between species (Christensen and Pauly 1995; McCann 2000). Measurements of biomass transfer between functional groups and trophic efficiency provide information which can be used to evaluate the impact of change on particular groups and the way changes are propagated through the whole ecosystem via the trophic web (Ulanowicz 1986; Christensen and Pauly 1993). Futhermore, in ecosystem-based fisheries management, prey and predators cannot be managed independently (Kitchell et al. 2004), so that an understanding of trophic structure is essential for fishery assessment and management (McCann 2000). The complexity of ecosystems can be handled using ecological models (Jørgensen 1994; FAO 2007), in the present context defined as descriptions that emphasize some aspects of the system in order to understand how they work, which are ecosystem representations that permit an understanding of complexity in energy terms, which identify levels of production, which allow comparisons between ecosystems, and for the evaluation of the functional responses to natural and/or anthropogenic impacts (Christensen and Pauly 1992a). Models can be physical, verbal, graphical or mathematical, reflecting the interest of the modeler (Haddon 2001, cited in Freire et al. 2008), and are useful to help managers identify how decisions can affect the various components of an ecosystem (Janjua 2007).

In the case of mangrove systems, the ecosystem of focus here, many studies have examined the incorporation of mangrove production into organisms ranging from zooplankton (Bouillon et al. 2000) to mobile marine invertebrates (Fry and Smith 2002; Werry and Lee 2005) and fishes (Nagelkerken and van der Velde 2004a, b).

These mobile organisms may serve as a pathway for export of mangrove-derived nutrients (Vega-Cendejas and Arreguín-Sánchez 2001), which is incorporated into food webs both within and adjacent to mangroves (Odum and Heald 1972). However, more recent studies indicate that less obvious primary producers (phytoplankton, micro-and macro-algae) may be more important than mangrove leaves or detritus, because of the higher nitrogen content of microalgae and macroalgae compared to mangrove matter (Loneragan et al. 1997). To understand mangrove ecosystem dynamics, a study of multispecies interactions is needed including trophic fluxes, assimilation efficiencies and energy transfer and dissipation (Ulanowicz 1997), reflected in the diversity, abundance, distribution and persistency of the biological components ultimately regulated by primary productivity (Oksanen et al. 1981; Oksanen 1983), environmental variability (Pimm and Kitching 1987) and a combination of both (Persson et al. 1992).

Mangroves occur in tropical areas, often in developing countries, where about 60% of the world's fish catch is taken. Yet the dynamics of these resources have not been well studied (Bundy and Pauly 2001). The fisheries are typically multispecies, with over a 100 or more species landed for immediate consumption, trade, fishmeal and other animal food or fish sauce (Pauly 1996). This is especially true in southeast Asia, where marine biodiversity is very high (Eckman 1967), and where the fisheries are also extremely diverse (Pauly 1988). Mangroves serve as both a source of energy and a habitat for young fish.

The Gulf of Thailand estuaries are recognized as productive systems that serve as important nursery areas for juvenile marine fishes and invertebrates (Vathanachai 1979; Monkolprasit 1994; Boonruang and Satapoomin 1997; Janekitkarn et al. 1999; Vidthayanon and Premcharoen 2001, 2002; Ikejima et al. 2003). One of the reasons for this is the large cover by mangrove forest, and the outflows from the four main rivers, Tha Chin, Mae Klong, Chao Phraya and Bang Pakong. These rivers transport a large amount of silt annually to create deltas, providing a food supply which is richer and more predictable than in the open sea (Menasveta 1976). However, the Gulf is located in a region strongly affected by contaminants from industrial wastes, agricultural wastes, waste from aquaculture and from the municipalities along the coastline, where major biogeochemical transformations are important and rapid. These wastes have caused a low quality of seawater and a nutrient enrichment problem in the inner Gulf, endangering many valuable marine resources, like fisheries and aquacultures as well as reducing the aesthetic value of the inner Gulf (Wiriwutikorn 1996).

"The rapid expansion of fisheries in the Gulf of Thailand has raised much economic and environmental concern about its management. In particular, the ecosystem of the Gulf has changed dramatically as a consequence of over-exploitation of demersal stocks. An increasing proportion of undersized fish and decreasing volume of commercially important species in the composition of the fish catch in recent years suggests symptoms of biologically overfished resource stocks, threatening the fisheries" (Ahmed et al. 2007). These stocks were replaced by squid, but there is a fear that, without proper management, these will in turn be overexploited and possibly be replaced by a non-commercial species such as non-edible jellyfish (Mohamed et al. 2005), these and other species shifts now occurring throughout the world (see Table 4.1). Moreover, overfishing in the coastal regions has forced fishing fleets further offshore, leading to conflicts between neighboring countries. Many offshore transboundary and migratory stocks are, in fact, shared by several countries, stressing the need for an increased understanding of offshore oceanography and marine ecology, with respect to primary and secondary production, identification and understanding of spawning grounds, egg and larval transport and species diversity (Ahmed et al. 2007).

Simultaneously, a number of ministerial laws and regulations have been issued in response to the marine resources situation in the Gulf of Thailand, including the Department of Fisheries (Thailand) 1997:

- Prohibition of motorized trawl and push net fishing within 3 km was issued on July 29, 1972;
- Prohibition of coral and coral reef fishing was issued on January 10, 1978;

Table 4.1 Example of documented shifts towards smaller, high-turnover species in exploited multispecies communities (Daskalov 2002; Vibunpant et al. 2003; Pitcher 1998, cited in Mohamed et al. 2005).

Fishing grounds/Stocks (period)	**Documented species shift**
Gulf of Thailand Demersal stocks (1960-1980)	Stock biomass reduced by 90%; residual biomass dominated by trash fish
Philippine shelf Small pelagics (1950-1980)	Gradual replacement of sardine-like fishes by anchovies
Carigara Bay, Philippines All fish (1970-1990)	Fish replaced by jellyfish, now an export item
Black sea	Small planktivorous fish and jellyfish replace large table fish
North sea	Halibut and small sharks extinct; cod and haddock threatened; demersal omnivores and small pelagics favoured
Humboldt Current, Chile	Large hake depleted, small pelagics favoured
North Pacific	First marine mammal depletion, followed by huge trawl fisheries: Pollock favoured
South China Sea, Hong Kong	Crockers and groupers almost extinct; small pelagics now bulk of fishery

- Prohibition of squid fishing using light attraction with mesh sizes of less than 3.2 cm was issued on November 5, 1981;
- Prohibition of landing any berried crabs (*Scylla serrata*, *Portunus pelagicus*, *Charybdis ferriatus*) was issued on July 11, 1983;
- Prohibition of fishing all species of marine turtles including their eggs was issued on March 13, 1989;
- Prohibition of purse seine fishing with light attraction and with mesh size of less than 2.5 cm was issued on November 14, 1991;
- Prohibition of any fishing using light attraction with mesh size less than 2.5 cm was issued on March 15, 1966. Anchovy fishing boats with sizes (LOA) of less than 16 cm as well as lift net and 'drop net' were exempted from the regulation;
- Requirement that shrimp trawls should install and use a Turtle Excluding Device for fishing was announced on September 16, 1996;
- Prohibition of motorized push net fishing in Pattani Gulf and the coastal area of Pattani Province was issued on February 26, 1998;
- Prohibition of 'drop net' and lift net with light targeting anchovy in the area of Songkhla Province was issued on July 28, 1998 (Songkhla Province Offfice 1998).

Exemption from these regulations can be granted only to activities involving scientific research upon approval of the Director-General of the Department of Fisheries.

The complex trophic interactions between mangrove fisheries and ecosystem effects of fishing can be explored with the mass-balance ecosystem model Ecopath (Christensen and Pauly 1992 a, b; Vega-Cendejas 2003). This model allows the refinement of knowledge and management through an iterative approach to learning and adaptive (or experimental) approach to conservation and fisheries management (Okey, 2004). This method has relatively limited data requirements, yet provides an ecological perspective for the assessment and management of multispecies, multigear fisheries (Bundy and Pauly 2001).

The purpose of the model is to provide accessible 'views' of the whole system and to predict how it might respond to changes in human action or other stresses. The model may also provide insights into the underlying ecological mechanisms operating in the system and explore possible solutions to conservation problems. Through the biomass and production rate data and knowledge of the trophic interactions, a better understanding of the possible fluctuations in group abundances under the influence of fisheries or natural impacts can be obtained (Velasco and Castello 2005). Such analyses can be used to generate hypotheses about the dynamics of a particular system and to address questions such as:

- Which functional groups currently exert large effects on the system?
- What are the potential ecosystem consequences of removing particular species from the system?
- Are any species in this system currently being fished at an unsustainable level?
- To what extent will fisheries exclusion zones alleviate declines of overfished species or restore previous abundances?

Mass-balance models require basic data on biomass of different fish groups. Fisheries statistics from mangrove estuaries in the inner Gulf of Thailand have never been collected on a regular basis, but some data are available from previous research on the fishery in the Gulf of Thailand (Christensen 1998; Khongchai et al. 2003; Kongprom et al. 2003; Vibunpant et al. 2003) and these, together with data from the present study collected from Mae Klong Estuary during December 2005 and August 2006, can be used to build an ecosystem model. In previous studies, the following types of surveys were conducted: trawl survey, landings survey, and fishing gear inventory. These surveys provide data on species composition, trawlable biomass, length and weight of fish, fish landings, fishing effort, and use and number of different fishing gears.

The rational behind the present study is to present a tool for evaluation and management of fisheries and mangroves in Mae Klong Estuary. Specifically to (1) make a mass balance model of trophic interactions in Mae Klong Estuary among seasons; (2) analyze the energy flow patterns using Ecopath with Ecosim (EwE) with special emphasis on the fish community; (3) explore the potential and limitations of EwE for assisting in ecosystem-based management.

Before discussing the methodology and approach used to construct the model, it is relevant to discuss the application of EwE to fisheries, so that the results can be placed in context.

4.2 Ecosystem-based fisheries management

Globally, fisheries resources have been declining since late 1980's (Watson and Pauly 2001; Christensen et al. 2003), with many large-scale fisheries around the world collaspsing (Pauly et al. 2002). At the same time, the number of overexploited stocks increased by a factor of 2.5 between 1980 and 1990 (Alverson and Larkin 1994). The recent collapse of some fish stocks along with the uncertainty involved in managing marine systems have prompted fisheries scientists to suggest a precautionary approach (Sanchirico et al. 2006). What was perceived for a long time as an inexhaustible resource suddenly seems quite limited (Rosenberg et al. 1993). The most important pressures being exerted on the ocean ecosystem are overfishing, destruction of coastal ecosystems, pollution through oil spills and illicit disposal, land-based contamination and climate change (Constanza et al. 1998).

Throughout all oceans of the world, intensive exploitation has led to dramatic changes in the structure and productivity of marine ecosystems (Fogarty and Murawski 1998), directly (fisheries catch) or indirectly (changes in the food web structure, habitat disturbance). At present, the United Nations Food and Agriculture Organization reports that roughly 70% of fish stocks for which data are available are fully exploited or overfished (Pomeroy 2003). Overfishing has become more important and simultaneously more difficult to manage (Ludwig et al. 1993). Overfishing reduces catches and diminishes the genetic diversity and ecological resilience of the exploited populations (Botsford et al. 1997; Pauly et al. 1998). As a result, the long-term sustainability of many fish stocks and the stability of large marine ecosystems appear threatened. It is important to focus on these issues not only to preserve the biodiversity of our planet, but also because more than one billion people now rely on fish as their main source of animal protein, income and/or livelihood (Pomeroy 2003).

For many years, marine systems have been studied and managed from a single species point of view. However, there is an awareness that this traditional way of managing fisheries is not sufficient (Hofmann and Powell 1998). With the necessity to understand in detail the nature and dynamics of exploited marine ecosystems, and more precisely the complexity of species interactions, the development of an ecosystem approach (Kröger and Law 2005) for management of the marine system is becoming more and more important (Kröger and Law 2005; Choi et al. 2005).

There has been considerable recent interest in ecosystem-based fisheries management (EBFM) (known as ecosystem approach to fisheries (EAF) in Europe), as evidenced by several important reports (Christensen et al. 1996; Link 2002; Metcalf et al. 2008). The concept of ecosystem-based fisheries management emerged within the 1982 UN convention on the Law of the Sea (UNEP 2001). Several factors have contributed to the current relevance and awareness of this issue, including conflicts between stakeholders and legislation, debate over the most important processes in an ecosystem, limitations of single species management, and the use of this perspective to justify many different positions (Link 2002). EBFM requires recognition of system-component interactions in determining management targets. Some argue that ecosystem-based fisheries management has the potential to account for risks inherent in managing interacting populations in uncertain and changing environments (Hofmann and Powell 1998), while others directly equate EBFM with taking a precautionary approach (Gerrodette et al. 2002). A comprehensive ecosystem-based fisheries management approach would require managers to consider all interactions that a target fish stock has with predators, competitors, and prey species; the effects of weather and climate on fisheries biology and ecology; the complex interactions between fishes and their habitat; and the effects of fishing on fish stocks and their habitat (Ecosystem Principles Advisory Panel 1996). An ecosystem-based approach to fisheries management also addresses human activities and environmental factors that affect an ecosystem, the response of the ecosystem, and the outcomes in terms of benefits and impacts on humans. Human activities, include commercial and recreational activities from which coastal communities derive income, pleasure, and cultural identity. Human benefits and impacts can also include non-consumptive values arising from nature watching, or the value that an inland resident may place on knowing that an ecosystem is healthy (Pomeroy 2005).

The goal of ecosystem-based fisheries management is to maintain ecosystem health, integrity and sustainability. One of the distinguishing features of ecosystem-based fisheries management is an emphasis on protecting the productive potential of the system that produces resource flows, as opposed to protecting an individual species or stock as a resource. For an ecosystem that is already degraded, however, sustainability requires restoring those parts of the ecosystem that will sustain a diversity of species. The restoration of degraded ecosystems poses particularly difficult decisions related to balancing human need with resource productivity requirements. The human component of marine ecosystems may exhibit irreversible regime shifts with poorly understood thresholds and limits, similar to those more commonly associated with the living marine resource components. The ecosystem approach also recognizes the complexity and uncertainty in predicting responses to management actions (Pomeroy 2003, 2005).

To address the world's ever-growing environmental problems, a comprehensive understanding of the structure, function and regulation of major ecosystems is essential (Pahl-Wostl 1993; McCann 2000). Constructing models to examine the behavior of an ecosystem is therefore the focus of much contemporary research. In the present study, ecological network analysis, such as can be done within Ecopath with Ecosim, is used to develop structural ecosystem models so that the effects of species changes can be assessed and various high-level metrics related to the health of the system can be estimated (Christensen and Walters 2004; Dame and Christian 2006).

4.3 Network analysis of food webs

The importance of interactions in an ecosystem has resulted in the development of a theoretical approach and a set of computational methods called "Ecological Network Analysis (ENA)" (Ulanowicz 1986). ENA is a modelling technique used for understanding the structure and flow of material within ecosystems, and is most commonly used for evaluating food webs (Christensen and Pauly 1993). Trophic flows in ecosystems can be studied by computation of biomass, production and bioenergetics parameters, such as consumption. The measurement of energy and material flows between the various ecosystem components provides significant insight into the fundamental structure and function of the system. The efficiency with which energy and material is transferred, assimilated, and dissipated conveys significant information about the structure and function of food webs (Ulanowicz 1986, 2005). ENA has been used widely in aquatic ecosystems as an empirical tool to study carbon and energy flow between trophic levels and for examining the dependence of various functional groups on sources of energy which change in time and space (Johnson et al. 2001).

ENA can be used to quantify the health, integrity and maturity of ecosystems and also help to evaluate the magnitude of stress imposed on an ecosystem (Christensen and Pauly 1998). Odum (1969) formulated a set of hypothesis to predict long term responses of ecosystems under stress that incorporate elements of trophic links, size, structure and functioning of communities. Odum's ideas defined system characteristics that explain the maturity, stability, and resilience of an ecosystem, and these have been developed by Ulanowicz (1980) so that they can be represented by indices, such as total system throughput (T), ascendancy (A), system capacity (C), and system overhead (see below).

Research using trophic network analyses have produced methodological, theoretical and empirical advances and development of software packages for ecological trophic analysis. There are numerous examples of ENA in the ecological literature, and its acceptance as an established methodology is apparently growing. Two software packages are typically employed for ENA: Ecopath (http://www.ecopath.org)and NETWRK (http://www.cbl.umces.edu/~ulan/ntwk.html) (Christensen and Pauly 1992a). These network models use the trophic relationships among primary producers, herbivores, intermediate consumers, top predators and detritus in food webs to provide opportunities for comparative analysis of whole ecosystems (Table 4.2). Network analysis of food webs and dynamic simulation capabilities, such as those used in the mass-balanced trophic modelling approach Ecopath with Ecosim (EwE), exemplify this advancement (Polovina 1984; Christensen and Pauly 1992a, b; Walters et al. 1997, Walters et al. 1999; Pauly et al. 2000; Christensen et al. 2005). Such whole system modelling approaches are built on empirically based characterizations of food webs, and sometimes represent knowledge distilled from major scientific programs, or from many decades of empirical research. These new approaches to ecosystem synthesis and analysis can help provide unprecedented insights into how nature works and how humans influence nature (Gaedke 1995).

4.4 Mass balance models: Ecopath with Ecosim (EwE)

Mass balance models are a well established group of ecosystem models (Rice 2000). Ecopath trophic models are mass-balanced models, (or more accurately mass-continuity models (Okey 2004)), an approach developed over the last 20 years by Villy Christensen working chiefly with Daniel Pauly and Carl Walters at the Fisheries Centre of the University of British Columbia, Canada, as well as with a large number of collaborators (BSRP 2004). The idea of Ecopath is based on Lindemans' tropho-dynamics ideas, which views the ecosystem in terms of trophic relations defined by energy transfer (Lindeman 1942). This model includes all biotic components of an ecosystem, represented by trophically linked biomass 'pools', the typical currency of which is biomass wet-weight (used here) and their interactions for a given period (e.g., a year or season) (Christensen et al. 2005). The biomass pools consist of a single species, or species group representing ecological guilds (e.g. as producers, primary consumers and secondary consumers) each successively dependent upon the preceding level as a source of energy (Lindeman 1942). Ecopath was originally

Table 4.2 Example of output and indices from Ecopath and NETWRK. Trophic structure analysis and information analysis are similar in each, but matrics given by input-output analysis differ. Also, Ecopath characterizes flow pathways whereas NETWRK focuses on cycling structure (Dame and Christian 2006).

<table>
<tr><td>Input-Output Analysis[A] – quantifies direct and indirect relationships between compartments.
• Mixed Trophic Impact Matrix – sums the positive and negative impacts of each compartment on every other compartment</td><td>Input-Output Analysis[B] – quantifies direct and indirect relationship between compartments.
• Total Contribution Matrix – gives the percent of flow through a compartment that passes into another.
• Total Dependency Matrix – gives the percent of flow through a compartment that had once passed through another (e.g. extended diet)</td></tr>
<tr><td colspan="2">Trophic Structure Analysis[C] – provides information based on the trophic concepts of Lindeman (1942)
• Effective trophic level –fractional value of a compartment's trophic level that takes into account degrees of omnivory.
• Trophic efficiency - the proportion of consumption passed up the food chain.
• Omnivory Index – variance of trophic levels in a consumer's diet.</td></tr>
<tr><td>Pathway Analysis[A] – characterizes the pathway of flows.
• Pathway from any primary producer to a selected consumer through a specified prey.
• Primary production required to sustain the consumption of each group.
• Herbivory: Detrivory Ratio- quantifies the ratio of flow along grazing and detrital food webs.</td><td>Biogeochemical Cycle Analysis[B] – evaluates the characteristics of cycle within the system.
• Number of cycles organized by the smallest common flow.
• Length of cycles and dritribution of flow along them.
• Finn Cycling Index – amount of flow involved in cycling.</td></tr>
<tr><td colspan="2">Information Analysis[C]– quantifies attributes characteristic of the growth and development of the system.
• Total System Throughput – sum of all flows occurring in a system.
• Development Capacity – index of the potential of a network to develop given its particular set of connection and throughout.
• Ascendency – index of the size and developmental potential a system has attained.</td></tr>
</table>

[A] Ecopath software output.
[B] NETWRK software output.
[C] Output of both Ecopath and NETWRK.

proposed in the 1980s (Polovina and Own 1983; Polovina 1984), and was subsequently expanded by various researchers to include temporal and spatial ecological analyses (Walters et al. 1997, 1999, 2000) and policy optimizations

(Walters et al. 2002). Later, Christensen and Pauly (1992a) started to extend the idea and developed the 'new' ECOPATH II software for PC's (released in 1992). The software was distributed widely from ICLARM (International Center for Living Aquatic Resources Management), Metro Manila-Philippines.

The Ecopath model has been widely applied to aquatic systems (Christensen and Pauly 1993; Pauly and Christensen 1995; Walters et al. 1997; Walters et al. 1999; Pauly et al. 2000; Christensen et al. 2005), mostly from the fisheries point of view (Christensen and Pauly 1992a, b). A list of the many applications of Ecopath can be found at: http://www.ecopath.org, along with the freely distributed software and documentation.

EwE includes three main modules (Figure 4.1). Ecopath itself is used to organize historical data on trophic interactions and population sizes based on an assumption of mass-balance; Ecosim builds dynamic predictions by combining the data with foraging arena assumptions; Ecospace is a spatial and temporal dynamic module primarily designed for exploring impact and placement of protected areas (BSRP 2004). Jointly, the modules are used to describe ecosystem resources and their interactions; evaluate ecosystem effects of fishing (including indirect effects, such as through habitat modification); evaluate the effects of environmental changes; predict and verify bioaccumulation-patterns of persistent pollutants; evaluate the impact and placement of marine protected areas; evaluate uncertainty in the management process; and to explore management policy options incorporating economic, social, legal, and ecological considerations (Bundy 2001; BSRP 2004; Christensen and Walters 2004).

In 2007, the Ecopath modelling approach was recognized as one of the top 10 breakthroughs in marine science by NOAA, in a special web site celebrating 200 years of science, service, and stewardship http://www.celebrating200years.noaa.gov. NOAA recognized Ecopath modelling as the first to apply a type of statistics called "path analysis" to the field of marine ecology. The model simplicity and its ability to accurately identify ecological relationships, "have revolutionized scientists' ability worldwide to understand complex marine ecosystems" (NOAA 2007). Ecopath has over 3000 registered users in 124 countries with more than 150 published models (Christensen and Walters 2004) applied to 41 different ecosystems (Delos 1995). According to Google Scholar, the use of the term "Ecopath" increased from 17 publications in the 1980s to 370 in the 1990s, to 968 in the 2000s (Morissette 2007). Most of these studies use Ecopath to characterize a single ecosystem (e.g. Baird and Ulanowicz 1989). Others use it as a tool for comparing ecosystems (e.g. Baird and Ulanowicz 1993; Christian et al. 2005), and a few use it to evaluate the magnitude of stress imposed on a system (e.g. Baird and Heymans 1996).

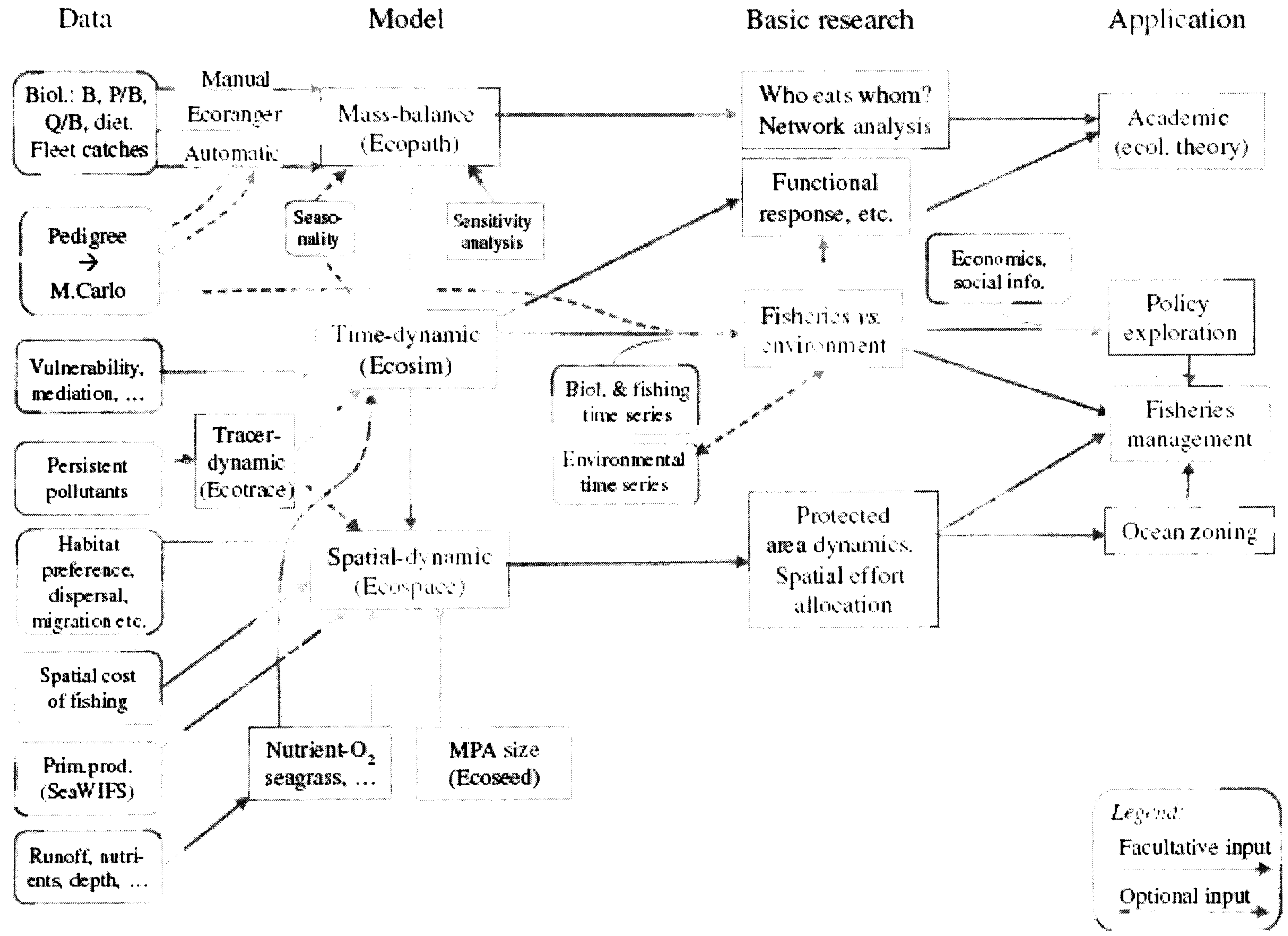


Figure 4.1 Overview of the modules and data types for Ecopath with Ecosim (EwE) modelling (BSRP 2004).

Ecopath data requirements are relatively simple, and generally already available from stock assessment, ecological studies, or the literature: biomass estimates, total mortality estimates, consumption estimates, diet compositions, and fishery catches (Christensen et al. 2005). The data requirements of an Ecopath model are expressed by its master equation. The basic condition is that input to each group is equal to the output from it (equilibrium conditions). Then, a series of biomass budget equations are determined for each group as:

Production = fisheries catch + predation mortality + biomass accumulation + net migration + other mortality

In addition the groups in the system are linked through predators consuming prey. Such consumption can be described by

Consumption = Production + non-assimilated food + respiration

The terms of this equation may be replaced by:

Production by i = $B_i \times P/B_i$,

Predator losses of i = $\sum_j (B_j \times Q/B_j \times DC_{ji})$, and

Other losses of i = $(1 - EE_i) \times B_i \times P/B_i$

Whereby:

Predation mortality is the factor that links the different functional groups in an ecosystem. The network flows of biomass within an ecosystem link the plants with the herbivores, and the latter with the carnivores and predators. The linkages are commonly represented as a food web and the position of each functional group within the food web is identified as its trophic level.

The equation developed by Polovina (1984) can be presented as follow:

$$P_i = Y_i + B_i.M2_i + BA_i + EX_i + P_i.(1 - EE_i) \qquad \text{eq. 4.1}$$

Where:

i is the component (stock, species, group of species) of the model,

j is any of predators of i,

B_i is the biomass for species or group (i),

P_i is the total production rate of i,

Y_i is the total fishery catch rate of i,

$M2_i$ is the total predation rate for i,

BA_i is the biomass accumulation,

EX_i is the export out of the system (migration or fisheries catches) for species or group i,

EE_i is the ecotrophic efficiency, i.e. the proportion of the ecological production which is consumed by predators and usually assumed to range from 0.7 to 0.99 (Polovina 1984)

$P_i.(1- EE_i)$ is the other loss rate

To incorporate most of the production components in the form of predation or mortality, equation 1 can be re-expressed as:

$$B_i . (P/B)_i - \sum_j (B_j . Q/B_j . DC_{ji}) - (P/B)_i (1- EE_i) - EX_i - BA_i - Y_i = 0 \qquad \text{eq. 4.2}$$

Where:

B_j is the biomass of predator (j),

P/B_i is the production/biomass ratio, usually assumed equal to the total mortality (Z_i),

Q/B_j is the consumption per unit of biomass for predator j,

DC_{ji} is the fraction of prey (i) in the average diet of predator (j),

Therefore, a system with n groups (boxes) will have n linear equations. Since Ecopath links the different groups, it allows the estimation of one unknown parameter for each group. Required inputs for creating an Ecopath model are three of four following parameters: B_i, $(P/B)_i$, $(Q/B)_j$ and EE_i. although it is recommended that B_i, $(P/B)_i$ and

$(Q/B)_j$ are specified (Christensen et al. 2005). Once three parameters are entered for each group, a diet composition matrix is constructed. The diet matrix is constructed by calculating the percent of each prey that occurs in each predator's diet. The Ecopath model then is checked for steady-state conditions. The element of the diet matrix or the values of the three inputted parameters are adjusted until the EE_i for each group is between zero and one. A value of ecotrophic efficiency less than zero would imply that $P_i.(1- EE_i)$ must be greater than P, which according to equation (1) would require one of the other terms to be negative. A value of ecotrophic efficiency greater than one would require P to be negative. The data required for Ecopath are assembled and standardized to tonnes/km^2 and tonnes/km^2/year.

In most cases, the model does not balance initially due to uncertainties in model parameters. In this case, the value of one or more of the terms can be changed iteratively until a balance is obtained. Indeed, there is more than one way to construct an Ecopath model and there is no unique solution to any model. However, if uncertainty associated with specific input parameters is low, then the number of plausible solutions is reduced. For the less certain parameters, sensitivity analyses can be used to examine their impacts on the model outputs (Morissette 2007).

Once a balanced Ecopath model is obtained, the flows of biomass among the groups and higher-order indices of ecosystem functioning can be interpreted. Ecopath provides the flows of biomass among groups that satisfies the steady–state condition, and that are also consistent with the inputted values of production (P), biomass (B), and consumption (Q). Several network analysis indices are also produced by Ecopath, which are useful for determining an ecosystem's structure, maturity, and stability (Odum 1969; Ulanowicz 1980).

4.5 Overview of application of Ecopath with Ecosim (EwE) to fisheries management in Thailand

There have been a few studies in the wider Gulf of Thailand using Ecopath and Ecopath with Ecosim during the last 1990s. These studies included both freshwater and marine ecosystems.

Pauly and Christensen (1993) constructed two preliminary Ecopath models of the Gulf of Thailand, one covering the 0-10 m depth zone and another covering the 10-50 m depth zone. These models were based mainly on catch statistics data from FAO and they did not incorporate fisheries information from research cruises carried out in the Gulf of Thailand. Subsequently, Christensen (1998) constructed two mass-balance trophic models based on information from the research vessel cruises. One of these described the initial phase of fisheries development in 1963 (10-50 m depth zone) and the other the phase of severe depletion of the early-1980s, when the demersal stocks were heavily exploited. Christensen further used the dynamic simulation model Ecosim to study if the changes in catch composition and abundance over the time period could be explained by the impact of the fisheries, concluding that this was likely.

Chookajorn et al. (1994) studied the evolution of trophic relationships in Ubolratana Reservoir (Thailand) using a multispecies trophic model and Jutagate et al.(2002) studied the freshwater ecosystems in Sirinthorn (Thailand) and Nam Hgum (Laos) Reservoirs. The output from Ecopath model indicated similar ecosystems in both reservoirs. Both man-made reservoirs were productive, with the zooplankton-eating fish being the target species of fishing operations.

Supongpan et al. (2000) reported on the use of ecosystem models to investigate multispecies management strategies for capture fisheries in the Gulf of Thailand. EwE was used to simulate both open and closed loop policies to maximize the economic, social sustainability and ecosystem stability. The results of the open loop simulation showed the optimum fishing efforts over time to get the best economic profit required reducing the efforts of pair trawlers by about 20% and beam trawl and push net effort by 50 % compared to the present. Otter board trawl, purse seine and other gears should be reduced by about 40 % to 10 % and 90 %, respectively, to achieve balance within the whole fisheries and to get the best profit.

Vibunpant et al. (2003) made the most extensive use of the available long time series of data on catch and effort including economic information. Changes in the relative effort for each of six fleets considered during the period 1973 to 1993 were used to drive the EwE model over this time period. The results indicated that a complete ban on push net fishing would have minor effects on biomass, catches and profits, perhaps reflecting the overall low catch level by the push net fleet. Avoiding the capture of juveniles by banning all small mesh sizes led to a marked decrease in overall catch level, while the value of the catch only decreased marginally. The reduced catches of small fish does not lead to any marked improvement in the state of the overall system, indicating that such a measure would be inadequate for changing the gross overfishing in the Gulf of Thailand.

4.6 Research questions

In this chapter, I use Ecopath with Ecosim to address the following questions:

a) How healthy is the Mae Klong estuary fish community, as reflected in the ecosystem-level metrics based on Odum's conjectures?
b) What is the extent of seasonal variation in the food web and hence these ecosystem health metrics?
c) What are the potential and limitations of mass-balance models for evaluating the health of ecosystems like the Mae Klong?

4.7 Materials and methods

This chapter introduces the principle of the Ecopath model as applied to mangrove-fisheries in the study area. For the purpose of my study, the Ecopath with Ecosim (EwE) versions 5 and 6 (http://www.ecopath.org) were used. In my application, the data from Chapter 3 on the Mae Klong Estuary were used to construct three season-specific Ecopath models (dry, hot and rainy).

Knowledge of prey-predator relationships within the these versions of the food webs for all major species or aggregate species group in the ecosystem are required for an Ecopath model, and information was not available for some groups in the present study. In such cases, I referred to Fishbase (http://www.fishbase.org), a biological database developed at the International Centre for Living Aquatic Resources Management (ICLARM), in collaboration with FAO and other organizations.

4.7.1 Biomass estimation

The biomasss of a fish species (or group of fish species) was assumed to be constant for the period covered by the model. This parameter is expressed in tonnes wet weight per km^2.

CPUE (catch per unit effort) values were used to estimate biomass. The biomass of fish from the study (see Chapter 3) was estimated using the swept area method (Sparre and Venema 1992) as follows:

$$B = \frac{CPUE}{a \cdot X_1} \cdot A$$

Where,

A (total area) = 15.9 km^2

a (swept area) = 0.11112 km^2

X_1 (proportion of fishes in the path of the trawl retained by it) = 0.5

The swept area was estimated from the equation:

$$a = t \cdot v \cdot h \cdot X_2$$

Where,

t (time spent trawling) = 6 hrs

v (trawling speed) = 1 knot

(multiplied by 1.852 to convert to $km.hr^{-1}$)

h (length of trawl head rope) = 20 m

X_2 (effective width of the trawl relative to its head rope) = 0.5

4.7.2 Diet

The diet matrix data for each functional group was constructed from field data whenever possible. However, for a few species these data were not available and for such species diet data were obtained mostly from literature reports for species or species groups in the similar area (Table 4.3) and with the help of FishBase (Froese and Pauly 2004 [http://www.fishbase.org]). Imports were not included in the matrix due to the lack of information on net migration rate for most of the species.

4.7.3 Defining functional groups

There are many species in ecosystems, which can make functional group division difficult. The state variables selected for the food web in the present study (Table 4.4) were based on the following criteria:

- Ecological or taxonomic related species
- Typical and abundant species
- Species of economic and social importance
- Species for which there are historical data and information

On the basis of the above criteria, 21 functional groups were selected. Most groups represent the most important trophic links of this system (Vibunpant et al. 2003). Only those of particular interest remained as an individual group; such as the commercially important shrimp, crab, sardine, anchovy, catfish and threadfin. Nekton and sergestid shrimp are separated from zooplankton as a discrete group. Additionally, some fish groups were divided into pelagics (4 groups), benthopelagics (9 groups) and benthics (14 groups).

4.7.4 Strategy for model balancing

The first Ecopath Eq. 4.1 states that each group must be mass-balanced, i.e., catches, consumption, biomass accumulation and export do not exceed production for a group. Therefore, balancing the model requires adjustment of the input parameters so that ecotrophic efficiencies (EE) do not exceed 1. This manual procedure relies on knowledge to decide which adjustments have to be done (Kavanagh et al. 2004), and must be rigorously applied according to realistic hypotheses. If $EE > 1$, this indicates predation on that compartment is greater than production by the compartment. If EE <

Table 4.3 EwE Model inputs and sources for groups in Mae Klong Estuary.

Group	Input	Source	Group	Input	Source
Rays	B	From this study	Pelagics	B	From this study
	P/B	Vibunpant et al. (2003)		P/B	Vibunpant et al. (2003)
	Q/B	N/A		Q/B	N/A
	Diet	From this study		Diet	From this study
Anchovies	B	From this study	Benthopelagics	B	From this study
	P/B	Garces et al. (2003)		P/B	Garces et al. (2003)
	Q/B	Garces et al. (2003)		Q/B	Garces et al. (2003)
	Diet	From this study		Diet	From this study
Sardines	B	From this study	Benthics	B	From this study
	P/B	Garces et al. (2003)		P/B	Vibunpant et al. (2003)
	Q/B	Garces et al. (2003)		Q/B	N/A
	Diet	From this study		Diet	From this study
Catfishes	B	From this study	Nekton	B	N/A
	P/B	Vibunpant et al. (2003)		P/B	Vibunpant et al. (2003)
	Q/B	N/A		Q/B	N/A
	Diet	From this study		Diet	Vibunpant et al. (2003)
Mullets	B	From this study	Sergestid shrimps	B	N/A
	P/B	Garces et al. (2003)		P/B	Vibunpant et al. (2003)
	Q/B	Garces et al. (2003)		Q/B	N/A
	Diet	From this study		Diet	Vibunpant et al. (2003)
Perchets	B	From this study	Shrimps	B	N/A
	P/B	Garces et al. (2003)		P/B	Vibunpant et al. (2003)
	Q/B	Garces et al. (2003)		Q/B	N/A
	Diet	From this study		Diet	Vibunpant et al. (2003)
Ponyfishes	B	From this study	Crabs	B	N/A
	P/B	Vibunpant et al. (2003)		P/B	Vibunpant et al. (2003)
	Q/B	N/A		Q/B	N/A
	Diet	From this study		Diet	Vibunpant et al. (2003)
Threadfin	B	From this study	Benthic invertebrates	B	N/A
	P/B	Garces et al. (2003)		P/B	Vibunpant et al. (2003)
	Q/B	Garces et al. (2003)		Q/B	N/A
	Diet	From this study		Diet	Vibunpant et al. (2003)
Croakers	B	From this study	Zooplankton	B	N/A
	P/B	Vibunpant et al. (2003)		P/B	Vibunpant et al. (2003)
				Q/B	Vibunpant et al. (2003)
				Diet	Vibunpant et al. (2003)
	Q/B	N/A	Phytoplankton	B	Vibunpant et al. (2003)
	Diet	From this study		P/B	Vibunpant et al. (2003)
Spotted scat	B	From this study	Detritus	B	Vibunpant et al. (2003)
	P/B	Garces et al. (2003)			
	Q/B	Garces et al. (2003)			
	Diet	From this study			

Table 4.4 Composition of ecological groups used for EwE modelling of the Mae Klong Estuary .

Ecological groups	**Taxa**
Rays	Dasyatidae : *Dasyatis fluviorum* *Himantura imbricata*
Anchovies	Engraulidae: *Setipinna taty, Stolephorus commersonii, Thryssa hamiltoni, T. setrirostris*
Sardines	Clupeidae: *Anodontostoma chacunda ,Escualosa elongata, E. thoracata, Herklotsichthys dlispilonotus, Hilsa kelee, Sardinella albella, S. fimbriata, S. gibbosa, S. lemuru*
Catfishes	Ariidae: *Arius caelatus, A. sagor, Cryptarius truncatus, Ketengus typus, Osteogeneiosus militaris* Plotosidae : *Plotosus canius*
Mullets	Mugillidae : *Chelon tade, Liza subviridis, Moolgarda seheli*
Perchlets	Ambassidae: *Ambassis gymnocephalus, A. nalua*
Ponyfishes	Leiognathidae: *Leiognathus decorus, Secutor insidator, S. ruconius*
Threadfin	Polynemidae: *Eleuthronema tetradactylum*
Croakers	Sciaenidae: *Aspericorvina jubata, Chrysochir aureus, Dendrophysa russelli, Nibea albiflora, Panna microdon*
Spotted scat	Scatophagidae: *Scatophagus argus*
Pelagics	Atherinidae: *Hypoatherina valenciennei* Belonidae: *Strongylura strongylura* Hemirhamphidae: *Hyporhamphus (Hyporhamphus) limbatus, Rhynchorhamphus naga*
Benthopelagics	Elopidae: *Elops machnata* Sillaginidae: *Sillago sihama* Carangidae: *Alepes djedaba, Scomberoides commersonnianus, S. tol, Selaroides leptolepis* Gerreidae: *Gerres erythrourus* Drepanidae: *Drepane punctata* Teraponidae: *Terapon theraps*
Benthics	Ophichthidae : *Pisodonophis boro* Eleotridae: *Butis butis, Ophiocara porocephala* Gobiidae: *Acentrogobius caninus, A. chlorostigmatoides, Aulopareia cyanomos, Bathygobius fuscus, Favonigobius aliciae, Pseudapocrptes lanceolatus* Cynoglossidae : *Cynoglossus lingue, C. puncticeps* Triacanthidae : *Triacanthus biaculeatus* Tetraodontidae : *Lagocephalus lunaris* Uranoscopidae: *Uranoscopus bicinctus*
Nekton	Juvenile fishes
Sergestid shrimps	Sergestidae: *Acetes* spp.
Shrimps	Includes all juvenile and adult shrimp of *Alpheus* spp., *Penaeus* spp. and *Metapenaeus* spp.
Crabs	Portunidae, Scyllaridae, Ocypodidae
Benthic invertebrates	Polychaetes, Bivalves, Gastropods, Barnacle, Sipunculid, Eunicid, Bryozoa, Nematode, Trematode, Nemertean
Zooplankton	Zoea of crab, Megalopa, Mysids, Amphipod, Copepod, Isopods, Ostracods, Cladocera (Daphnia), Cumacean, Euphausid, Tintinnid, Lucifer larva, Bivalve larvae, Cirripedia larvae, Stomatopod larvae, Bivalve larvae, Planktonic foraminiferans, Fish eggs
Phytoplankton& benthic producers	Dominated by diatoms: *Actinocyclus, Amphipleura, Amphora, Anabaena, Anomocnema, Asterionella, Asteronellopsis, Bacillaria,Ceratium,Closterium, Cocconeis, Coscinodiscus, Cyclotella, Cymbella, Diploneis, Ditylum, Epithemia,*

	Eucampid, Eutonia, Fragilaria, Grammetophora, Gyrosigma, Lauderia, Mastogloia, Navicula, Nitzschia, Odontella, Oscillatoria, Phagus, Pinnularia, Pleurosigma, Pseudonitzschia, Rhizosolenia, Rhopaladia, Scenedesmus, Skeletonema, Surirella, Thalassiosira, Thalassiothrix, Thalassionema, Urotrix, and dinoflagellates*(Dinophysis)* Marine algae
Detritus	Particulate and dissolved organic matter

1 for a group, this indicates an excess of biomass at the end of the considered period (12 months in my case), that may accumulate in the system, migrates out the system, or is lost due to other mortality. The model represents an average annual situation so I assumed no fishery harvest (Y=0), and no accumulation of biomass of any groups within each season (BA=0). Although fluxes of water coming into the estuary are unknown, the water circulation is expected to export living or detrital matter out of the estuary. Therefore, a group with a low EE was expected to lose biomass through export via the water fluxes passing through the estuary (Marie-Bozec et al. 2004).

I also assume no significant inter-annual differences. This is a common and simplifying assumption done in order to allow the modelling of complex systems (Christensen and Walters 2004). I applied the following strategy to achieve mass balance for all groups. First, adjustments of diets were given priority since feeding habits of some groups are highly variable and mainly dependent on which food sources are available in the ecosystem. Second, I gave preference to the adjustments of parameters that were not estimated in the field.

4.8 Results

After balancing the model, various indicators based on trophic flow description, thermodynamic concepts, information theory and network analysis were derived (Christensen et al. 2005).

The balanced parameter estimates of the Mae Klong Estuary food web of each seasons are shown in Tables 4.5-4.7, whereas the diet matrices are displayed in Tables 4.8-4.10. These parameter estimates include trophic level, biomass estimates, production/biomass estimates, consumption/biomass estimates, ecotrophic efficiencies and production to consumption ratios. The biomass, production and Ecopath-derived biomass flow among groups were compared for the three seasons. Trophic level and flow of each group, system indices and network characteristics were compared among seasons.

4.8.1 Model sensitivity

Pedigree indices of 0.321 were obtained for the models of the three seasons, a measure of the model quality. This ranks well within values from 50 previously constructed models where pedigree values ranged between 0.164 and 0.676 (Morisette 2007), and which reflects the overall good quality of an Ecopath model as discussed by Christensen et al. (2005).

Once the models were balanced, the Ecoranger routine (Pauly et al.2000) was then used for each model in order to obtain the 'best-fitting' model. A number of acceptable runs (200/10000) were obtained with deviations of 1.644, 1.624 and 1.648 in dry, hot and rainy seasonal models, respectively. These values indicate that the three models were tightly fitted; the initial inputs and outputs based on field data were very close to the mean values generated by Ecoranger. Ratios of respiration to assimilation (R/A), production to respiration (P/R) (Tables 4.12-4.14) and estimated EE*s* for all considered groups are less than 1.

Table 4.5 Input and parameters estimates by Ecopath (in brackets) for the Mae Klong Estuary in the dry season, 2005.

Group name	Trophic level	Biomass in habitat area (t/km^2)	Production /biomass (/year)	Consumption /biomass (/year)	Ecotrophic efficiency	Production/ consumption
1 Rays	3.17	0.221	0.500	(2.500)	(0.357)	0.200
2 Anchovies	3.00	0.143	2.700	7.900	(0.902)	(0.342)
3 Sardines	2.93	0.036	2.700	7.900	(0.900)	(0.342)
4 Catfishes	3.09	1.357	2.000	10.000	(0.689)	0.200
5 Mullets	2.21	1.060	0.430	10.750	(0.793)	(0.040)
6 Perchets	2.73	0.041	2.150	10.750	(0.893)	(0.200)
7 Ponyfishes	3.06	0.070	3.500	(14.000)	(0.944)	0.250
8 Threadfin	2.98	0.239	1.740	8.700	(0.841)	(0.200)
9 Croakers	3.12	0.552	1.500	(7.500)	(0.950)	0.200
10 Spotted scat	3.00	0.035	2.150	10.750	(0.794)	(0.200)
11 Pelagics	3.24	0.769	3.000	(12.000)	(0.823)	0.250
12 Benthopelagics	3.14	0.076	2.150	10.750	(0.831)	(0.200)
13 Benthics	2.44	0.245	3.000	(12.000)	(0.925)	0.250
14 Nekton	2.50	(0.312)	4.000	(16.000)	0.950	0.250
15 Shrimps	2.00	(0.520)	10.000	(16.000)	0.950	0.250
16 Sergedtid shrimps	2.00	(1.956)	5.000	(20.00)	0.950	0.250
17 Crabs	2.61	(1.129)	3.000	(12.000)	0.950	0.250
18 Benthic invertebrates	2.22	(14.751)	5.000	(25.000)	0.650	0.200
19 Zooplankton	2.00	(5.543)	40.000	280.000	0.200	(0.143)
20 Phytoplankton	1.00	(18.286)	200.000		0.440	
21 Detritus	1.00	10000.000			(0.111)	

Table 4.6 Input and parameters estimates by Ecopath (in brackets) for the Mae Klong Estuary in the hot season, 2006.

	Group name	Trophic level	Biomass in habitat area (t/km^2)	Production /biomass (/year)	Consumption /biomass (/year)	Ecotrophic efficiency	Production/ consumption
1	Rays	3.24	0.118	0.500	(2.500)	(0.254)	0.200
2	Anchovies	3.03	0.201	2.700	7.900	(0.891)	(0.342)
3	Sardines	3.01	0.072	2.700	7.900	(0.877)	(0.342)
4	Catfishes	3.26	2.094	2.000	(10.000)	(0.709)	0.200
5	Mullets	2.15	1.271	0.430	10.750	(0.864)	(0.040)
6	Perchets	3.02	0.291	2.150	10.750	(0.894)	(0.200)
7	Ponyfishes	3.15	0.288	3.500	(14.000)	(0.868)	0.250
8	Threadfin	3.04	0.754	1.740	8.700	(0.845)	(0.200)
9	Croakers	3.03	0.435	1.500	(7.500)	(0.866)	0.200
10	Spotted scat	2.86	0.171	2.150	10.750	(0.884)	(0.200)
11	Pelagics	3.15	1.069	3.000	(1.000)	(0.877)	0.250
12	Benthopelagics	3.12	0.157	2.150	10.750	(0.945)	0.200
13	Benthics	3.00	0.070	3.000	(12.000)	(0.906)	0.250
14	Nekton	2.50	(3.795)	4.000	(16.000)	0.950	0.250
15	Shrimps	2.00	(1.316)	10.000	(40.000)	0.950	0.250
16	Sergestid shrimps	2.00	(1.190)	5.000	(20.000)	0.950	0.250
17	Crabs	2.61	(0.951)	3.000	(12.000)	0.950	0.250
18	Benthic invertebrates	2.22	(19.333)	5.000	(25.000)	0.650	0.200
19	Zooplankton	2.00	(11.496)	40.000	280.000	0.200	(0.143)
20	Phytoplankton	1.00	(38.114)	200.000	-	0.440	-
21	Detritus	1.00	10000.000	-	-	(0.069)	-

Table 4.7 Input and parameters estimates by Ecopath (in brackets) for the Mae Klong Estuary in the rainy season, 2006.

	Group name	Trophic level	Biomass in habitat area (t/km^2)	Production /biomass (/year)	Consumption /biomass (/year)	Ecotrophic efficiency	Production/ consumption
1	Rays	3.05	0.081	0.500	(2.500)	(0.315)	0.200
2	Anchovies	3.04	0.269	2.700	7.900	(0.853)	(0.342)
3	Sardines	3.10	0.192	2.700	7.900	(0.870)	(0.342)
4	Catfishes	3.27	4.224	2.000	(10.000)	(0.745)	0.200
5	Mullets	2.23	2.743	0.430	10.750	(0.880)	(0.040)
6	Perchets	3.17	0.594	2.150	10.750	(0.894)	(0.200)
7	Ponyfishes	3.13	0.166	3.500	(14.000)	(0.871)	0.250
8	Threadfin	3.04	0.694	1.740	8.700	(0.919)	(0.200)
9	Croakers	3.07	0.517	1.500	(7.500)	(0.866)	0.200
10	Spotted scat	2.72	1.135	2.150	10.750	(0.870)	(0.200)
11	Pelagics	3.32	1.514	3.000	(12.000)	(0.802)	0.250
12	Benthopelagics	3.24	1.161	2.150	10.750	(0.851)	(0.200)
13	Benthics	3.07	0.038	3.000	(12.000)	(0.799)	0.250
14	Nekton	2.50	(5.886)	4.000	(16.000)	0.950	0.250
15	Shrimps	2.00	(0.921)	10.000	(40.000)	0.950	0.250
16	Sergestid shrimps	2.00	(1.604)	5.000	(20.000)	0.950	0.250
17	Crabs	2.61	(1.610)	3.000	(12.000)	0.950	0.250
18	Benthic invertebrates	2.22	(85.598)	5.000	(25.000)	0.650	0.200
19	Zooplankton	2.00	(33.841)	40.000	280.000	0.200	(0.287)
20	Phytoplankton	1.00	(113.292)	200.000	-	0.440	-
21	Detritus	1.00	10000.000	-	-	(0.084)	-

Table 4.8 Diet matrix of the Mae Klong Estuary in the dry season, 2005.

Prey\ Predator	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19
Rays																			
Anchovies																			
Sardines																			
Catfishes																			
Mullets																			
Perchets																			
Ponyfishes																			
Threadfin																			
Croakers																			
Spotted scat																			
Pelagics																			
Benthopelagics																			
Benthics											0.103								
Nekton		0.001		0.038					0.017		0.396	0.160	0.056						
Shrimps	0.05	0.023	0.289	0.483		0.250	0.060	0.160	0.295	1.000	0.082	0.029	0.141						
Sergestid shrimps	0.435	0.876			0.007	0.122		0.776	0.303		0.259	0.212	0.113						
Crabs	0.285			0.169					0.164		0.002		0.029						
Benthic invertebrates		0.017	0.125	0.129	0.141	0.004	0.273	0.007	0.084		0.020	0.254	0.009				0.500	0.100	
Zooplankton	0.230	0.082	0.486	0.122	0.030	0.356	0.667	0.032	0.130		0.077	0.346	0.046	0.500				0.100	
Phytoplankton			0.050		0.539				0.007					0.500				0.200	1.000
Detritus		0.001	0.050	0.059	0.283	0.268		0.025			0.061		0.606		1.000	1.000	0.500	0.600	
Import																			
Sum	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000

Table 4.9 Diet matrix of the Mae Klong Estuary in the hot season, 2006.

Prey\ Predator	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19
Rays																			
Anchovies																			
Sardines																			
Catfishes																			
Mullets																			
Perchets																			
Ponyfishes																			
Threadfin																			
Croakers																			
Spotted scat																			
Pelagics																			
Benthopelagics																			
Benthics																			
Nekton		0.031		0.346	0.016	0.028		0.052	0.060		0.458	0.243							
Shrimps		0.057		0.060		0.062	0.002	0.225	0.110		0.121	0.094	0.667						
Sergestid shrimps	0.392	0.461	0.245	0.119	0.001	0.707	0.007	0.370	0.349		0.167	0.465	0.333						
Crabs	0.392		0.036	0.120					0.002		0.002	0.018							
Benthic invertebrates		0.063	0.127	0.138	0.073	0.013	0.696	0.074	0.011	0.050	0.081	0.138					0.500	0.100	
Zooplankton	0.217	0.388	0.547	0.201	0.039	0.190	0.295	0.279	0.468	0.800	0.072			0.500				0.100	
Phytoplankton			0.010		0.583					0.150				0.500				0.200	1.000
Detritus			0.035	0.016	0.288						0.099	0.042			1.000	1.000	0.500	0.600	
Import																			
Sum	1.001	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000

Table 4.10 Diet matrix of the Mae Klong Estuary in the rainy season, 2006.

Prey\ Predator	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19
Rays																			
Anchovies																			
Sardines																			
Catfishes																			
Mullets																			
Perchets																			
Ponyfishes																			
Threadfin																			
Croakers																			
Spotted scat																			
Pelagics																			
Benthopelagics																			
Benthics																			
Nekton		0.006	0.222	0.272		0.002	0.005		0.015		0.553	0.032							
Shrimps	0.317	0.045		0.123		0.007	0.054	0.085	0.291		0.017	0.012							
Sergestid shrimps	0.017	0.613	0.091	0.013		0.120		0.708	0.198		0.052		0.010						
Crabs	0.083	0.004	0.004	0.037				0.012	0.036			0.223							
Benthic invertebrates		0.143	0.329	0.500	0.168	0.772	0.608	0.128	0.328	0.584	0.200	0.67	0.333				0.500	0.100	
Zooplankton	0.583	0.189	0.269	0.055	0.020	0.099	0.324	0.068	0.103	0.009	0.176		0.657	0.500				0.100	
Phytoplankton			0.002		0.477				0.029	0.407				0.500				0.200	1.000
Detritus			0.083		0.335		0.009				0.002	0.063			1.000	1.000	0.500	0.600	
Import																			
Sum	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000

4.8.2 Trophic level and flow

Biomass flows as calculated under steady-state conditions by Ecopath are shown in Figure 4.2. Some biological parameters of biomass, production, consumption, respiration and flow to detritus are shown in Table 4.11.

Figure 4.2 which shows the main biomass flows between functional groups. Detritus and phytoplankton displayed the highest values for biomass and production, while consumption rate and metabolic waste (respiration) were highest for zooplankton, followed by benthic invertebrates. This has to be noted, since they are the main food supply for fish groups and show a strong relation with primary producers, including detritus.

Biomass for most of the groups generally peaked in the rainy season, except for ponyfishes, threadfin, spotted scat and shrimps, which have the highest biomass in the hot season; rays, crockers, benthic fishes and sergestid shrimps have the highest biomass in the dry season. The difference between maximum and minimum biomass was greatest for benthic invertebrates, zooplankton and phytoplankton, with the ratios of maximum and minimum biomass around 6-fold (dry to rainy seasons). Production, consumption, respiration and flow to detritus were also higher with decreasing biomass (Table 4.11).

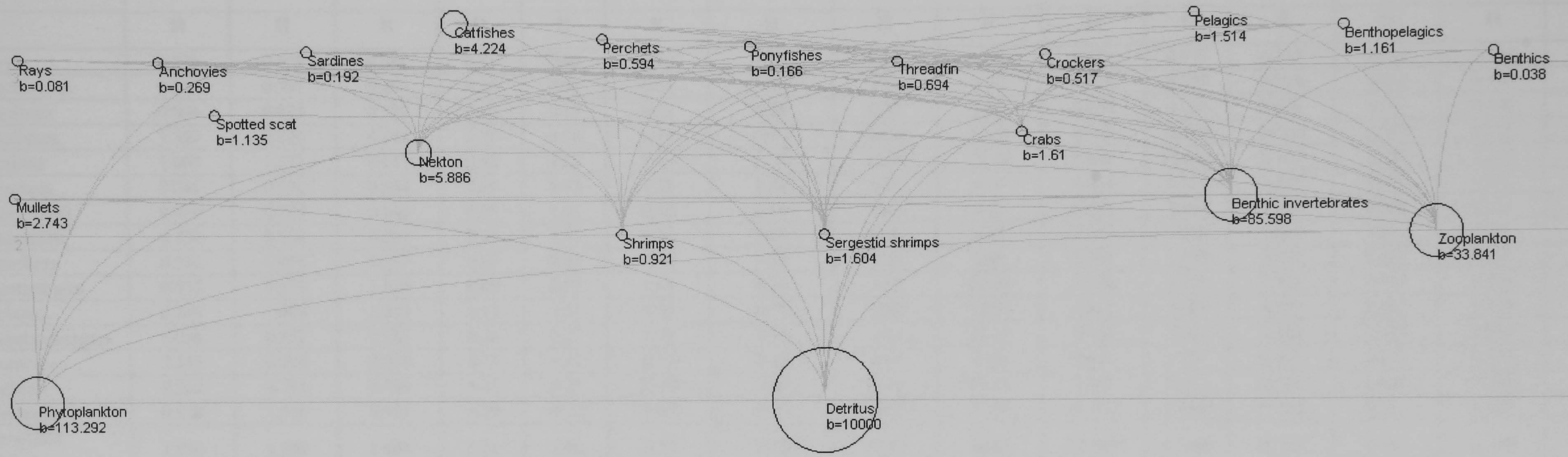
Throughout the study period, most of the fish groups were characterized by small sizes and feeding at low trophic levels (TL). Functional groups were organized within three integer trophic levels (TL) (Tables 4.5-4.7). The groups with TLs between 3.3 and 3.0 were rays, anchovies, sardines, catfishes, perchets, ponyfishes, threadfin, crockers, pelagic fishes, benthopelagic fishes and benthic fishes, in hot and rainy seasons. Sardines, perchets and threadfin had lower TLs than in the dry season. Invertebrates were classified between 2.0 and 2.6 and the lowest, by definition, were the primary producers and detritus groups (TL=1).

Figure 4.2 Flow diagram of Mae Klong Estuary in three seasons; (a) dry, (b) hot, (c) rainy.

(a)


Rays
b=0.118
Anchovies
b=0.201
Spotted scat
b=0.171
Sardines
b=0.072
Catfishes
b=2.094
Perchets
b=0.291
Ponyfishes
b=0.288
Threadfin
b=0.754
Crockers
b=0.435
Pelagics
b=1.069
Benthopelagics
b=0.157
Benthics
b=0.07
Crab
b=0.951
Nekton
b=3.795
Mullets
b=1.271
Shrimps
b=1.316
Sergestid shrimps
b=1.19
Benthic invertebrates
b=19.333
Zooplankton
b=11.496
Phytoplankton
b=38.114
Detritus
b=10000

(b)


Rays
b=0.081
Anchovies
b=0.269
Sardines
b=0.192
Catfishes
b=4.224
Perchets
b=0.594
Ponyfishes
b=0.166
Threadfin
b=0.694
Crockers
b=0.517
Pelagics
b=1.514
Benthopelagics
b=1.161
Benthics
b=0.038
Spotted scat
b=1.135
Nekton
b=5.886
Crabs
b=1.61
Mullets
b=2.743
Shrimps
b=0.921
Sergestid shrimps
b=1.604
Benthic invertebrates
b=85.598
Zooplankton
b=33.841
Phytoplankton
b=113.292
Detritus
b=10000

(c)

Table 4.11 Biological parameters for the Mae Klong Estuary in dry (D), hot (H) and rainy (R) seasons. Parameters for biomass (B), production (P), consumption (C), respiration (Res) and flow to detritus (Fl) are expressed in t/km^2/year.

							Parameters								
Group name		B			P			C			Res			Fl	
	D	H	R	D	H	R	D	H	R	D	H	R	D	H	R
Rays	0.221	0.118	0.081	0.11	0.06	0.04	0.55	0.29	0.20	0.332	0.177	0.122	0.192	0.103	0.070
Anchovies	0.143	0.201	0.269	0.39	0.54	0.73	1.13	1.59	2.13	0.518	0.728	0.974	0.612	0.860	1.151
Sardines	0.036	0.072	0.192	0.10	0.19	0.52	0.28	0.57	1.52	0.130	0.261	0.695	0.154	0.308	0.822
Catfishes	1.357	2.094	4.224	2.71	4.18	8.44	13.57	20.94	42.24	8.142	12.564	25.344	3.582	5.528	11.151
Mullets	1.060	1.271	2.743	0.46	0.55	1.19	11.39	13.66	29.49	8.660	10.384	22.410	2.735	3.279	7.077
Perchets	0.041	0.291	0.594	0.09	0.63	1.28	0.44	3.13	6.39	0.264	1.877	3.831	0.176	1.251	2.554
Ponyfishes	0.070	0.288	0.166	0.25	1.01	0.58	0.98	4.03	2.32	0.539	2.218	1.278	0.208	0.857	0.494
Threadfin	0.239	0.754	0.694	0.42	1.31	1.21	2.08	6.56	6.04	1.248	3.936	3.623	0.832	2.624	2.415
Crockers	0.552	0.435	0.517	0.83	0.65	0.77	4.14	3.26	3.88	2.484	1.958	2.327	0.869	0.685	0.814
Spotted scat	0.035	0.171	1.135	0.08	0.37	2.44	0.38	1.84	12.20	0.226	1.103	7.321	0.151	0.735	4.881
Pelagics	0.769	1.069	1.514	0.15	3.21	4.55	0.77	12.83	18.17	0.461	7.055	9.992	0.161	2.726	3.861
Benthopelagics	0.076	0.157	1.161	0.16	0.37	2.49	0.82	1.69	12.48	0.490	1.103	7.488	0.172	0.354	2.621
Benthics	0.245	0.070	0.038	0.74	3.21	0.11	2.94	0.84	0.46	1.617	7.055	0.251	0.625	0.179	0.097
Nekton	0.312	3.795	5.886	1.25	0.21	23.57	5.00	60.73	94.18	2.747	0.462	51.799	1.062	13.406	20.013
Shrimps	0.520	1.316	0.921	9.79	15.20	8.03	39.12	23.81	32.08	21.515	33.400	17.646	8.313	4.558	6.818
Sergestid shrimps	1.956	1.190	1.604	5.21	5.96	9.22	20.78	52.65	36.85	11.449	13.095	20.267	4.417	11.189	7.831
Crabs	1.129	0.951	1.610	3.39	13.18	4.83	13.54	11.42	19.32	7.449	28.959	10.628	2.878	2.426	4.106
Benthic invertebrates	14.751	19.333	85.598	73.68	2.09	427.56	368.77	483.33	2139.96	221.260	6.279	1283.976	99.567	130.499	577.789
Zooplankton	5.543	11.496	33.841	217.72	62.93	1351.21	1526.76	3218.78	9475.49	1003.298	289.99	6226.752	479.838	1024.59	2978.01
Phytoplankton	18.286	38.114	113.292	-	-	-	-	-	-	-	-	-	2048.07	4322.84	12688.74
Detritus	10000.00	10000.00	10000.00	-	-	-	-	-	-	-	-	-	-	-	-

Table 4.12 Estimates of respiratory flows and respiration assimilation and production respiration ratios of Mae Klong Estuary in the dry season.

Group name	Respiration ($t/km^2/yr$)	Assimilation ($t/km^2/yr$)	Assimilation/ Respiration	Production/ respiration	Respiration/ biomass
Rays	0.332	0.442	0.750	0.333	1.500
Anchovies	0.518	0.904	0.573	0.746	3.620
Sardines	0.130	0.228	0.573	0.746	3.620
Catfishes	8.142	10.856	0.750	0.333	6.000
Mullets	8.660	9.116	0.950	0.053	8.170
Perchets	0.264	0.353	0.750	0.333	6.450
Ponyfishes	0.539	0.784	0.688	0.455	7.700
Threadfin	1.248	1.663	0.750	0.333	5.220
Croakers	2.484	3.312	0.750	0.333	4.500
Spotted scat	0.226	0.301	0.750	0.333	6.450
Pelagics	0.461	0.615	0.750	0.333	0.600
Benthopelagics	0.490	0.654	0.750	0.333	6.450
Benthics	1.617	2.352	0.688	0.455	6.600
Nekton	2.747	3.996	0.688	0.455	8.800
Shrimps	21.515	31.294	0.688	0.455	11.000
Sergestid shrimps	11.449	16.628	0.687	0.455	22.000
Crabs	7.449	10.835	0.688	0.455	6.600
Benthic invertebrates	221.260	295.013	0.750	0.333	15.000
Zooplankton	1003.298	1221.408	0.821	0.217	184.000
Phytoplankton	0.000	-	-	-	-
Detritus	0.000	-	-	-	-

Table 4.13 Estimates of respiratory flows and respiration assimilation and production respiration ratios of Mae Klong Estuary in the hot season.

Group name	Respiration ($t/km^2/yr$)	Assimilation ($t/km^2/yr$)	Assimilation/ Respiration	Production/ respiration	Respiration/ biomass
Rays	0.177	0.236	0.750	0.333	1.500
Anchovies	0.728	1.270	0.573	0.746	3.620
Sardines	0.261	0.455	0.573	0.746	3.620
Catfishes	12.564	16.752	0.750	0.333	6.000
Mullets	10.384	10.931	0.950	0.053	8.170
Perchets	1.877	2.503	0.750	0.333	6.450
Ponyfishes	2.218	3.226	0.688	0.455	7.700
Threadfin	3.936	5.248	0.750	0.333	5.220
Croakers	1.958	2.610	0.750	0.333	4.500
Spotted scat	1.103	1.470	0.750	0.333	6.450
Pelagics	7.055	10.262	0.688	0.455	6.600
Benthopelagics	1.103	1.350	0.750	0.333	6.450
Benthics	7.055	0.672	0.688	0.455	6.600
Nekton	0.462	48.582	0.687	0.455	8.800
Shrimps	33.400	19.048	0.688	0.455	11.000
Sergestid shrimps	13.095	42.122	0.688	0.455	22.000
Crabs	28.959	9.133	0.688	0.455	6.600
Benthic invertebrates	6.279	386.665	0.750	0.333	15.000
Zooplankton	289.999	2575.025	0.821	0.217	184.000
Phytoplankton	0.000	-	-	-	-
Detritus	0.000	-	-	-	-

Table 4.14 Estimates of respiratory flows and respiration assimilation and production respiration ratios of Mae Klong Estuary in the rainy season.

Group name	Respiration ($t/km^2/yr$)	Assimilation ($t/km^2/yr$)	Assimilation/ Respiration	Production/ respiration	Respiration/ biomass
Rays	0.122	0.162	0.750	0.333	1.500
Anchovies	0.974	1.700	0.573	0.746	3.620
Sardines	0.695	1.213	0.573	0.746	3.620
Catfishes	25.344	33.792	0.750	0.333	6.000
Mullets	22.410	23.590	0.950	0.053	8.170
Perchets	3.831	5.108	0.750	0.333	6.450
Ponyfishes	1.278	1.859	0.688	0.455	7.700
Threadfin	3.623	4.830	0.750	0.333	5.220
Croakers	2.327	3.102	0.750	0.333	4.500
Spotted scat	7.321	9.761	0.750	0.333	6.450
Pelagics	9.992	14.534	0.687	0.455	6.600
Benthopelagics	7.488	9.985	0.750	0.333	6.450
Benthics	0.251	0.365	0.688	0.455	6.600
Nekton	51.799	75.344	0.687	0.455	8.800
Shrimps	17.646	25.666	0.688	0.455	11.000
Sergestid shrimps	20.267	29.480	0.687	0.455	22.000
Crabs	10.628	15.459	0.687	0.455	6.600
Benthic invertebrates	1283.976	1711.968	0.750	0.333	15.000
Zooplankton	6226.752	7580.394	0.821	0.217	184.000
Phytoplankton	0.000	-	-	-	-
Detritus	0.000	-	-	-	-

The average trophic level of each group revealed that pelagic fishes occupied their highest trophic level during the dry and rainy seasons (3.24 and 3.32 respectively), while catfishes showed their highest trophic level during the hot season (3.26). There were no changes the TLs of nekton, shrimps, sergestid shrimps, crabs, benthic invertebrates, zooplankton, phytoplankton and detritus across the three seasons.

Tables 4.15-4.17 show the distribution of relative flows by trophic level. Import of biomass was greatest at the trophic level III in all seasons. Most of the flows in trophic level II (detritivores and herbivores) are due to zooplankton (the dominant herbivores in this ecosystem) and shrimps and sergestid shrimps (the dominant detritivores). Flows in trophic level III are attributed to crabs and benthic invertebrates and an array of fish groups. At level IV, flows are dominated by pelagic fishes (in dry and rainy seasons) and benthic fishes (in the hot season) and at level V by top predators such as rays. Since the magnitude of flows at trophic levels greater than the fifth is very low, representing only a small fraction of the flows associated with the top predators, these levels were omitted from further consideration.

Mangrove plays an important role in detritus accumulation due to the large amount of leaf material that is incorporated within the soil. None of the species within the models feed directly on mangrove biomass. This detritus is utilized by several groups in the food web. Phytoplankton also contributes to the productivity of higher trophic levels that are dependent on detritus. Table 4.11 shows the flows to detritus, from primary and secondary trophic levels, representing the main flow of energy in the food web. Particularly important are the flows from phytoplankton, zooplankton and benthic invertebrates, which are 2048.071, 479.838 and 99.567 t/km^2/year in the dry season, 4322.845, 1024.592 and 130.499 t/km^2/year in the hot season and 12688.740, 2978.012 and 577.789 t/km^2/year in the rainy season, respectively.

4.8.3 Structure analysis

Some whole system properties which can be used to assess the status of the ecosystem in terms of maturity (sensu Odum 1969), and for comparisons among ecosystems, are given in Table 4.19.

Table 4.15 Relative flows by trophic levels of Mae Klong Estuary in the dry season.

Group/Trophic level	**I**	**II**	**III**	**IV**	**V**
Rays	0.000	0.000	0.858	0.127	0.016
Anchovies	0.000	0.001	0.997	0.002	0.000
Sardines	0.000	1.000	0.886	0.014	0.000
Catfishes	0.000	0.059	0.823	0.108	0.000
Mullets	0.000	0.822	0.162	0.160	0.000
Perchets	0.000	0.268	0.732	0.000	0.000
Ponyfishes	0.000	0.000	0.970	0.030	0.000
Threadfin	0.000	0.025	0.974	0.001	0.000
Croakers	0.000	0.007	0.893	0.091	0.009
Spotted scat	0.000	0.000	1.000	0.000	0.000
Pelagics	0.000	0.061	0.697	0.237	0.004
Benthopelagics	0.000	0.000	0.892	0.108	0.000
Benthics	0.000	0.060	0.351	0.042	0.002
Nekton	0.000	0.500	0.500	0.000	0.000
Shrimps	0.000	1.000	0.000	0.000	0.000
Sergestid shrimps	0.000	1.000	0.000	0.000	0.000
Crabs	0.000	0.500	0.444	0.056	0.000
Benthic invertebrates	0.000	0.889	0.111	0.000	0.000
Zooplankton	0.000	1.000	0.000	0.000	0.000
Phytoplankton	1.000	0.000	0.000	0.000	0.000
Detritus	1.000	0.000	0.000	0.000	0.000

Table 4.16 Relative flows by trophic levels of Mae Klong Estuary in the hot season.

Group/Trophic level	I	II	III	IV	V
Rays	0.000	0.000	0.804	0.174	0.022
Anchovies	0.000	0.000	0.977	0.023	0.000
Sardines	0.000	0.045	0.923	0.030	0.002
Catfishes	0.000	0.016	0.736	0.242	0.007
Mullets	0.000	0.871	0.113	0.016	0.000
Perchets	0.000	0.000	0.985	0.016	0.000
Ponyfishes	0.000	0.000	0.922	0.078	0.000
Threadfin	0.000	0.000	0.966	0.034	0.000
Croakers	0.000	0.000	0.968	0.032	0.000
Spotted scat	0.000	0.150	0.844	0.006	0.000
Pelagics	0.000	0.099	0.662	0.239	0.000
Benthopelagics	0.000	0.042	0.812	0.145	0.001
Benthics	0.000	0.000	0.667	0.334	0.000
Nekton	0.000	0.500	0.500	0.000	0.000
Shrimps	0.000	1.000	0.000	0.000	0.000
Sergestid shrimps	0.000	1.000	0.000	0.000	0.000
Crabs	0.000	0.500	0.444	0.056	0.000
Benthic invertebrates	0.000	0.888	0.112	0.000	0.000
Zooplankton	0.000	1.000	0.000	0.000	0.000
Phytoplankton	1.000	0.000	0.000	0.000	0.000
Detritus	1.000	0.000	0.000	0.000	0.000

Table 4.17 Relative flows by trophic levels of Mae Klong Estuary in the rainy season.

Group/Trophic level	I	II	III	IV	V
Rays	0.000	0.000	0.959	0.037	0.005
Anchovies	0.000	0.000	0.979	0.021	0.000
Sardines	0.000	0.850	0.765	0.149	0.000
Catfishes	0.000	0.000	0.790	0.208	0.002
Mullets	0.000	0.812	0.169	0.019	0.000
Perchets	0.000	0.000	0.913	0.087	0.001
Ponyfishes	0.000	0.009	0.921	0.070	0.000
Threadfin	0.000	0.000	0.980	0.020	0.001
Croakers	0.000	0.029	0.909	0.060	0.002
Spotted scat	0.000	0.407	0.528	0.065	0.000
Pelagics	0.000	0.002	0.699	0.299	0.000
Benthopelagics	0.000	0.063	0.735	0.190	0.012
Benthics	0.000	0.000	0.963	0.037	0.000
Nekton	0.000	0.500	0.500	0.000	0.000
Shrimps	0.000	1.000	0.000	0.000	0.000
Sergestid shrimps	0.000	1.000	0.000	0.000	0.000
Crabs	0.000	0.500	0.444	0.056	0.000
Benthic invertebrates	0.000	0.889	0.111	0.001	0.000
Zooplankton	0.000	1.000	0.000	0.000	0.000
Phytoplankton	1.000	0.000	0.000	0.000	0.000
Detritus	1.000	0.000	0.000	0.000	0.000

Total system throughput represents the 'size of the entire system in terms of flow', that passes through the system from input to output, and is the transfer of energy between all groups (Ulanowicz 1986), expressed in t/km^2/year. It is estimated as the sum of four components of the flows, i.e., Total consumption + Total export + Total respiration + Total flow to detritus. If the total system throughput is high, it means that the system is capable of growth, implying that it is vigorous and healthy (Costanza et al. 1998). The total system throughput estimated for the Mae Klong Estuary was 8321, 16999 and 50901 t/km^2/year in dry, hot and rainy seasons respectively, which is comparatively high, but is consistent with tropical marine ecosystems with a high turnover.

The system also seems to have become more productive in the rainy season, reflected in the values for 'net primary production' which were 3657, 7622.859 and 22658.46 t/km^2/year in dry, hot and rainy seasons, respectively.

The total primary production/ total respiration ratio is considered by Odum (1971) to be an important index of the 'maturity' of an ecosystem. In the early development stages of a system, production is expected to exceed respiration, leading to a ratio greater than 1. In systems suffering from organic pollution, this ratio is expected to be less than 1. In 'mature' systems, the total primary production/total respiration should approach 1; the energy that is fixed is approximately balanced by the cost of maintenance. The ratio can take any positive value and is dimensionless. The results for the Mae Klong Estuary imply it is in a developing stage with this ratio being greater than 1 (2.829, 3.012 and 2.944 in dry, hot and rainy seasons, respectively). There are only small differences between seasons.

The total primary production/total biomass also reflects the system's maturity. In immature systems, production exceeds respiration and as a consequence one can expect biomass to accumulate over time. This, in turn, will influence the total primary production/total biomass ratio, which may decrease. The total primary production/total biomass ratio behaves like that of individual groups; it has a dimension of per unit time and it can take any positive value. Total primary production/total biomass ratios of Mae Klong Estuary were found to be 77.402, 91.634 and 88.481 in dry, hot and rainy seasons, respectively.

Net system production (or yield) is the difference between total primary production and total respiration. System production will be large in immature systems and close to zero in mature ones (Odum 1969). Systems with large imports may have a negative system production. System production has the same units as the flows from which it is computed, t/km^2/year. Net system production values obtained for Mae Klong Estuary were 2354.458, 5092.194 and 14961.73 t/km^2/year in dry, hot and rainy seasons, respectively.

The system biomass/total throughput ratio can take any positive value (0.006 in dry and 0.005 in hot and rainy seasons in Mae Klong Estuary), and has time as a dimension. The values obtained from the study revealed that biomass ratios for the hot and rainy seasons were lower than for the dry season. A low ratio is the characteristic of an immature system (Odum 1969).

The connectance index (CI) is, for a given food web, the ratio of the number of actual links to the number of theoretically possible links. Feeding on detritus (by detritivores) is included in the count, but the converse links (i.e., detritus 'feeding' on other groups) are disregarded. The number of possible links in a food web is roughly proportional to the number of groups in the system (Nee 1990). Hence, the connectance index can be expected to be correlated with maturity of the system because a food chain structure changes from linear to web-like as a system matures (Odum 1969, 1971). The value of the connectance index is (at least in aquatic ecosystems) largely determined by the level of taxonomic detail used to represent prey groups, and this precludes meaningful inter-system comparisons. The system omnivory index is suggested as an alternative.

The system omnivory index (OI) is a measure of how the feeding interactions are distributed between trophic levels. For the Mae Klong Estuary, system omnivory indices of 0.117, 0.101 and 0.113 were obtained in dry, hot and rainy seasons, respectively. The maximum omnivory index (Table 4.18) was observed for crabs and highly specialized feeding was observed for shrimps and sergestid shrimps. These values were similar for all three seasons. The CI and OI (0.190 and 0.101) in the hot season were lower than in other seasons, suggesting that this season has a more web-like structure.

4.8.4 Network analysis

Ascendency (*A*) measures the structure of an ecosystem in terms of the amount and organization of biomass flow within the system. Based upon Odum's (1969) interpretation of the attributes of ecosystems, more speciation, finer specialization, longer retention, and more cycling within the system indicates that an ecosystem is more mature. Higher ascendancy values indicate that there is an increase in one or more of these properties. The upper limit to ascendancy is the development capacity (*C*) of the ecosystem. System overhead is the difference between capacity and ascendancy. System overhead is the upper limit to how much ascendancy can increase to counteract unexpected perturbations. Higher overhead indicates that a system has a larger amount of energy reserves with which it can react to perturbations, so that the system should be more able to maintain stability when perturbed. Ascendency values of 10361.6, 21399.9, 63152.1 were obtained from the Mae Klong estuary in dry, hot and rainy seasons, respectively (Table 4.20) which are typical values for a coastal or an estuarine ecosystem (see Table 4.21). Overhead and capacity were highest in the rainy season (73233.3 and 136385.5) and lowest in the dry season (13019.5 and 23381.1). This implies that the rainy season food web was the most resistant to perturbation. The lowest ascendency value for the dry season and highest ascendency value for the rainy season implies that the dry season food web was the least developed and the rainy season was the most developed food web.

In all three seasons the ecosystem has a large overhead, suggesting that all should be resilient, reflected in the high values for resilience in Table 4.20. However, there are

Table 4.18 Omnivory index describing the trophic structure of Mae Klong Estuary in the three seasons.

Group name	Omnivory index		
	Dry season	**Hot season**	**Rainy season**
Rays	0.076	0.089	0.028
Anchovies	0.002	0.010	0.009
Sardines	0.101	0.065	0.148
Catfishes	0.130	0.086	0.034
Mullets	0.204	0.162	0.220
Perchets	0.197	0.007	0.009
Ponyfishes	0.010	0.010	0.024
Threadfin	0.025	0.015	0.010
Croakers	0.062	0.015	0.057
Spotted scat	0.000	0.133	0.359
Pelagics	0.146	0.196	0.048
Benthopelagics	0.034	0.102	0.131
Benthics	0.320	0.056	0.011
Nekton	0.250	0.250	0.250
Shrimps	0.000	0.000	0.000
Sergestid shrimps	0.000	0.000	0.000
Crabs	0.373	0.373	0.373
Benthic invertebrates	0.200	0.200	0.200
Zooplankton	0.000	0.000	0.000
Phytoplankton	0.000	0.000	0.000
Detritus	0.201	0.192	0.195

differences among the three seasons for any of the measures in Table 4.20 are small and may not be ecologically significant.

4.8.5 Mixed trophic impact

The impact of direct and indirect interactions (including competition) among components of the system were evaluated using the mixed trophic impact routine (Leontief 1951). This analysis (Figure 4.3) showed the importance of detritus and lower trophic levels (nekton, shrimps, sergestid shrimps, crabs, benthic invertebrates and zooplankton) in the system. These groups have the most pronounced positive impacts on the system through direct and indirect consumption by other groups. Detritus has a positive impact on nearly all groups in the system, emphasizing the importance of detritus as the base of the food web. All groups (except detritus) showed a negative impact on themselves and this may show within-group competition for the same resources (Christensen et al. 2005), while predators showed

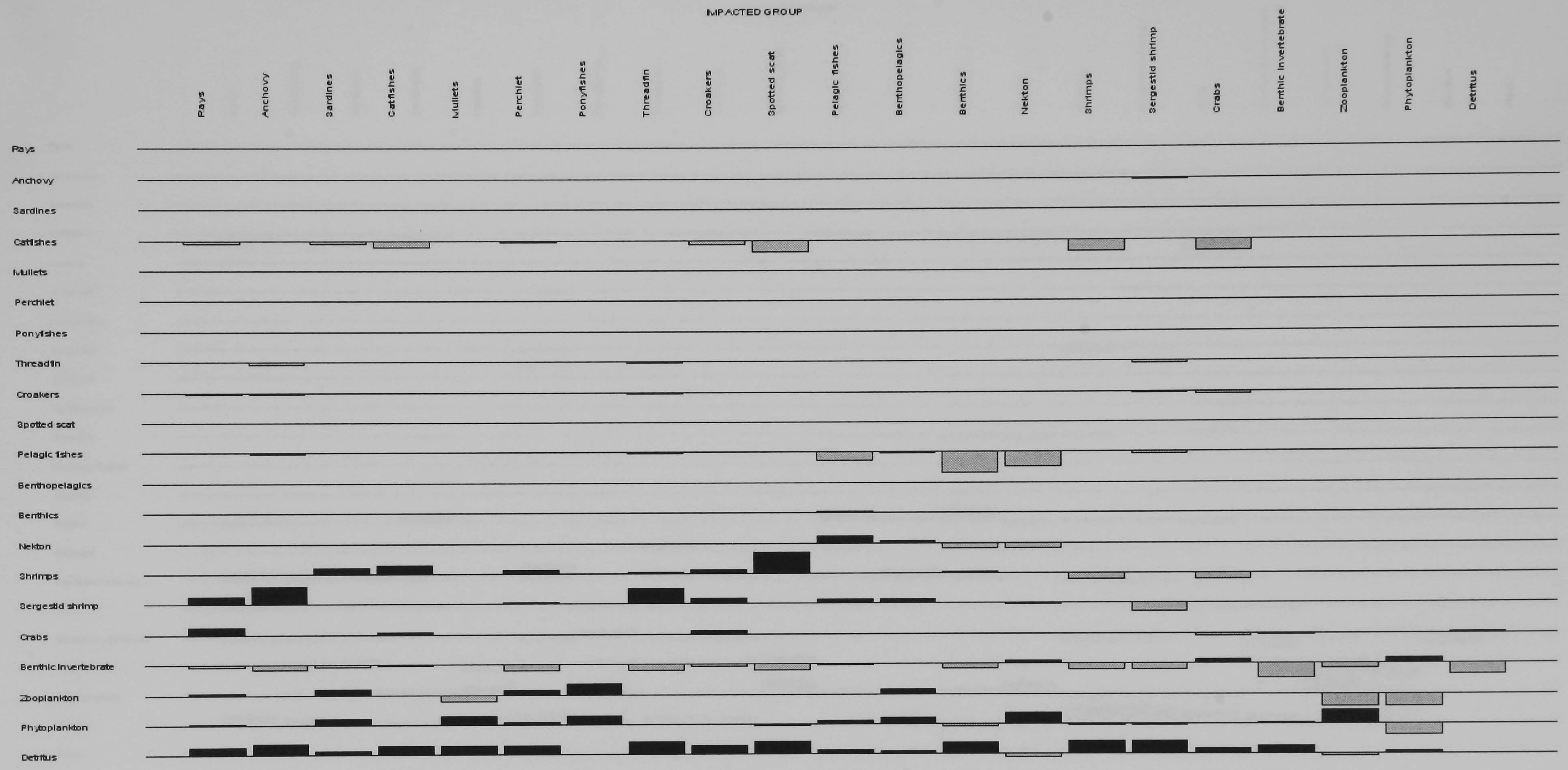
Table 4.19 Global flow parameters for the Mae Klong Estuary in the three seasons.

Parameters	Value			Units
	Dry	Hot	Rainy	
Sum of consumption	2013.439	3921.950	11935.390	t/km^2/year
Sum of all exports	2360.645	5084.186	14947.900	t/km^2/year
Sum of respiratory flows	1292.812	2530.668	7696.720	t/km^2/year
Sum of flows into detritus	2654.615	5461.986	16321.310	t/km^2/year
Total system throughput	8321.511	16999.000	50901.320	t/km^2/year
Sum of all production	3973.000	8230.000	24510.000	t/km^2/year
Calculated total net primary production	3657.000	7622.859	22658.460	t/km^2/year
Total primary production/total respiration	2.829	3.012	2.944	
Net system production	2364.459	5092.194	14961.730	t/km^2/year
Total primary production/total biomass	77.402	91.634	88.481	
Total biomass/total throughput	0.006	0.005	0.005	
Total biomass (excluding detritus)	47.250	83.188	256.082	t/km^2
Connectance index	0.195	0.190	0.193	
System Omnivory Index	0.117	0.101	0.113	
Throughput cycled (excluding detritus)	36.88	48.33	214.00	t/km^2/year
Throughput cycled (including detritus)	135.15	171.00	685.35	t/km^2/year
Total no. of pathways	62	55	55	
Mean length of pathways	2.5	2.91	2.93	

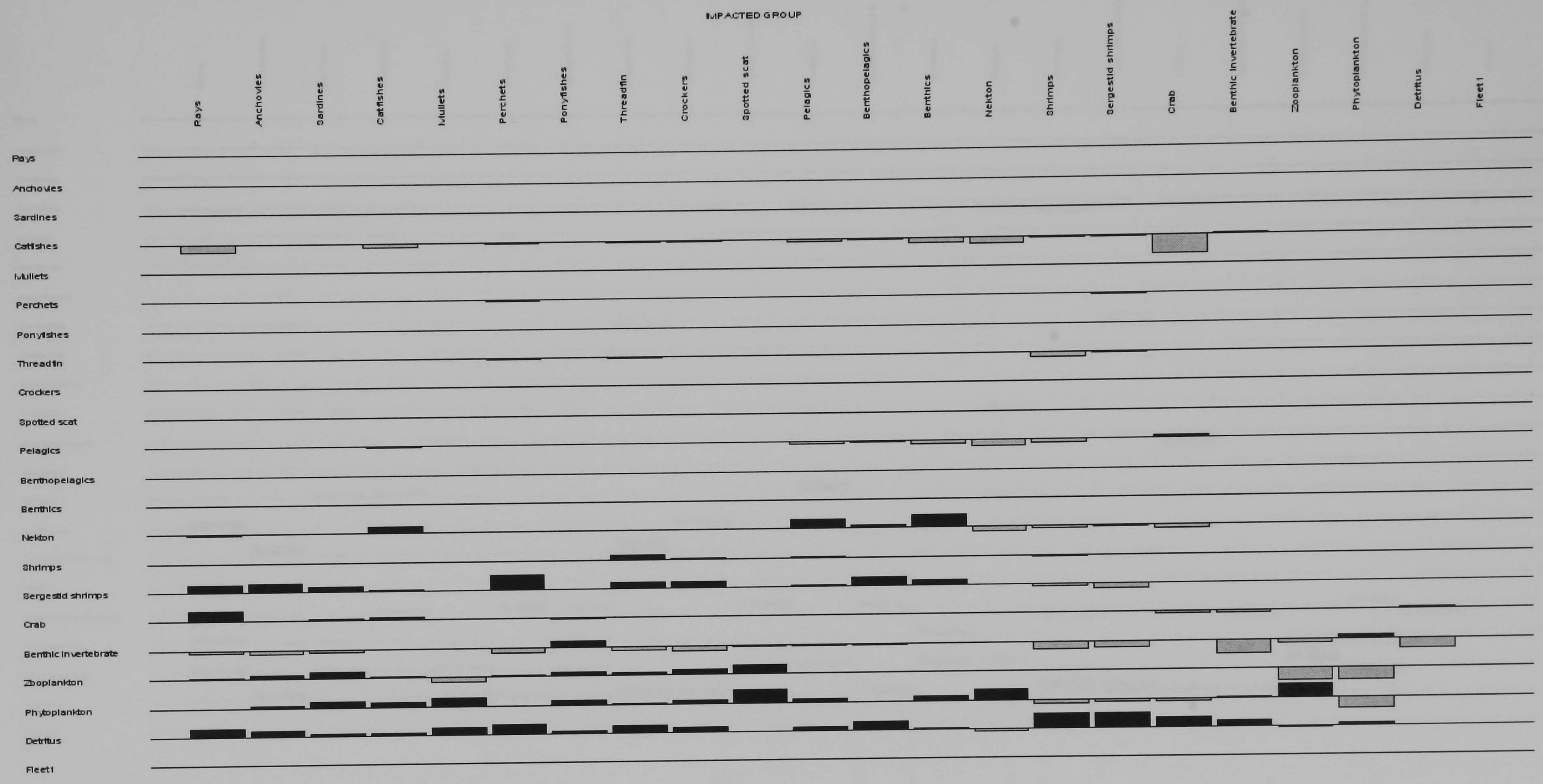
Table 4.20 Network characteristics of Mae Klong Estuary in the three seasons.

Parameters	Value			Units
	Dry	Hot	Rainy	
Total system throughput	8321.511	16999.000	50901.320	t/km^2/year
Predatory cycling index	1.22	0.80	1.17	% of throughput w/o detritus
Finn's cycling index	1.62	1.01	1.35	% of total throughput
Finn's mean path length	2.278	2.232	2.248	-
Finn's straight-through path length	2.301	2.381	2.344	without detritus
Finn's straight-through path length	2.241	2.210	2.218	with detritus
Ascendency (A)	10361.6	21399.9	63152.1	flowbits
% value	44.3	47.0	46.3	
Overhead (O)	13019.5	24157.0	73233.3	flowbits
% value	55.7	53.0	53.7	
Capacity (C)	23381.1	45556.9	136385.5	flowbits
%value	100.0	100.0	100.0	
Overheads on exports	399.7	525.7	1895.6	flowbits
% value	1.7	1.2	1.4	
Internal ascendency (Ai)	3281.6	7714.2	22598.9	flowbits
Internal capacity (Ci)	14194.9	27365.7	82165.3	flowbits
Internal relative ascendency (Ai/Ci)	0.231	0.282	0.275	
Flow diversity (C/T)	2.810	2.680	2.679	
Transfer efficiencies (%)	4.9	3.7	3.0	
Information	1.245	1.259	1.241	

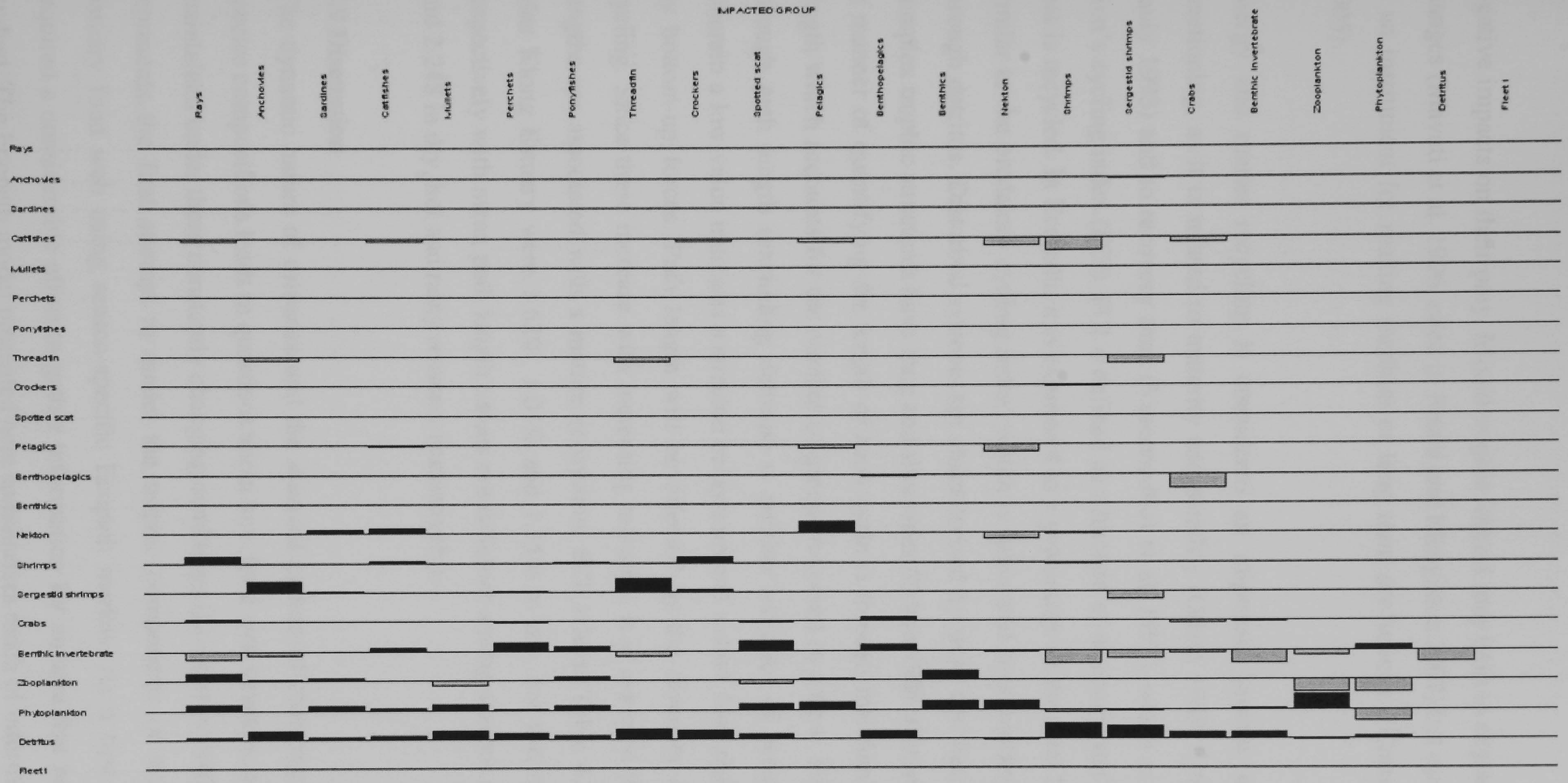
Figure 4.3 Mixed trophic impact of Mae Klong Estuary in the (a) dry season, (b) hot season, (c) rainy season. The impact in each group is positive when placed above the line and negative when below.



(a)


IMPACTED GROUP
Rays
Anchovies
Sardines
Catfishes
Mullets
Perchets
Ponyfishes
Threadfin
Crockers
Spotted scat
Pelagics
Benthopelagics
Benthics
Nekton
Shrimps
Sergestid shrimps
Crab
Benthic invertebrate
Zooplankton
Phytoplankton
Detritus
Fleet1

(b)


IMPACTED GROUP
Rays
Anchovies
Sardines
Catfishes
Mullets
Perchets
Ponyfishes
Threadfin
Crockers
Spotted scat
Pelagics
Benthopelagics
Benthics
Nekton
Shrimps
Sergestid shrimps
Crabs
Benthic invertebrate
Zooplankton
Phytoplankton
Detritus
Fleet1

(c)

negative impacts on their prey. Mixed trophic impact may help to explain short term changes (Mavuti et al. 1996, cited in Fetahi and Mengistou 2007) but cannot be taken as an instrument for making medium or long term predictions (Christensen et al. 2005).

Energy and matter recycling is considered an important process in ecosystem functioning, as it is related to maturity and stability (Odum 1969; Christensen and Pauly 1993) and to recovery time (Vasconcellos et al. 1997), which is measured as Finn's cycling index (FCI). FCI is defined as a fraction of an ecosystem's throughput that is recycled. In Ecopath, it is expressed as a percentage of the total flows. This is similar to the predatory cycling index, which is calculated by excluding the cycling through detritus. Disturbed systems are characterized by short and fast cycles while complex trophic structures have long and slow ones (Odum 1969; Christensen 1995). A manner of quantifying the length of each cycle is through the Finn's mean path length which accounts for the number of groups involved in a flow. Finn's straight-through path length (excluding detritus) is another indicator of ecosystem health wherein a low value indicates a stressed ecosystem and a short food chain controlled by bottom-up forces. Path length will be affected by the diversity of flows and cycling. Since these increase with increasing maturity, it is assumed that long path lengths are associated with a mature ecosystem. FCI values (Table 4.20) from the Mae Klong Estuary were 1.62%, 1.01% and 1.35% in dry, hot and rainy seasons, respectively with mean path length values not different among seasons (2.278, 2.232 and 2.248 in dry, hot and rainy seasons, respectively).

4.9 Discussion

The dynamic nature of estuaries and the seasonal pattern of changing biomass and species compositions leads to questions about how food web structure and function is maintained under these constantly changing conditions (Livingston 2002). This study represents the first attempt to model the trophic components of the Mae Klong Estuary food web using season-specific Ecopath models. As a first attempt, this required a considerable effort to gather information for an area that has never been studied. The Ecopath model presented here summarizes much of information that is available for the Mae Klong mangrove estuary ecosystem. The description of the Mae Klong ecosystem is based on estimations of the biomass and the fish production and on the components in the fish diet that gave an indication of the relationships between the 21 functional groups. The characteristics of the ecosystem model in this study are discussed here and at the same time it is compared with the 41 aquatic systems analysed by Christensen and Pauly (1993) and with other coastal ecosystem models (Table 4.21).

The Mae Klong Estuary ecosystem was examined as a whole using the model's global parameters. With Ecopath, functional groups are aggregated into discrete trophic levels sensu Lindeman (1942), as suggested by Ulanowicz (1995), which allows estimation of flows to detritus and upper trophic levels, and of transfer efficiencies. Some network attributes (Ulanowicz 1986) and flow indices were analyzed to describe holistic properties of the system, specifically total system throughput, ascendency, Finn's cycling index (Finn 1976) and Finn's mean path length. The ratios of net primary production to total biomass (PP/B) and net primary production to total respiration (PP/R) were also examined, as they are important indices of system maturity (Odum 1969).

4.9.1 Trophic level, energy flow and pathways

Fish and invertebrates are good environmental indicators to track environmental health and ecological changes, especially in estuaries and lagoons (Villanueva et al. 2006). Fish are the main top predators in these systems and may play a significant role in transferring energy out of the system due to feeding within the estuary and subsequent emigration to adjacent areas (Yãñez-Aranbicia and Nugent 1977). These authors also suggested that fish may play a significant role in transferring energy from primary producers to higher trophic levels within the estuary. In this study, fish themselves occupy the higher trophic levels, acting as top predators with several taxonomic groups represented in all seasonal models.

Estimated ecotrophic efficiencies of the fish groups were generally within the range 0.7-0.9, as usually assumed for fish (Ricker 1969). The high ecotrophic efficiencies

Table 4. 21 Comparison of the Mae Klong Estuary with other coastal ecosystems.

Ecosystems	Throughput ($t/km^2/yr$)	PP/B (/yr)	PP/R	Net system production ($t/km^2/yr$)	Relative Ascendency (%)	Finn's index (%TT)	Finn's mean path length (food chain steps)	References
Quintana Roo, Yucatan, Mexico	4,815,000	36.1	3.17	2,341,047	-	-	-	Vidal & Basurto 2003
SW Gulf of Mexico	7,712	-	-	3,535	-	11.5	6.8	Manickchand-Heileman et al. 1998
Tonamega lagoon, Mexico	2,853	-	0.56	-	32.7	-	-	Sophie Avila Foucat 2006
Mont Saint Michel Bay, France	9400	24.6	6.1	3,700	-	0.64	2.1	Leloup et al.2008
Eastern Scotian Shelf, Canada (1980-1985)	7,669	11.3	-	0.35	21.74	4.89	2.76	Bundy 2004
Eastern Scotian Shelf, Canada (1995-2000)	7,124	7.3	-	0.62	23.51	6.61	3.13	Bundy 2004
Pearl river delta, China	15,244	18.1	2.86	3,139	-	-	-	Lijie et al. 2008
Gulf of Paria, Venezuela and Trinidad	2,285	34	3.8	1,019	41.7	7.2	6.2	Manickchand-Heileman et al. 2004
North coast of central Java, Indonesia	6,745	11.0	2.87	1,598	-	8.58	2.75	Nurhakim 2003
São Sebastião Channel, SE Brazil	11,442	11.2	0.7	-621	25.4	30.1	-	Rocha et al.2007
Caeté Estuary, Brazil	10,558	-	-	-	27.4	17.9	3.4	Woff et al. 2000
Karnataka Arabian Sea, India	11,522	29.9	-	904	33.0	6.03	2.81	Mohamed et al. 2005
Somme Bay, France	2,312	-	-	-	35.0	12.2	-	Rybarczyk et al. 2003
Orbetello lagoon, Italy (1995)	24553	14.5	3.46	6,304	30.2	7.1	2.77	Brando et al.2004
Ébrié lagoon, West Africa	6240	41.5	5.15	2119	34.0	2.57	2.38	Villanueva et al. 2006
West coast of Sabah, Malaysia	3152	19.6	2.07	605	-	-	-	Garces et al. 2003
West coast of Sarawak, Malaysia	1414	19.3	2.08	273	-	-	-	Garces et al. 2003
Brunei, SE Asia	1816	28.6	-	300	29.4	16.3	2.8	Silvestre et al. 1993
Mae Klong Estuary (dry season)	**8,321**	**77.4**	**2.82**	**2,364**	**44.3**	**1.62**	**2.27**	**From this study**
Mae Klong Estuary (hot season)	**16,999**	**91.6**	**3.01**	**5,092**	**47.0**	**1.01**	**2.23**	**From this study**
Mae Klong Estuary (rainy season)	**50,901**	**88.4**	**2.94**	**14,961**	**46.3**	**1.35**	**2.24**	**From this study**

for most fish groups suggest that trophic relationships are tight and most of the system's secondary production is consumed by predators. The low ecotrophic efficiencies of detritus indicates that more detritus is entering this box than is leaving it, or that a significant quantity of detritus is being buried, or exported to the sea floor (Manickchand-Heileman et al. 1998).

The predominance of fractional trophic levels <4.0 found in the present study has also been reported for other coastal areas in the Gulf of Mexico (Odum and Heald 1972; Vega-Cendejas and Arreguín-Sánchez 2001; Vidal and Basurto 2003), west coast of Sabah and Sarawak, Malaysia (Garces et al. 2003), as well as for the Swartkops Estuary (South Africa), the Ems Estuary (Germany) and Chesapeake Bay, USA (Baird et al 1991). This may be attributed to the dependence of the food web on detritus and the abundance of juvenile fish which use the estuary as a nursery area (Yáñez-Aranbicia et al. 1998), whose production depends directly and indirectly on primary producers (Arreguín-Sánchez 2001). In contrast, higher fractional trophic levels were found on the continental shelf in the south-western Gulf of Mexico (Arreguín-Sáachez et al. 1993; Manickchand-Heileman et al. 1998), where adult fish are expected to be more abundant.

Mangroves play an important role in detritus accumulation due to the large amount of leaf material that is incorporated within the soil. About half of the detritus produced by fallen leaves is exported to adjacent aquatic regions mostly by tidal flush (Jacobi and Schaeffer-Novelli 1990). The other half is used by juvenile stages as a source of food by direct grazing on leaves and indirectly by detritus consumption (Lugo and Snedaker 1974; Thayer et al. 1987). The importance of these biological and energetic processes within these swamps is shown by the dependence on detritus of two-thirds of the world fisheries (Lai 1984). Increased cycling and storage both tend to increase the ratio of indirect to direct flows and contribute to network amplification and homogenization of available energy over all trophic levels (Patten et al. 1990). Detritus recycling or re-utilization involves the subsequent transformation of previously utilized but not dissipated energy-matter by consumers (Higashi et al. 1993).

The importance of detritus and primary production pathways in ecosystems, such as mangrove estuaries was noted by Vega-Cendejas and Arreguín-Sánchez (2001). De Sylva (1985) indicated that estuarine nekton follow either a detritus-based or a phytoplankton-based food chain. Primary producers and detritus are energy sources that play differing roles and significance in the diet of fish of higher TLs in the Mae Klong Estuary. My results showed that phytoplankton and detritus are the key food sources that sustain mainly the zooplankton secondary production, similar to observations in the Sundarban, India (Ray et al. 2000) and the Yucatan Peninsula, Mexico (Vega-Cendejas and Arreguín-Sánchez 2001) mangrove ecosystems. Energy flow in the Mae Klong estuary is also consistent with what is known about coastal lagoons and estuaries in general. The dominance of the detrital pathway as observed in this study has been reported for other shallow estuaries and coastal lagoons in the Gulf of Mexico (Odum and Heald 1972; Vega-Cendejas and Arreguín-Sánchez 2001), Caeté mangrove estuary, North Brazil (Wolff et al. 2000) and elsewhere, for example, the Swartkops Estuary of south-east South Africa and in Chesapeake Bay in the eastern USA (Baird et al. 1991), Bay of Dublin, Ireland (Wilson and Parkes 1998) and the Kromme Estuary of southern South Africa (Heymans and Baird 1995). The high biomass of TL1 (detritus and primary producers) and its significant role in supporting the energy utilized indicate a bottom-up control in the Mae Klong Estuary.

4.9.2 Maturity of the Mae Klong Estuary: Comparison among seasons

Mae Klong Estuary was characterized by a higher level of organization in the rainy season than in the dry and hot seasons. This could be linked to a higher redundancy of the flows in dry and hot seasons. This higher level of organization in the rainy season implies a lower adaptation capacity (Heymans et al. 2002).

Ecosystem indices in the different seasons illustrate a pattern of food web development throughout the year from low values in the dry season to the highest level of organization in the hot and rainy seasons (Table 4.20). Capacity and overhead peaked in the rainy season indicating that the rainy season is a robust food web that can recover quickly from perturbations. The high potential for development embodied in high values of capacity and overhead was used up as the system became more organized and the food web became more fully developed until the system reached its peak ascendency in the hot and rainy seasons. The cycle begins again in the dry season as the ascendency, overhead and capacity were reduced by seasonal shifts in species composition, biomass and production patterns. In the development of ecosystems sensu Odum (1969), the Mae Klong Estuary shows a succession of communities: an initial developmental stage in the dry season, which then becomes more organized into a mature community in the hot and rainy seasons. A similar pattern of succession in seasonal dynamics have been found in Chesapeake Bay (Baird and Ulanowicz 1989) and Weeks Bay (Althauser 2003) estuaries. There are few other studies that quantify the seasonal succession of estuarine food webs, so conclusions regarding patterns of estuarine development must be considered preliminary (Ulanowicz 1995).

Finn's cycling index obtained from this study was relatively low and the values are similar to all seasons (1.6, 1.0 and 1.3 in dry, hot and rainy seasons, respectively). It can be concluded that the Mae Klong Estuary has low recycling in general. In comparing among seasons, the dry season has a higher capacity to recycle detritus than other seasons and shows greater ability for recovery.

4.9.3 Maturity of the Mae Klong Estuary: Comparison with other coastal ecosystems

A comparative approach with other coastal ecosystems is helpful to characterize the structure and material flows in the Mae Klong Estuary. However, there are very limited quantitative descriptions of food webs for tropical/subtropical ecosystems (Lin et al. 2007).

The model estimate of total system throughput (*T*) in the rainy season of 50901 t/km^2/yr (Table 4.19) appears high when compared to other coastal systems (Table 4.20). The high biomass and production values for benthic producers, including mangroves (phytoplankton and detritus in this study), and the large organic nutrient loading from the upper reaches are probably the reasons for the high throughput values (Lin et al. 2007). These throughput values are still low, however, when compared to Quintana Roo, Yucatan, Mexico, which had *T*-values of 4,815,000 t/km^2/yr.

Odum (1969) demonstrated that the primary production/respiration (PP/R) ratio reflects the maturity of an ecosystem. He suggested that the rate of primary production exceeds the rate of community respiration during early stages of ecosystem development, and hence PP/R is greater than one. However, in a mature system the ratio approaches 1 because the energy fixed tends to be balanced by the energy cost of maintenance. In their comparative study of 41 aquatic ecosystems, Christensen and Pauly (1993) found that the bulk of PP/R ratios were in the range between 0.8 and 3.2, although the extreme values were <0.8 and >6.4. PP/R values of 2.8-3.0 obtained from the Mae Klong Estuary are larger than 1 which is similar to other coastal ecosystems like Quintana Roo, Yucatan (Mexico), Pearl river delta (China), North coast of central Java (Indonesia) and West coast of Sabah and Sarawak (Malaysia) (Table 4.20). This value implies that the Mae Klong Estuary and those other ecosystems are in an early developing stage and are prone to ecological perturbations, including anthropogenic impacts (Fetahi and Mengistou 2007). In contrast, the PP/R ratio of 0.56 in Tonameca lagoon, Mexico, indicates that Tonameca lagoon is probably mature and with a low level of organic matter (Avila Foucat 2006).

The PP/R ratios of the Mae Klong Estuary also indicate moderate eutrophication when compared with the value of 1.12 from Lake Nokoué, West Africa (Villanueva et al. 2006), which indicated a level close to "eutrophic status" as total system respiration approaches its production, which is the common feature in highly polluted systems. However, this may not be true if based on recent environmental domestic and industrial pollutions loads (Villanueva et al. 2006). Besides, system ascendency (*A*) and *T* can also be used as indicators of eutrophication in ecosystems (Mann et al. 1989). This is characterized by an increased value in *A*, as a function of elevated *T* parallel to a fall in information (*I*) (Ulanowicz 1986).

Estimated net system production (NSP or yield) in the Mae Klong Estuary, however, is higher than in those ecosystems such as the Eastern Scotian Shelf, Canada, Gulf of Paria, Venezuela and Trinidad, North coast of central Java, Indonesia and Karnataka Arabian Sea, India, etc. However, the NSP values from this study are similar to other ecosystems, for instance SW Gulf of Mexico, Pearl river delta, Chinaand Orbetello lagoon, Italy, while the values were relatively low compared to those obtained by Vidal and Basurto (2003) in Quintana Roo, Yucatan, Mexico, 150 times greater than for the Mae Klong Estuary.

The estimated values of some properties, such as ascendency and development capacity are tools to evaluate the organization, maturity and tolerance to perturbations, as well as for ecosystem comparisons (Mann et al. 1989; Baird et al. 1991). According to Ulanowicz (1986), these properties tend to increase with maturity and decrease in systems under natural or anthropogenic stress. Relative ascendency values of 44-47 % of Mae Klong Estuary are relatively high when compared with many other coastal ecosystems (Table 4.20), but similar to the Gulf of Paria in Venezuela and Trinidad (41%); these values imply that the Mae Klong Estuary is more mature than other coastal ecosystems. However, Christensen (1995) and Aoki (1997) argue that ascendency is not the best indicator of the degree of eutrophication and maturation, and has a negative correlation with them, suggesting that relative ascendency should be called relative mutual information, which provides a measure of the distribution of flows in a system network in relation with the total flow (Patten 1995).

The model identified the Mae Klong Estuary as a highly productive ecosystem and the Leontief matrix routine demonstrated that it is largely controlled from the bottom-up which results from high nutrient inputs from river discharges draining mangroves and surrounding aquaculture ponds. However, when compared with other ecosystems (Table 4.21), global indicators (high PP/B and PP/R, low Finn cycling index and mean path length) suggest that the Mae Klong Estuary ecosystem is immature, in line with Odum (1969), Finn (1976) and Ulanowicz (1986, 1995). Low maturity status is commom in megatidal coastal and estuarine systems, such as the bay of Mont Saint Michel (Leloup et al. 2008), due to the low rate of transfer of primary production (Le Pape and Menesguen 1997). Even if it is sometimes difficult to compare different systems which have different degrees of compartment aggregation, the very low values of the cycling index in the Mae Klong Estuary reflect an especially immature system.

The discrepancy in the Finn cycling index could change the interpretation of the developmental state of the ecosystem in the Mae Klong Estuary analysis. Odum (1969) found that cycling increases as systems mature (thus the FCI increases), although some discrepancies have been recorded in the interpretation of cycling with regards to ascendency and overhead. Baird et al. (1991) concluded that FCI shows the reverse rank-order correlation with ascendency, and FCI is not a measure of systems maturity but of stress, while Ulanowicz (1986) defined FCI as a measure of maturity. Subsequently, Christensen (1985) has shown that not ascendency, but overhead, is related to a system's maturity, and thus an increase in FCI with an increase in overhead is an indication of system maturity. Vasconcellos et al. (1997) also found that recycling is the "chief positive feedback mechanism that contributes to stability in mature systems by preventing overshoots and destructive oscillations due to external impacts". Taking into consideration the controversy surrounding maturity and cycling of systems, it would be prudent to be careful when comparing the FCIs of systems. Futhermore, when comparing FCIs, consideration should be given to the currency used for comparison (Field et al. 1989).

The immature status of the Mae Klong Estuary trophic network may be explained partly by the intensive human exploitation of the estuary, through shellfish (blood cockle and horse mussel) farming (Alongi 2002)). There may also be impacts in the estuary due to wider fishing activity offshore in the Gulf of Thailand (Pauly and Christensen 1995; Christensen and Pauly 1998) because many commercial species breed in the estuary and use it as a nursery ground. These are large losses of primary production due to hydrodynamic exchanges (Le Pape et al. 1999).

It can be seen that the Mae Klong Estuary has a mixture of characteristics of a mature system (high total system throughput, ascendency and overhead) as well as an immature system (high PP/B and PP/R, low Finn's cycling index and mean path length). In addition, detritus-based food webs, high fish and flow diversities (Tables 4.15-4.17) are typically related to maturity (Vega-Cendejas and Arrguín-Sánchez 2001). This is consistent with the system experiencing a moderate level of exploitation, driving its development back to earlier developmental stages (Odum 1972).

CHAPTER 5

Dynamic simulation of mass-balance models for fish community of Mae Klong Estuary

5.1 Introduction

In 2002, 72% of the world's marine fish stocks were being harvested faster than they could reproduce (UNEP 2004), and with fishing the major form of direct utilization (Pauly et al. 2002; Robinson and Frid 2003), this has lead on a global scale to a general decline in fish biomass, stock depletion (Botsford et al. 1997), reduction in the mean trophic level of the catches (Bundy and Pauly 2001), marine habitat disturbance (Hall 1999) with many species now of conservation concern (Bundy and Pauly 2001; Pauly et al. 2002). Not unexpectedly, there is debate about the ultimate causes of over-fishing, including poor management practices and increased fishing pressure. Unsustainable fishing practices, along with an excessive level of investment in fishing capacity, have resulted in serious stock depletion, creating new pressure on alternative fishing grounds (Pauly et al. 2002).

Declining biomass is expected from fishing on populations and is necessary for the density-dependent increase in production that is the basis for sustainable fisheries harvests (OSB 2006). However, in many cases, overfishing has resulted in the collapse of populations and the fisheries that depended on them (e.g., north Atlantic cod) (Árnason et al. 2009). Numerous papers point to the decline of food fish biomass in various areas: the North Atlantic (Christensen et al. 2004), Gulf of Thailand (Christensen 1998), the Gulf of California (Sala et al. 2004) and more generally around the world (Gulland 1988).

In addition to effects on fish populations, there are effects on the wider ecosystem. Fishing has been described as a force that structures ecosystems from the top-down (Pauly 1979). Fishing also directly exploits species at lower trophic levels (Bundy and Pauly 2001). These exploited species are part of the complex trophic network, so that assessing the impacts of multi-species fishing within such trophic networks means that both the direct effect of fishing and its indirect effects mediated through other species in the food web need to be taken into consideration. This is reflected in the substantial experience gained in recent years from fisheries science which suggests that an exploited stock is not the functional unit in a fishery (Gulland 1988; Christensen and Pauly 1992a, b, 1995). Stock fluctuations also depend to a large extent on interdependencies among other species in the ecosystem, which propagate through food webs as changes in biomass flows (Arreguín-Sánchez 2000).

Fisheries scientists recognize *ecological interdependence* when two stocks have a competitive or a predator-prey relationship (Seijo & Defeo 1994; FAO 1995). It can also account for intraspecific interactions (*e.g.* between recruits and adults: Defeo 1998). In fisheries science, one of the first approaches that incorporated interdependencies between species was the Lotka-Volterra (also known as predator-prey) model (Lotka 1925; Volterra 1926), which accounted for direct interdependencies through competition or predation (Walters et al. 1997) allowing the development of multispecies yield models (for example, Arreguín-Sánchez et al.1993; Arreguín-Sánchez 2000) and providing useful insights into population dynamics and stability (Knadler, Jr 2008). From these applications, it is evident that exploited (and unexploited) stocks are not independent or discrete units in an ecosystem; that trophic interdependencies usually are not two-species single systems and neither are only direct relationships relevant; and that the variability of a given stock is a consequence of the totality of interactions in the ecosystem (Arreguín-Sánchez 2000).

One approach to exploring the effects of harvesting within trophic networks like fisheries ecosystems is to construct balanced network models and then carry out harvesting "experiments". The balanced model developed here (Chapter 4) was used to explore the possible impact of varying fishing mortality on the biomass of other major groups in the network using the Ecosim routine (Walters et al. 1997), a dynamic extension of Ecopath. This approach allows an evaluation of the response of the entire system to different perturbations and to different exploitation regimes, under assumptions of bottom-up, mixed or top-down flow control mechanisms (Walters et al. 1997; Ortiz and Woff 2002). The balanced model described in Chapter 4 contains many target species which could be potentially harvested experimentally at different rates. Here, I focus on shrimps and sergestid shrimps because of their importance as a fishery in the Mae Klong Estuary as well as being important in the diets of other target species.

5.2 Shrimp fisheries

Globally, about 60% of shrimp production in the world comes from fishing and Asian countries account for 55% of the world shrimp catch (FAO 2008). In many of the Asian multi-species fisheries, primary trawl fisheries target various species of shrimp, operating in shallow waters close to the coast (Willmann 2005). Many tropical fisheries are inherently of a multiple species nature, with any given gear type exploiting a wide range of species (Pauly 1979; Welcomme 1985). Shrimps are the most valuable part of the demersal catch because of the high landings and/or the high market values (Willmann 2005). Kellecher (2005) indicated that shrimp trawl fisheries are the single greatest source of discards, accounting for 27.3% (1.86 million tonnes) of estimated total discards. The aggregate or weighted discard rate for all shrimp trawl fisheries is 62.3%, which is extremely high compared with other fisheries.

The exploited marine shrimp stocks belong mainly to two groups- the penaeid species (mostly *Matapenaeus* and *Penaeus*, and the non-penaeid species (mostly the sergestid shrimp, *Acetes*) (FAO 1989). In Thailand, the main gears of the shrimp fishery operated in the traditional sector are shrimp gillnets, tidal traps and push nets. The catches consist of small-size *Penaeus merguiensis* and sergestid shrimps *Acetes* spp. (FAO 1989), caught in significant amounts mostly by the small-scale sector and the density of these small shrimps in the inner Gulf of Thailand is much greater than elsewhere in the whole Gulf (SCS 1981).

Sergestid shrimps are one of the most important commercial shrimp resources (Rönnbäck 1999; Arshad et al. 2008) and form a significant part of the diet of many commercial fishes (SHARP 2004). Whilst not well known in many regions outside Asia, sergestid shrimps are very important in terms of global catches and are the basis for the largest shrimp fishery in the world (FAO 2008). Species of the genus *Acetes* live in the estuaries and coastal waters of tropical and subtropical regions (FAO 1989; Arshad et al. 2008) and are caught in large numbers in mangrove creeks (UNEP 1985; Rönnbäck 1999). *Acetes* was taken more than any other shrimp in the world in 2005, the catch amounting to 665,000 tonnes (FAO 2008), making sergestid shrimps major economically important shrimps in Asian and East African waters (UNEP 1985). In many Asian countries, only a small proportion of the catch is marketed as fresh shrimps; the greater proportion is dried, salted or fermented to be used in various forms of food, especially shrimp paste (UNEP 1985). Shrimp paste and shrimp sauce are manufactured extensively throughout Southeast Asia and are prized for their taste and nourishment (Omori 1977; Rönnbäck 1999).

Here, the trophic models previously constructed in Chapter 4 for the Mae Klong Estuary were used to analyse the effect of harvesting of shrimps and sergestid shrimps (*Acetes* sp.) on other key target species - anchovy, catfish, croaker, mullet and threadfin, and, if effects were found, what the likely mechanisms might be.

5.3 Dynamic simulation model: Ecosim

Ecosim is a dynamic simulation tool embedded in the EWE software. It estimates changes of biomass among functional groups in the ecosystem as functions of abundance among other functional groups and time varying harvest rates, taking into account predator–prey interactions and foraging behaviors (Pauly et al. 2000; Walters et al. 2000). "Ecosim contains specific hypotheses for surplus production that differ from traditional single-species management models. Specifically, Ecosim begins with an assumption that all species are tightly connected and energetic surplus does not arise through fishing, whereas single-species fishing theory implies that fishing leads to surplus by removing larger, older, less-productive fish from populations. Although Ecopath production ratios and single-species estimated production levels are both derived from the dynamics of von Bertalanffy consumption and growth equations, the dynamics of Ecosim differ from the implied bioenergetics of fishing as applied to age-structured populations" (Aydin 2004). The model behavior is based on a 'foraging arena' theory (Christensen et al. 2005), which assumes that predator and prey behaviors cause partitioning of prey populations, which are either available or unavailable to predators at any given point in time (Figure 5.1). There is continuous change between these two stages for any given potential prey, whether it is hiding from predation in some refuge, or it is out to feed. This availability of prey to predators is called 'vulnerability' in Ecosim (Christensen and Walters 2004). The foraging arena typically operates on a timescale of seconds to minutes, and a geographical scale measured in metres (BSRP 2004).

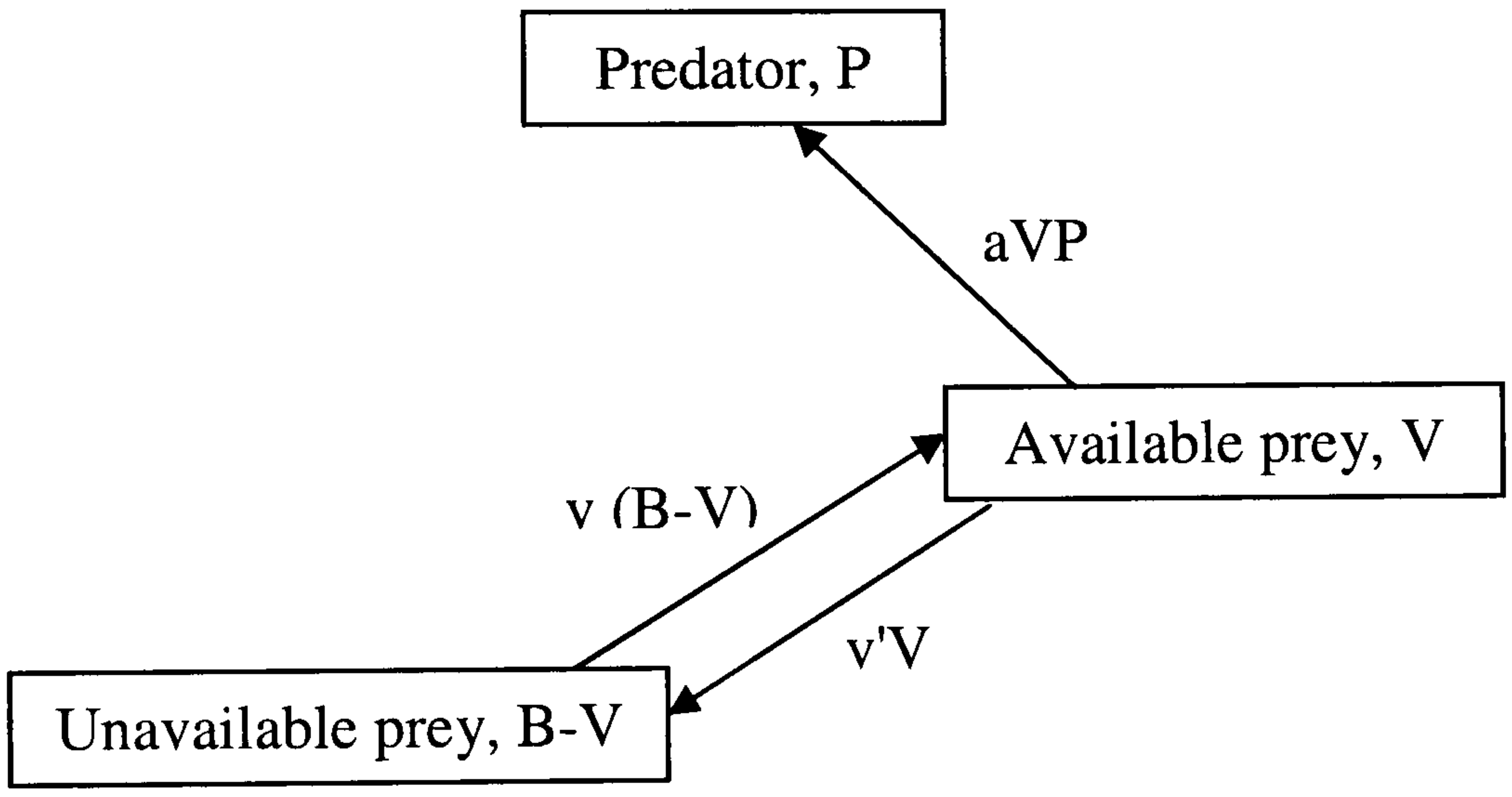


Figure 5.1 The foraging arena assumes that prey is only available to predators part of the time, typically when the prey themselves are feeding (Christensen et al. 2005).

Ecosim can be used to explore the direct and indirect ecological effects of fisheries, perturbations, and even physical forces (Walters et al. 2000; Okey 2004). The current version of Ecosim allows for the representation of ontogenetic changes in diets, mortality rates, and vulnerability to fishing for particular populations in the model (Walters et al. 1997). Ecosim II extends the age-structure submodel by providing for a delay-difference population model structure with monthly age categories for juvenile fish (Walters et al. 2000). Additionally, Ecosim (version 4.0) has an Ecospace component that is designed to analyse models with spatial structure (Walters et al. 1999). Two important limitations of Ecosim are that it does not employ prey switching in consumption functions and it depends strongly on the assumptions of Ecopath network construction to simplify parameter estimation. Nevertheless, it is a useful tool for analyzing broad fishery scenarios (Morissette 2007).

5.4 Material and methods

5.4.1 Ecosim modelling approach

I use Ecosim (Walters et al. 1997) to simulate the changes in harvesting rate of shrimps and sergestid shrimps on key commercial groups of fish. The Ecosim routine expresses the mass-balanced constraint in the dynamic context, primarily a biomass-based model of coupled different equations that can be re-expressed as:

Δ Biomass = Growth + Immigration – Predation - Mortality

$$\frac{dBi}{dt} = g_i \sum_j C_{ji} - \sum_j C_{ij} + I_i - (MO_i + F_i + E_i)B_i \qquad \text{eq. 5.1}$$

Where,

dB_i / dt = the growth rate in biomass of group i,

C_{ij} = the trophic flow of biomass per time, between prey (i) and predator (j),

g_i = net growth efficiency (production/consumption ratio),

MO_i = natural mortality rate of group i,

F_i = fishing mortality rate of group i,

E_i = emigration rate,

I_i = immigration rate

The emigration and immigration rate are considered absent, and fishing mortality is included in the total mortality as well as the natural mortality.

One of the pillars of Ecosim is the 'foraging arena theory', which state that preys are not always available to predators (see also above):

$$C_{ij} = v_{ij} \,.\, a_{ij} \,.\, B_i \,.\, B_j / v_{ij} + v'_{ij} + (a_{ij} \,.\, B_j) \qquad \text{eq. 5.2}$$

Where,

v_{ij} and v'_{ij} = prey vulnerability parameters, which default setting $v_{ij} = v'_{ij}$,

B_i and B_j = biomas of prey and predators, respectively,

a_{ij} = rate of effective search by predator $_j$ for prey $_{i.}$

Parameters v_{ij} and v'_{ij} represent prey vulnerabilities, or the rates of exchange of biomass between two prey behavioral states (i.e., movement between refugia and foraging area) of the functional groups in predator-prey interactions (Walter et al. 1997). Prey vulnerabilities can be specified by setting vulnerability parameters to control the extent to which the model moves towards top-down and away from bottom-up control (Plagányi and Butterworth 2004). Adjustment of the proportion of prey in vulnerable and invulnerable states (pools) is via adjustment of the v values. This parameter can range from 1.0 for top-down to 0.0 for bottom-up control. A value of 0.3 represents a mixed control (Ortiz 2001). The top-down control leads to rapid oscillations of prey and predator biomasses, and the bottom-up control often leads to unrealistically smooth biomass changes in prey and predator dynamics, which usually do not propagate through the food web (Zetina-Rejón et al. 2001). Not all prey biomass is vulnerable to predation at any given time, and predator-prey relationships are limited by behavioral and physical mechanisms, so Ecosim predictions are very sensitive to this parameter. Using default values for v has strong implications for assumptions about species abundance relative to their carrying capacity (Morissette 2007). The system of equations are solved on a monthly time step for up to one hundred years (Walters et al. 1997).

5.4.2 Model analysis: The scenarios

I tested a new harvesting scenario of the effects of shrimps and sergestid shrimps biomass change on the biomass of the key commercial groups: anchovy, catfish, threadfin, croaker and mullet. The shrimp and sergestid shrimp biomass changes in the Mae Klong Estuary used for the Ecosim simulations are shown in Table 5.1. The dry season model (21 functional groups) was used to produce the simulations of shrimps and sergestid shrimps biomass changes of 25%, 50% and 75% at 5, 10, 15 and 20 years. Default Ecosim settings were used (with no forcing function) and an average value of the vulnerability rate (Pauly and Christensen 2002; Christensen and Walters 2004), since I have no information on whether the system is controlled from the top-down or the bottom- up for this model. A *bottom-up* control means that the flow of energy between two compartments is limited by food resources or controlled by the preys; *top-down* control holds that the flow is regulated by the predators (Patten 1997).

Table 5.1 Shrimp and sergestid shrimp biomass change in Mae Klong Estuary used for Ecosim simulations.

Percent change of the initial biomass	Shrimp biomass (t/km^2)	Sergestid shrimp biomass (t/km^2)
0%	0.52	1.96
-25%	0.39	1.47
-50%	0.26	0.98
-75%	0.13	0.49

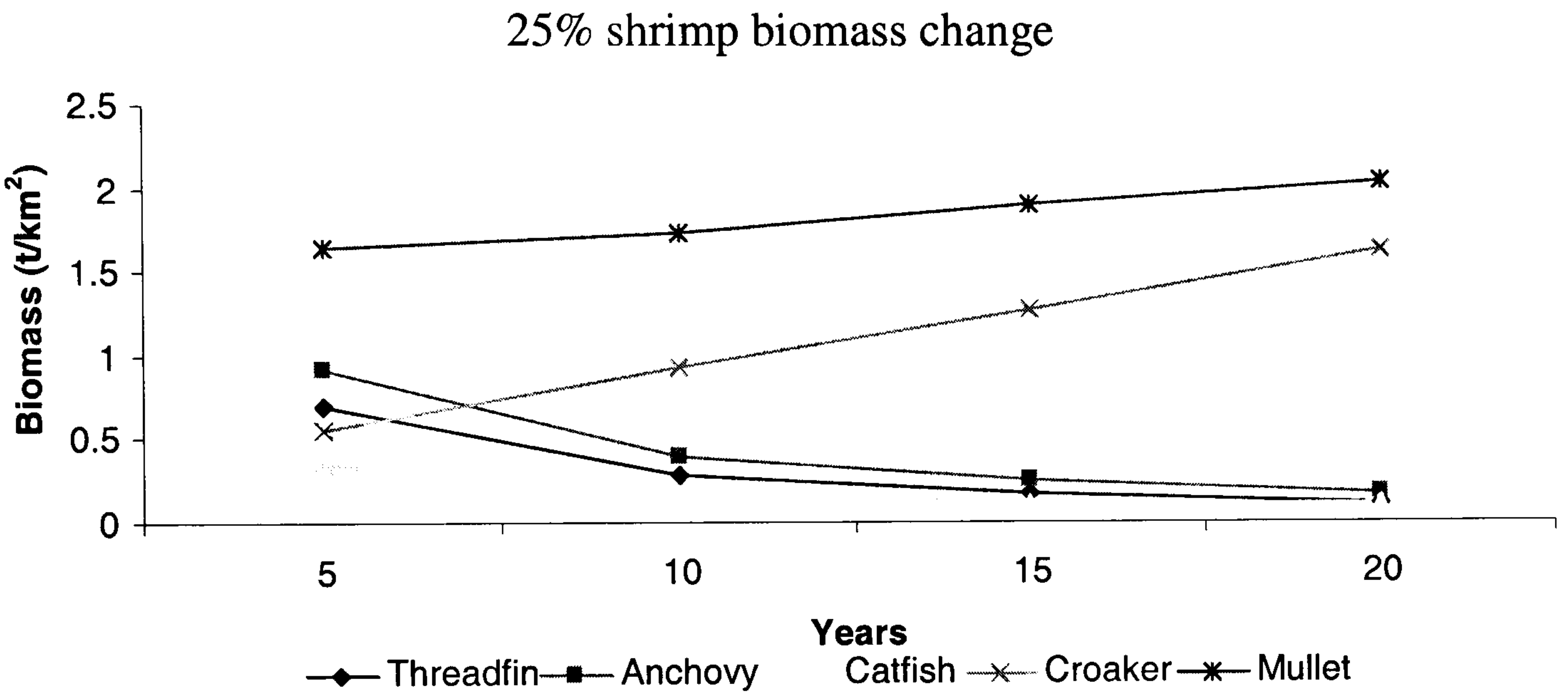
5.5 Results

Two types of simulation were done. Figures 5.2 and 5.3 show change in biomass in response to the key target fish species over a 20-year period under different shrimp harvesting scenarios.

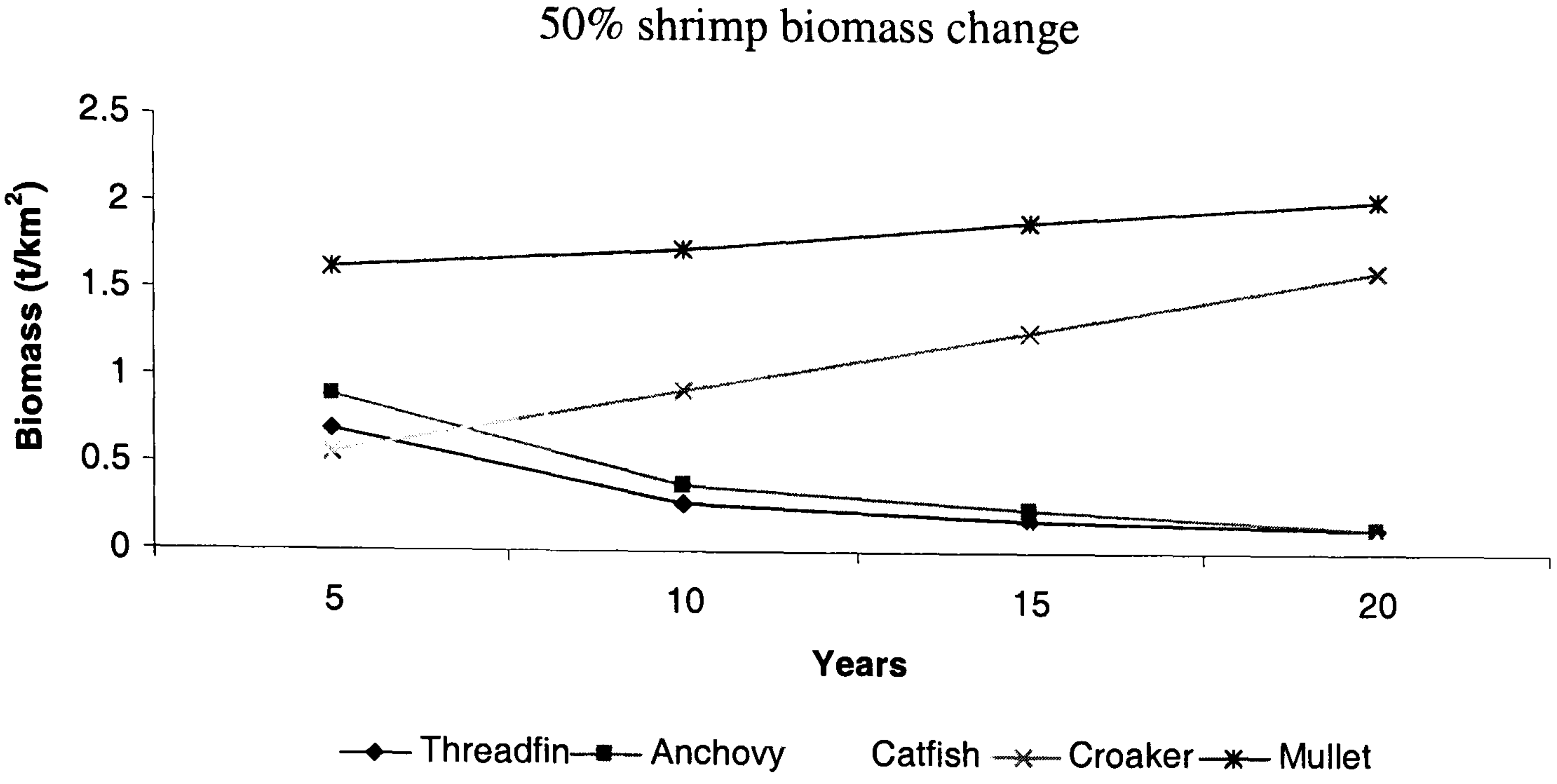
The patterns of change are broadly the same for both shrimps and sergestid shrimps. That is, one group of fish (anchovy, threadfin, and catfish) increase slightly in biomass and then decline again towards a new stable equilibrium observable after 5-10 years.

The second group of fish (mullet and croaker) continue to increase in biomass over the 20 year time period, with little indication of returning to their original biomasses or of stabilising over time.

These patterns are almost identical whether shrimp or sergestid shrimp biomasses are removed, and whether 25%, 50% or 75% of shrimp and sergestid shrimp biomasses are removed (Figures 5.2, 5.3).

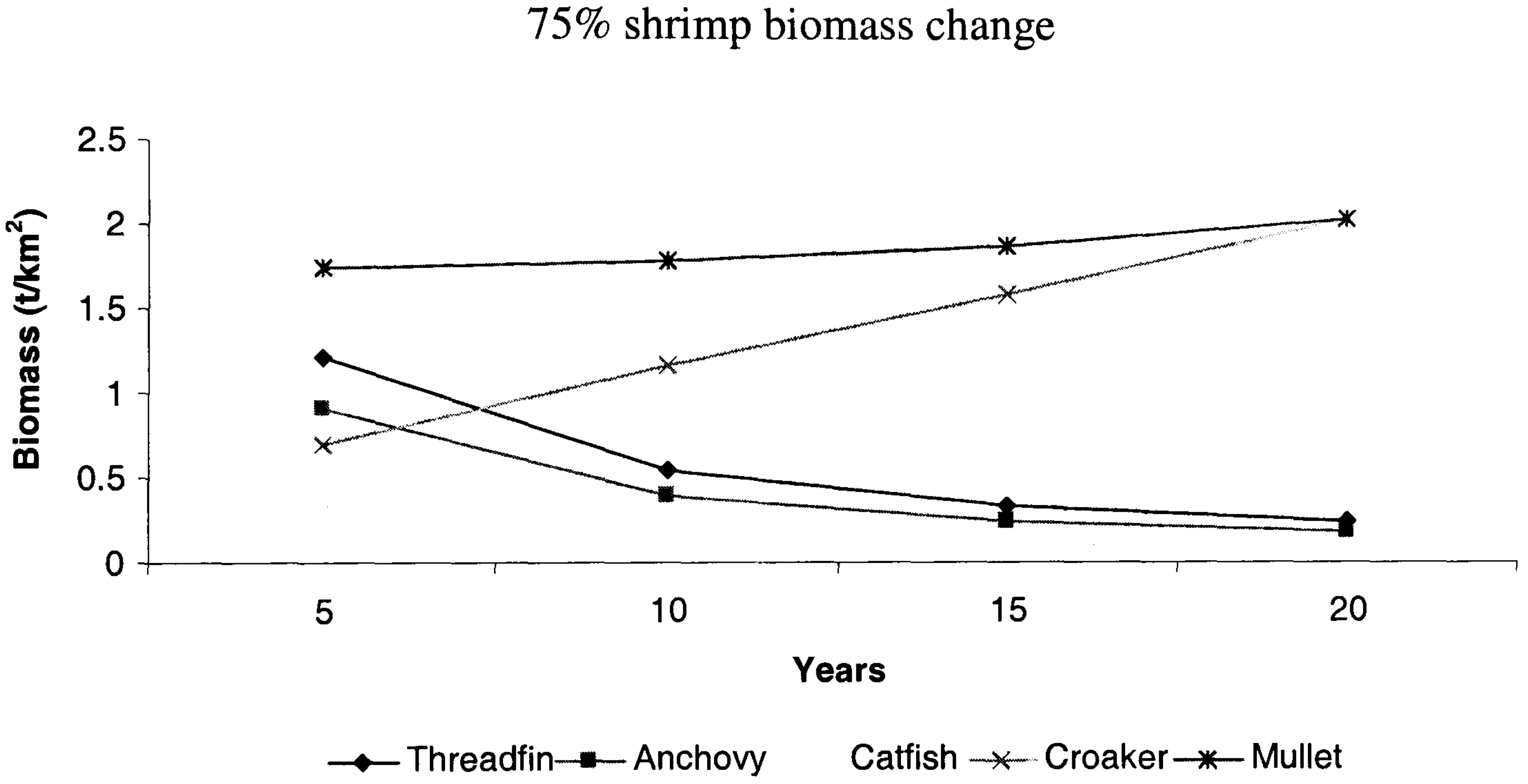


(a)



50% shrimp biomass change

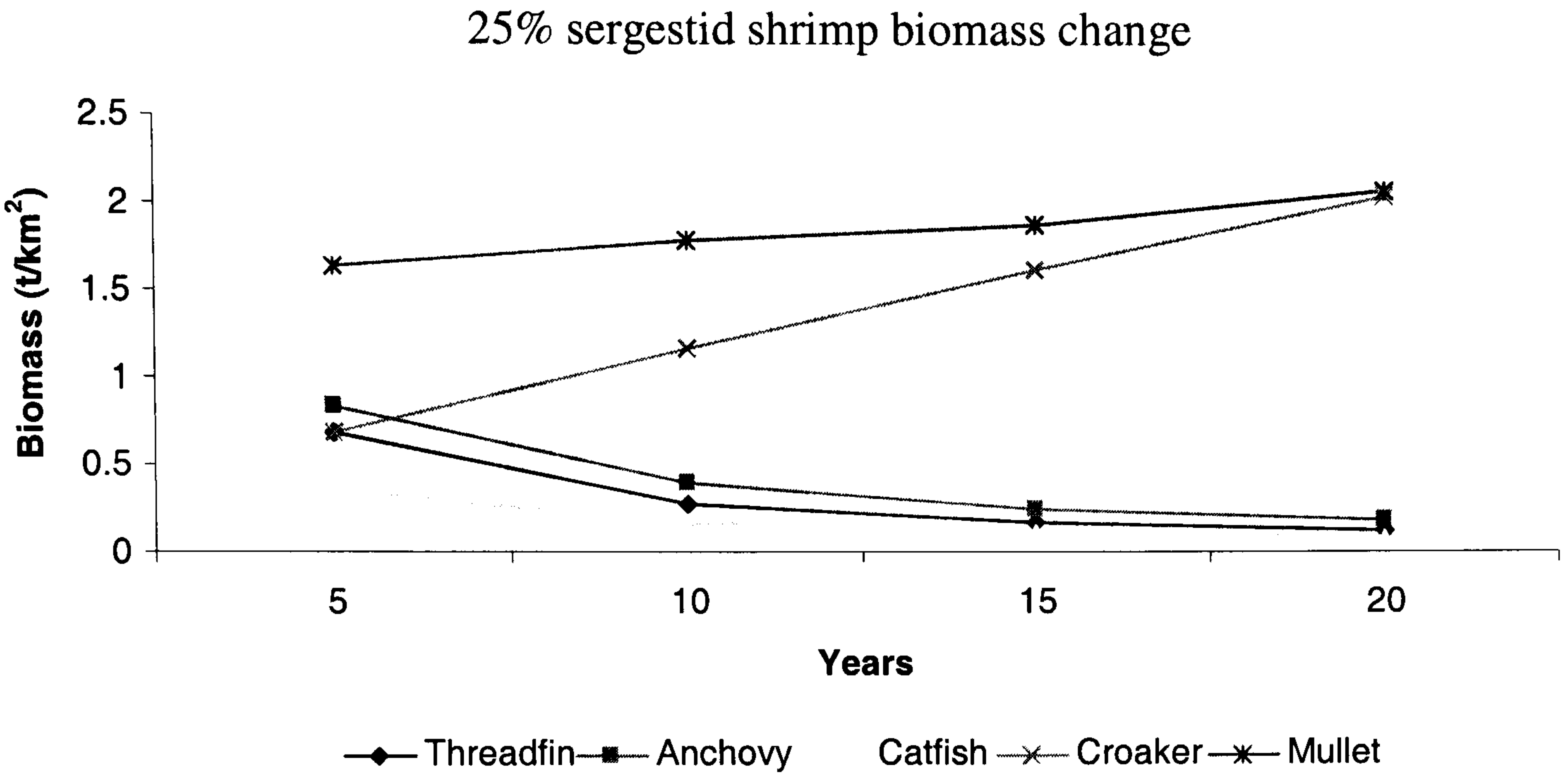
(b)



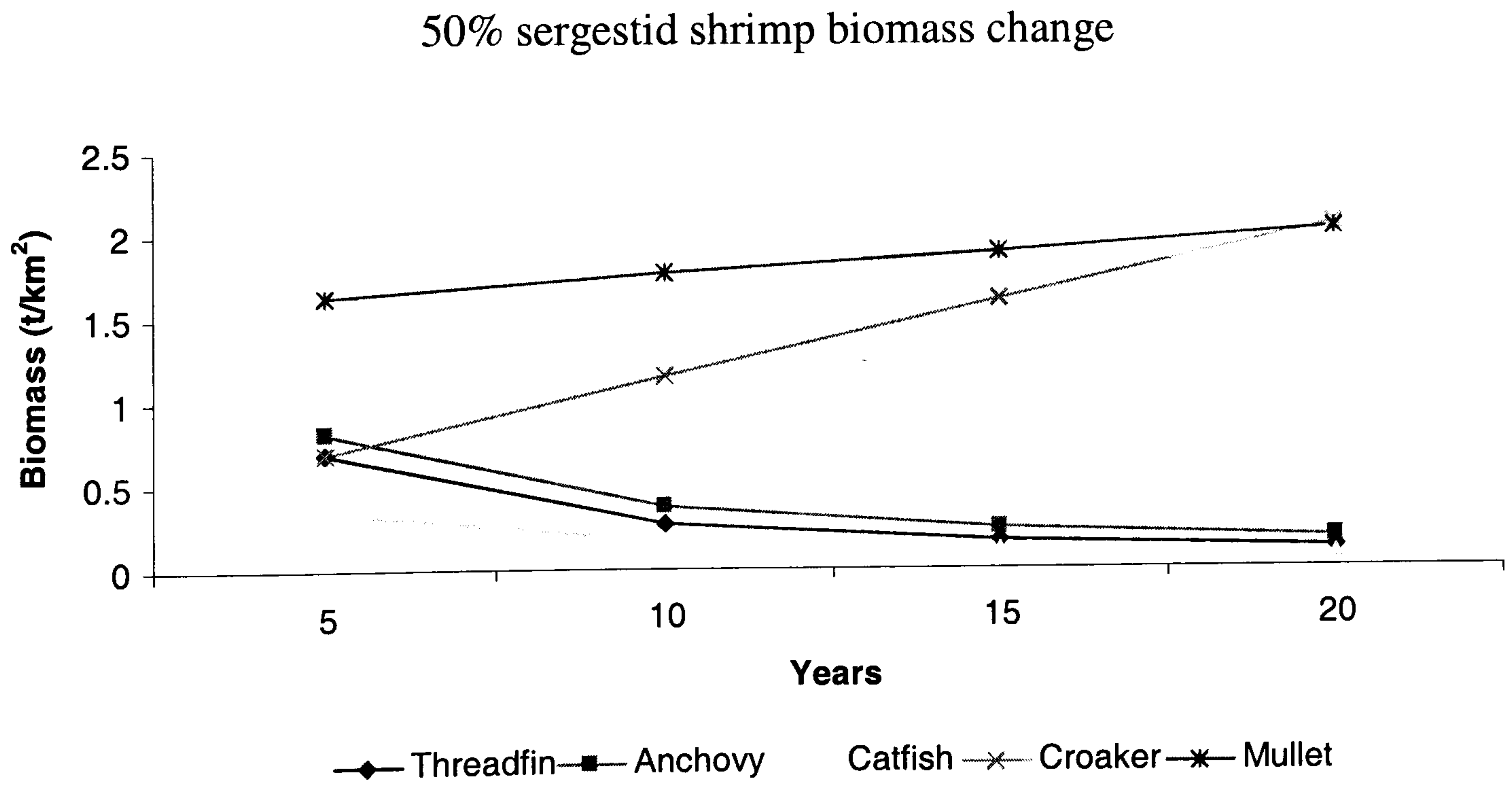
75% shrimp biomass change

(c)

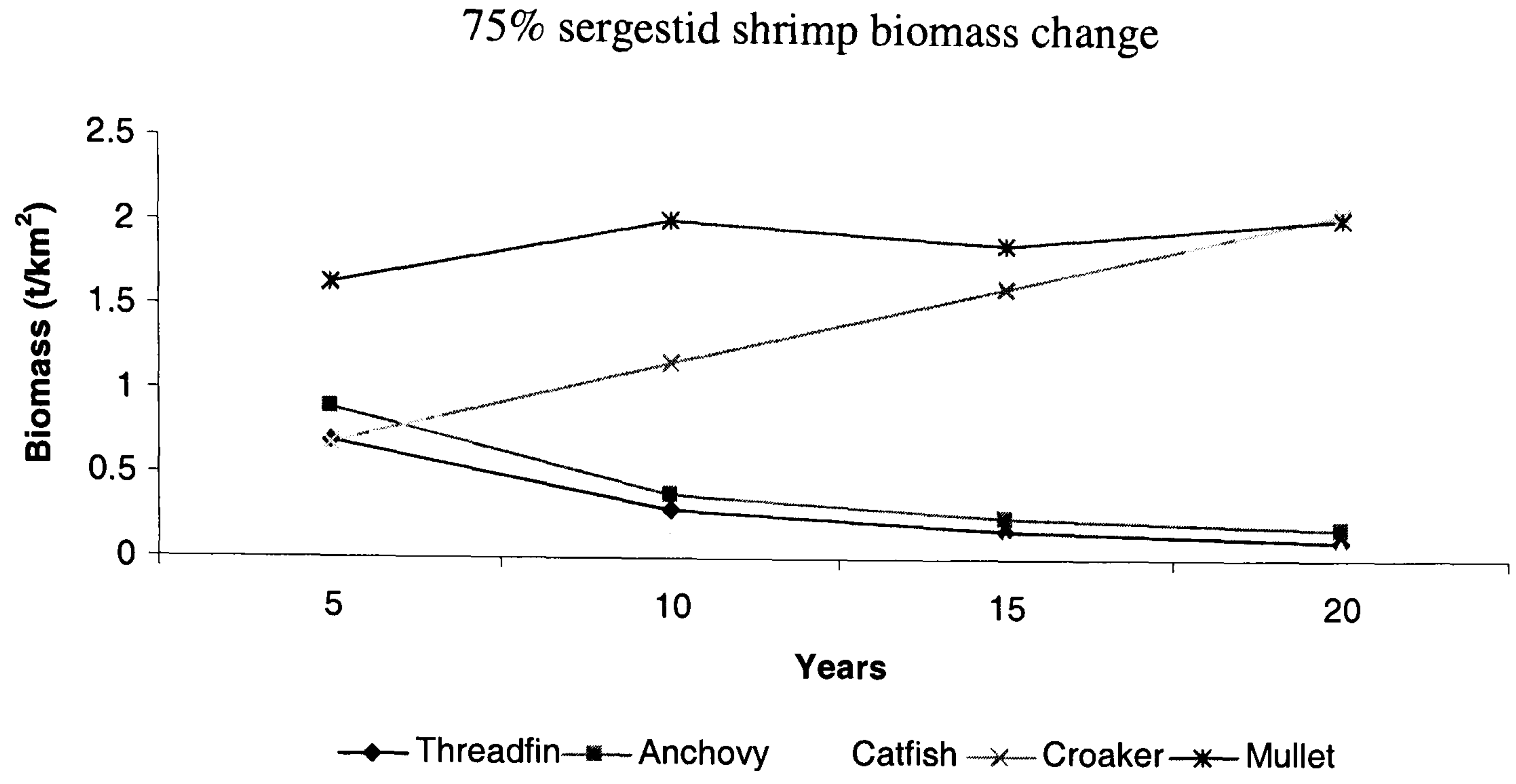
Figure 5.2 Changes in fish biomass over time at (a) 25%, (b) 50% and (c) 75% shrimp removals.


25% sergestid shrimp biomass change

(a)


50% sergestid shrimp biomass change

(b)



(c)

Figure 5.3 Changes in fish biomass over time at (a) 25%, (b) 50% and (c) 75% sergestid shrimp removals.

5.6 Discussion

The output from the simulation model suggests both direct and indirect effects of predation. Predator-prey interactions are a component of the regulation of fisheries resources, and their effects on fish resources are diverse and complex (Sanders 1995). As Weatherley (1963) concluded: ".....for the multi-species fish communities of no fixed feeding habits,.....competition would probably never be more than a fleeting problem. These fish change their diets so readily and grow satisfactorily on such a wide range of items that they are scarcely likely to suffer prolonged disadvantage from competition induced food storage". At the community and population scales, prey selection by predators, behavior of prey species, life histories (Kitchell et al. 2004), prey availability and mobility, prey abundance, prey energy content, prey size selection (Bachok et al. 2004), habitat complexity (Webster and Hard 2004) and seasonal changes (Brönmark et al. 2008) may be the factors which determine food preference of predatory fishes. Therefore, both direct and indirect predation effects are important aspects that should be kept in mind when interpreting the data presented here.

After simulation, removing shrimps over an extensive range of biomasses (25%, 50% and 75%) had similar impacts on the biomasses of other fish species; threadfin, anchovy and catfish all increased slightly and then began to decline again, whereas mullet and croaker responded quite differently, continuing to increase, at least for 20 years. These changes can be interpreted in the context of the direct and indirect interactions within the trophic network in which all these species are embedded (Chapter 4).

In terms of direct trophic effects, removal of either shrimp or sergestid shrimp might be expected to lead to reductions in biomass of those fish species which consume large quantities of shrimps. Table 4.8 (Chapter 4) shows the diets of the key fish species in this analysis. It can be seen that ~90% of the diet of anchovy is sergestid shrimp (87.6%) or shrimp (2.3%), 48.3% of the diet of catfish is shrimp, and shrimps comprise 93.6% of the diet of threadfin (77.6% sergestid shrimp, 16% shrimp). Thus, the patterns of biomass change in these three species may reasonably be accounted for by changes in their prey abundance (but see also below).

The patterns of change in mullet and croaker are more difficult to explain, but likely to be due to indirect trophic effects of shrimp removal on other interactions in the food web. For instance, the main prey of mullet is phytoplankton (53.9%) and detritus (28.3%). It is unlikely that shrimp biomass removal would lead to an increase in detritus availability for mullet, even though 100% of the diet both kinds of shrimp is detritus, because detritus is superabundant and not limiting in the Ecopath model. If the increase in mullet is due to release of their main prey, phytoplankton, then this in turn implies a reduction in predation on the phytoplankton, due to changes in biomass of other phytoplankton consumers. Inspection of Table 4.8 indicates that zooplankton predators are sardines (48.6% of diet), perchets (35.6%), ponyfish (66.7%) and bentho-pelagics (34.6%). Small increases in the abundance of all of these simultaneously might allow phytoplankton to increase, but such changes need only be small for each species and are not easy to detect in the Ecosim outputs. The increases in croaker could possibly be to due to a phytoplankton increase, although this species consumes relatively little phytoplankton (7.7%). The main prey of croaker is shrimps (29.5% shrimp, 30.3% sergestid shrimp) and given the patterns observed for anchovy, threadfin and catfish, one might expect croaker to show a similar trend to those species.

The changes in croaker and mullet imply complex indirect trophic, as well as, perhaps, competitive, interactions between all of these species. This may also in fact be true for catfish, which responded in a similar way to anchovy and threadfin to both shrimp and sergestid biomass changes, but was not recorded as eating sergestid shrimp in the original Ecopath model (Table 4.8). Further interpretation of the effects of mixtures of direct and indirect competitive interactions between these species would be very difficult, since many of the effects may be due to the cumulative effects of small changes in biomasses of many species. However, this difficulty illustrates well the need to approach the dynamics of multi-species fisheries using models like Ecopath with Ecosim. The present analysis has revealed effects of shrimp harvesting which would not have been easily thought of in advance, especially the changes in mullet and croaker, due to the ways in which direct and indirect competitive and trophic interactions spread through food webs like the Mae Klong when the system is perturbed, in this case by the harvesting of shrimps. Fisherman behaviour, such as switching from less abundant prey like threadfin and anchovy to more abundant species like croaker and mullet, would be further expected to modify the changes in the food web seen here, but these were not incorporated into the present model. The results of this exercise not only reveal the power of EwE to highlight possible unforeseen changes in multi-species fisheries, but also serve as a lesson for fisheries managers to manage their ecosystems in a multispecies way and to be cautious about the predictions of single species approaches which would not have revealed the complex interactions suggested here.

CHAPTER 6

Concluding remarks

Estuarine habitats are “nutrient traps” that support high primary productivity, which is turn promotes a high level of secondary production and high biomasses of secondary consumers, providing economic opportunities in term of fishery yields. Estuaries are often the receiving basins for major river systems which makes them vulnerable to anthropogenic influences in other parts of the catchment. Tropical estuaries are often dominated by mangrove forest, illustrating the complex food web and ecosystem dynamics which are present in these areas. To understand the nature and dynamics of exploited mangrove estuarine ecosystems, and more precisely the interactions between the species present, the development of an ecosystem-scale approach is essential.

This thesis focuses on one such tropical estuarine system, the Mae Klong Estuary in the inner Gulf of Thailand, one of the four major mangrove estuaries in the Gulf. The aims of the present study were to investigate fish assemblages in the area in order to construct mass balance models (Ecopath) for evaluating the ecosystem health of the Mae Klong and for exploring fisheries scenarios. The Mae Klong Estuary benefited from the large amount of data on its biological communities, from field and laboratory studies, that were used to construct the mass balance models.

Three separate Ecopath models were developed and used to compare the biomass, production, consumption, biomass flows, and higher order indices of ecosystem functioning of the Mae Klong Estuary in dry, hot and rainy seasons. Several higher order indices related to the ecosystem maturity indicators proposed by Odum (1969) and Ulanowicz (1986) were computed for the Mae Klong models and were compared with other coastal ecosystems around the world.

The results of those analyses indicated that the Mae Klong Estuary has a mixture of characteristics of a mature system (high total system throughput, ascendency and overhead) as well as an immature system (high PP/B and PP/R, low Finn's cycling index and mean path length), a mixture which has been encountered for several other estuarine systems.

An extension of Ecopath; Ecosim, was used to explore the effects of harvesting "experiments" on shrimp groups on the biomass of other target fish species within the system. These analyses revealed likely changes in some groups (mullet and croaker) which would have been difficult to predict from simple assumptions about species interactions and shows the power of multi-species models for fisheries planning and ecosystem management.

The modelling approach presented in this thesis- Ecopath with Ecosim- is innovative in that relatively few tropical mangrove ecosystems have been analysed in this way and because in Thailand that there has been no research in this field for coastal systems. Also, many new primary fish data were collected for this part of Thailand in order to parameterize the models. Not only is this work novel for Thailand, but it has also allowed Thai mangrove ecosystems to be placed within a broader, global context.

The Thai mangrove estuarine ecosystem is threatened by many factors such as shrimp farms, mining, climate change, port construction, tourism, infrastructure development and pollution of local waters (IUCN 2007). In addition, overfishing is responsible for a wide variety of impacts on fish communities, including changes in population structure and community composition and resilience of fish to other stressors in the area (Villanueva et al. 2006). Similar pressures occur worldwide in mangrove estuarine systems making comparative studies important and the present study adds to that information base.

Information about ecosystems is generally limited and uncertain, and these constraints affect the accuracy of the ecosystem models produced, particularly their predictive power. While the Ecopath with Ecosim model is a powerful tool to evaluate the relative impacts of alternative fishing policies, there are some limitations to the modelling approach used. Ecopath provides some answers to questions about energy flow and ecosystem development, and can generate more thoughtful questions and hypotheses about a specific system or component. As in any Ecopath model, confidence in the outputs are strongly related to quality of the input parameters. The biomass of a population is a function of many things including environmental conditions, prey availability and predator density (Dame and Christian 2006). The requirement of steady-state and the focus on predators consuming prey as the basis of all food web interactions are clearly stated assumptions, but limit the questions that can be addressed. In the present study, the weakness of the Mae Klong models is the uncertainty of several input parameters and diets. Whilst the fish diets have a high data pedigree, being collected from the site by myself, other taxa are poorly known, and some input parameters could not be based on local data, especially the biomass/input estimates for the lower trophic levels. Thus, there is less certainly of B, P/B and Q/B values for those taxa.

The models constructed here used 21 ecological groups, a relative small number compared to the real ecosystem and to some other existing models. Several groups could not be included in the models due to the lack of information, for example the higher predatory groups (amphibian, reptiles, marine birds and mammals). These predatory species inhabit the Mae Klong Estuary either temporarily or permanently, and they may certainly have effects upon the ecosystem. Mangrove forest was specifically not included, since there are no direct links to mangrove biomass and the main inputs to the fish system under study is detritus which is not thought to be limiting. The inclusion of predatory species may have given a more realistic picture of the Mae Klong Estuary food web and hopefully the next generation of models will be able to include these species.

My application of Ecosim, the dynamic application of Ecopath, to a small, open mangrove estuary revealed some limitations in the current version of the Ecosim software. The potential for a large influence of boundary conditions in my study area complicated the interpretation of the Ecosim results and limited the scope and realism of Ecosim scenarios that could be explored. Seasonal shifts in species composition, as transient species emigrate and immigrate, were the most difficult problem to deal with when using the Ecosim model. I was unable to easily force the seasonal patterns of juvenile movements into and out of the estuary. When considering the results of the Ecosim simulations of the Mae Klong Estuary, it must be understood that these predictions are as if the food web in Mae Klong is operating in isolation so that the biomass and production of a group from one year are the source of the population for the next year. While model simulations under such conditions are informative, more realistic simulations would have incorporated seasonal movement patterns of the foraging fish and top predator groups into and out of the system.

Walters et al. (1997) have reviewed some of the limitations and weaknesses of Ecosim, mainly the simple assumptions, such as diet relationships, the absence of complex life histories and that it does not take into account environmental variability. For my study, relatively short-term dynamics were considered and in that case Ecosim can be a useful tool in predicting qualitative directions of biomass change.

While development of an easily applied ecological model of an ecosystem is an admirable goal, the difficulties of attempting to incorporate many options for the user in one package were evident when using this software (Althauser 2003). The influence of abiotic factors or environmental variability, such as seasonal and diel temperature changes (difficult to predict in the case of the Mae Klong), salinity gradients and areas of hypoxia, were not possible to simulate in the current version of this software. Considerations beyond those of feeding interactions within the food web and patterns of productivity were possible in time-dynamic simulations. The dangers of creating the simple "black-box" into which numbers are fed and numbers come out cannot be ignored. Notwithstanding the natural limitations of broad-system modeling approaches, this model has potential to provide an accessible and useful view of the whole ecosystem for scientists, students and the general public. This approach can become a critical complement to other available assessment and management tools currently in use or being developed, and help bring us into a new era of ecosystem-based management.

As noted above, species or ecological groups should be aggregated based on functional rather than taxonomic similarity and the major challenge for this multispecies modelling was the lack of studies on the feeding ecology of some of the functional groups considered. Therefore, further investigation should be carried out on diets of such groups as one of the key priority objectives in this area. To improve the quality of the model, it would also be necessary to collect and improve the quality of data on the catch-landing statistics (that has never been collected in this area) in order to obtain routine data and precise estimations for a multispecies approach.

The research presented here for the Mae Klong Estuary should be considered simply as the results from just one modelling exercise, which may give insight to the structure and function of the ecosystem and the changes that have occurred. Alternative modelling approaches would provide support (or not) for the conclusions made here, and may offer alternative views of the ecosystem and of any changes that may have occurred. In addition, models can be constructed at different scales, e.g., larger, for the whole Gulf of Thailand, or smaller, and for areas not yet studied.

The ecosystem modelling approach, until now, has dealt mainly with ecological issues such as predator-prey relationships, fisheries management, biodiversity, etc. Now that modelling is becoming more routine, there is a need to focus on merging different fields to better understand the structure and functioning of ecosystems. For example, it seems that the role of genetic diversity of populations is an important as species diversity (Reusch and Hughes 2006). The evolution of the prey can also modify considerably predator-prey relationships (Yoshida 2006). In the context of climate change, oceanographic features are also to be considered when addressing ecosystem dynamics (Gilbert 2005). New approaches should aim to integrate these different fields so that model predictions have greater certainty and are thus useful for environmental management.

References

Abou-Seedo, F., Clayton, D.A., Wright, J.M. 1990. Tidal and turbidity effects on the shallow water fish assemblage of Kuwait Bay. Marine Ecology Progress Series 65, 213-223.

Abrantes, K., Sheaves, M. 2009. Food web structure in a near-pristine mangrove area of the Australian wet tropics. Estuarine, Coastal and Shelf Science 82, 597-607.

Adeel, Z., Pomeroy, R. 2002. Assessment and management of mangrove ecosystems in developing countries. Trees 16, 235-238.

Ahmed, M., Boonchuwongse, P., Dechboon, W., Squires, D. 2007. Overfishing in the Gulf of Thailand: Policy challenges and bioeconomic analysis. Environmental and Development Economics 12, 145-172.

Aiemsomboon, A., Tongnunui, P., Paphavasit, N. 1997. Seasonal changes in diversity and abundance of fish larvae at Klong Kone mangrove forest, Samut Songkhram Province. 10th Thailand National Mangrove Seminar, Songkhla, Thailand. 25-28 August 1997. The National Research Council of Thailand, Bangkok.

Aksornkoae, S. 1980. Ecological, management and research aspects of mangrove forests in Thailand. Proceeding of a Seminar on Southeast Asian Mangrove. November 5-7, 1980. Okinawa, Japan. Nodai Research Institute, Tokyo University of Agriculture, pp. 1-21.

Aksornkoae, S. 1993. Ecology and management of mangrove. IUCN, Bangkok. 176 p.

Aksornkoae, S. 1997. Scientific mangrove management in Thailand. In: Aksornkoae, A., Puangchit, L., Thaiutsa, B. (eds.), Tropical Forestry in the 21st Century. Vol.10: Mangrove Ecosystem. Fortop'96 International Conference, 25-28 November1996, Bangkok. Faculty of Forestry, Kasetsart University, pp. 118-126.

Aksornkoae, S. 2004. Sustainable use and conservation of mangrove forest resources with emphasis on policy and management practices in Thailand. In: Vannucci, M. (ed.), Mangrove management and conservation: Present and future. United Nations University Press, New York.

Aksornkoae, S., Priebprom, S., Saraya, A., Kongsangchai, J. 1985. Mangrove resources and the socio-economics of dwellers in mangrove forests in Thailand. In: Kunstadler, E.L., Bird, F., Sabhasri, S. (eds.), Man in the mangroves: The Socio-

Economic Situation of Human Settlements in Mangrove Forests. UN University and the National Research Council of Thailand, Bangkok, Thailand, pp.3-49.

Albaret, Jean-Jacques, Simier, M., Darboe, F.M., Ecoutin, Jean-Marc, Raffray, J., de Morais, L.T. 2004. Fish diversity and distribution in the Gambia Estuary, West Africa, in relation to environment variables. Aquatic Living Resources 17, 35-46.

Alongi, D.M. 1998. The role of soft-bottom benthic communities in tropical mangrove and coral reef ecosystems. CRC Critical Review in Aquatic Sciences 1, 243-280.

Alongi, D.M., 2002. Present state and future of the world's mangrove forest. Environmental Conservation 29, 331-349.

Althauser, L. L. 2003. An Ecopath/Ecosim analysis of an estuarine food web: Seasonal energy flow and response to river-flow related perturbations. MSc thesis, The Department of Oceanography and Coastal Sciences, Louisiana State University and Agriculture and Mechanical College.

Alverson, D.L., Larkin, P.A. 1994. Fisheries: fisheries science and management: century 21. In: Voigtlander, C.D. (ed.), The State of the World's Fishery Resources. Proceeding of the World Fishery Congress, Plenary Session. Oxford and IBH Publishing, New Delhi, pp. 150-167.

Amorim, P., Duarte, G., Guerra, M., Morato, T., Stobberup, K.A. 2004. Preliminary Eopath model of the Guinea-Bissau continental shelf ecosystem (NW-Africa). In: Palomares, M.L.D., Pauly, D. (eds.), West African Marine Ecosystems: Models and Fisheries Impacts. Fisheries Centre Research Reports 12(7). Fisheries Centre, UBC, Vancouver, pp. 95-112.

Aoki, I. 1997. Comparative study of flow-indices in lake-ecosystems and the implication for maturation process. Ecological Modelling 95, 165-169.

Árnason, E., Hernadez, U.B., Kristinsson, K. 2009. Intense habitat-specific fisheries-induced selection at the molecular *Pan* I locus predicts imminent collapse of a major cod fishery. PLos ONE 4(5), 1-14.

Arreguín-Sánchez, F. 2000. Octopus-red grouper interaction in the exploited ecosystem of the northern continental shelf of Yucatan, Mexico. Ecological Modelling 129, 119-129.

Arreguín-Sánchez, F. 2001. Toward the management of fisheries in the context of the ecosystem: the case of Mexico. EC Fisheries Cooperation Bulletin 14(1-4), 7-9.

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

Arshad, A., Nural Amin, S. M., Yu, G.T., Oh, S.Y., Bujang, J.S., Ghaffar, M. A. 2008. Population characteristics, length-weight and length-length relationships of *Acetes vulgaris* (Decapoda: Sergestidae) in the coastal waters of Pontian, Johor, Peninsular Malaysia. Journal of Biological Sciences 8(8), 1298-1303.

Avila Foucat, A.S. 2006. Ecological-economic model for integrated watershed management on Tonameca, Oaxaca, Mexico. PhD thesis, Environment Department, The University of York.

Aydin, K.Y. 2004. Age structure or functional response? Reconciling the energetics of surplus production between single-species models and Ecosim. African Journal of Marine Science 26, 289-301.

Bachok, Z., Mansor, M.I., Nordin, R.M. 2004. Diet composition and food habits of demersal and pelagic marine fishes from Terengganu waters, east coast of Peninsular Malaysia. NAGA Worldfish Center Q 27(3), 41-47.

Baird, D., Heymans, J.J. 1996. Assessment of ecosystem changes in response to fresh inflow of the Kromme River estuary, St. Francis Bay, South Africa: A network analysis approach. Water SA 22, 307-318.

Baird, D., Ulanowicz, R.E. 1989. The seasonal dynamics of the Chesapeake Bay ecosystem. Ecological Monographs 59, 329-364.

Baird, D., Ulanowicz, R.E. 1993. Comparative study on the trophic structure, cycling and ecosystem poperties of four tidal estuaries. Marine Ecology Progress Series 99, 221-237.

Baird, D., McGlade, J.M., Ulanowicz, J.M. 1991. The comparative ecology of six marine ecosystems. Philosophical Transactions: Biological Sciences 333(1266), 15-29.

Baldó, F., Drake, P. 2002. A multivariate approach to the feeding habits of small fishes in the Guadalquivir Estuary. Journal of Fish Biology 61(Supplement A), 21-32.

Barbier, E.B. 2003. Habitat-fisheries linkages and mangrove loss in Thailand. Contemporary Economic Policy 2(1), 59-77.

Barletta, M., Barletta-Bergan, A., Saint-Paul, U., Hubold, G. 2005. The role of salinity in structuring the fish assemblages in a tropical estuary. Journal of Fish Biology 66, 45-72.

Barry, J.P., Yoklavich, M.M., Cailliet, G.M., Ambrose, D.A., Antrim, B.S. 1996. Trophic ecology of the dominant fishes in Elkhorn Slough, California, 1974-1980. Estuaries 19(1), 115-138.

Bell, J.D., Pollard, D.A., Burchmore, J.J., Pease, B.C., Middleton, M.J. 1984. Structure of a fish community in a temperate tidal mangrove creek in Botany Bay, New South Wales. Australia Journal of Marine and Freshwater Research 36, 247-266.

Blaber, S.J.M. 1997. Fish and Fisheries of Tropical Estuaries. Chapman and Hall, London. 367 p.

Blaber, S.J.M. 2000. Tropical Marine Fishes: Ecology, Exploitation and Conservation. Blackwell Science, Oxford, London.

Blaber, S.J.M., Blaber, T.G. 1980. Factors affecting the distribution of juvenile estuarine and inshore fish. Journal of Fish Biology 17, 143-162.

Blaber, S.J.M., Milton, D.A., 1990. Species composition, community structure and zoogeography of fishes of mangrove estuaries in the Solomon Islands. Marine Biology 105, 259-267.

Boondao, S. 2006. Relationship between species composition and abundance of phytoplankton with zooplankton in Mae Klong Estuary, Samut Songkharm Province. MSc thesis, Fisheries Science, Department of Fishery Biology, Kasetsart University, Thailand.

Boonruang, P., Janekarn, V. 1985. Distribution and abundance of penaeid postlarvae in mangrove areas along the east coast of Phuket Island, southern Thailand. Phuket Marine Biological Center Research Bulletin 36, 1-29.

Boonruang, P., Satapoomin, S. 1997. The community structure of fish and their relation to food items in the mangrove area, Ranong Province. The 10th National Seminar on Mangrove Ecology: Mangrove Management and Conservation for the Lessons in the Twenty Years Round. 25-28 August 1997, Songkla. National Research Council of Thailand.

Boonyatumanond, R., Srilachai, S., Boonchalermkit, S., In-na, Y. 2003. Concentration of persistent organic pollutants in the coastal hydrosphere of Thailand. Capacity

Development Training for Monitoring of POPs in the East Asian Hydrosphere. 1-2 September, 2003, UNU Centre, Tokyo.

Botsford, L.W., Castilla, J.C., Peterson, C.H. 1997. The management of fisheries and marine ecosystems. Science 277, 509-515.

Brando, V.E., Ceccarelli, R., Libralato, S., Ravagnan, G. 2004. Assessment of environment effects in a shallow water basin using mass-balance models. Ecological Modelling 172, 213-232.

Briggs, M.R.P., Funge-Smith, S. 1994. Unsustainable shrimp culture-causes and potential solutions from experience in Thailand, pp. 1-31. In: Development of Strategies for Sustainable Shrimp Farming, Report to the Overseas Development Administration, Research Project R4751. Appendix 2. Stirling, UK.

Brinton, E., Newman, W. A. (eds.) 1974. Scientific results of marine investigators of the South China Sea and the Gulf of Thailand 1959-1961. NAGA report , Vol. 3 Part 1. The University of California. Scripps Institution of Oceanography La Jolla, California.

Brönmark, C., Skov, C., Broaderson, J., Nilsson, P.A., Hansson, L. 2008. Seasonal migration determined by a trade-off between predator avoidance and growth. PLoS ONE 3(4), e1957.

Brumer, J.P., 1978. Feeding ecology of four fishes from a mangrove creek in north Queensland, Australia. Journal of Fish Biology 12, 475-490.

BSRP 2004. Workshop on Ecopath modelling of Baltic Sea carbon and nutrient networks, Oct 18-22, 2004, Jurmala, Latvia. BSRP Productivity Cordination Center Pronotion Ecosystem-Based Approaches to Fisheries Conservation and LMEs.

Bucher, D., Saenger, P. 1994. A classification of tropical and subtropical Australian estuaries. Aquatic Conservation of Marine Freshwater Ecology 4, 1-19.

Bundy, A. 2001. Fishing on ecosystems: the interplay of fishing and predation in Newfoundland-Labrador. Canadian Journal of Fisheries and Aquatic Science 58, 1153-1167.

Bundy, A., 2004. Mass balance models of the eastern Scotian Shelf before and after the cod collapse and other ecosystem changes. Canadian Technical Report of Fisheries and Aquatic Sciences No. 2520. Science Brance, Marine Fish Division

Maritimes Region, Department of Fisheries and Oceans, Bedford Institute of Oceanography, Dartmouth, Nova Scotia.

Bundy, A., Pauly, D. 2001. Selective harvesting by small-scale fisheries: Ecosystem analysis of San Miguel Bay, Philippines. Fisheries Research 53, 263-281.

Burnett, W., Wattayakorn, K., Taniguchi, M., Dulaiova, H., Sojisuporn, P., Rungsupa, S., Ishitobi, T. 2007. Groundwater-derived nutrient inputs to the Upper Gulf of Thailand. Continental Shelf Research 27, 176-190.

Butler, A.J., Depers, A.M., McKillup, S.C., Thomas, D.P. 1975. The conservation of mangrove-swamps in South Australia. A Report to the Nature Conservation Society of South Australia. The University of Adelaide, Adelaide.

Carpenter, S.R. 1990. Large-scaled perturbation: opportunities for innovation. Ecology 71(6), 2038-2043.

Censi, P., Sprovieri, M., Saiano, F., Di Geronimo, S.I., Larocca, D., Placenti, F., 2007. The behavior of REEs in Thailand's Mae Klong estuary: suggestions from the Y/Ho ratios and lanthanide tetrad effects. Estuarine, Coastal and Shelf Science 71, 569-579.

Chansang, H. 1988. Coastal tin mining and marine pollution in Thailand. Ambio 17(3), 223-228.

Chapman, M.G., Underwood, A.J. 1995. Mangrove forests. In: Underwood, A.J., Chapman, M.G.(eds.) Coastal Marine Ecology of Temperate Australia. UNSW Press, Sydney, pp. 187-204.

Charuppat, T. 1998. Using Landsat-5 (TM) Imagery for monitoring the changes of mangrove area in Thailand. Forest Resources Assessment Division, Royal Forest Department, Bangkok, Thailand. 9 p.

Charuppat, T., Charuppat, J. 1997. Application of Landsat-5 (TM) of monitoring the changes of mangrove forest area in Thailand. Forest Technical Division, Royal Forest Department, Bangkok, Thailand. 69 p.

Cheevaporn, V. Menasveta, P. 2003. Water pollution and habitat degradation in the Gulf of Thailand. Marine Pollution Bulletin 47, 43-51.

Choi, J.S., Frank, K.T., Petrie, B.D., Leggett, W.C. 2005. Integrated assessment of a large marine ecosystem: A case study of the devolution of the Eastern Scotian shelf, Canada. Oceanography and marine biology: An annual review 43, 47-67.

Chong, V., Sasekumar, C., Leh, M.U.C., D'Cruz, R. 1990. The fish and prawn communities of a Malaysian coastal mangrove system, with comparison to adjacent mudflats and inshore waters. Estuarine, Coastal and Shelf Science 31, 703-722.

Chongprasith, P., Praekulvanich, E. 2003. Coastal pollution management in Thailand. Poster papers presented in Diffuse Pollution Conference, Dublin.

Chookajorn, T., Leenanond, Y., Moreau, J., Sricharoendham, B. 1994. Evolution of trophic relationships in Ubonratana reservoir (Thailand) as described using multi-species trophic model. Asian Fisheries Science 7, 201-213.

Chotiyaputta, C. 1992. Trawl survey of marine resources in the upper Gulf of Thailand from Cholburi to Sarat-tani provinces, 1989. Technical Paper No. 2/1992. Marine Resources Survey Unit, Marine Fisheries Development Centre, Department of Fisheries, Bangkok 48p.

Choudhury, J.K. 1997. Sustainable management of coastal mangrove forest development and social needs. XI World Forestry Congress, Antalya, Turkey, 13-22 October 1997.

Christensen, V. 1995. Ecosystem maturity-towards quantification. Ecological Modelling 77, 3-32.

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

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

Christensen, V., Pauly, D. (eds.) 1992b. A guild to the Ecopath II software system (version 2.1). ICLARM software 6 Manila, Philippines. ICLARM. 72 p.

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

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

Christensen,V., Pauly, D. 1998. Changes in models of ecosystems approaching carrying capacity. Ecological Applications 8(1), 104-109.

Christensen, V., Walters, C. 2004. Ecopath with Ecosim: method, capabilities and limitations. Ecological Modelling 172 (2-4), 109-139.

Christensen, V., Waters, C.J., Pauly, D. 2005. Ecopath with Ecosim: A user's guide. Fisheries Centre, University of British Columbia, Vancouver, Canada.

Christensen, V., Guénette, S., Heymans, J.J., Walters, C.J., Watson, R. Zeller, D., Pauly, D. 2003. Hundred years decline of North Atlantic predatory fishes. Fish and Fisheries 4, 1-24.

Christian, R.R., Baird, D., Luczkovich, J.J., Johnson, J.C., Scharler, U., Ulanowicz, R.E. 2005. Role of network analysis in comparative ecosystem ecology of estuaries. In: Belgrano, A., Scharler, U., Dunne, J., Ulanowicz, R.E. (eds.), Aquatic Food Webs: An Ecosystem Approach. Oxford University Press. Oxford, pp. 25-40.

Chullasorn, S. 1997. Review of small pelagic fisheries in Thailand. In: Chullasorn, S., Chotiyaputta, C. (eds.), Fishing status of Thailand. Proceeding of Regional Workshop on Responsible Fishing APFIC: WPMF/07/CR-9, Bangkok, Thailand, 24-27 June, 450 p.

Clarke, K.R., Gorley, R.N. 2001. PRIMER v5 Users Manual/Tutorial. PRIMER-E, Plymouth, 91 p.

Clarke, K.R., Warwick, R.M. 1994. Change in Marine Communities. Plymouth Marine Laboratory. 144 p.

Costanza, R., Andrade, F., Antunes, P., van den Belt, M., Boersma, D., Boesch, D.F., Catarino, F., Hanna, S., Limburg, K., Low, B., Molitor, M., Pereira, J.G., Rayner, S., Santos, R., Wilson, J., Young, M. 1998. Principles for sustainable governance of the oceans. Science 281, 198-199.

Crona, B.I., Rönnbäck, P. 2007. Community structure and temporal variability of juvenile fish assemblages in natural and replanted mangroves, *Sonneratia alba* Sm., of Gazi Bay, Kenya. Estuarine, Coastal and Shelf Science 74, 44-52.

Curran, S.R., Cruz, M. 2002. Markets, population dynamics, and coastal ecosystems. Ambio 31(4), 373-376.

Dahdouh-Guebas, F., Jayatissa, L.P., Di Nitto, D., Bosire, J.O., Lo Seen, D., Koedam, N. 2005. How effect were mangroves as a defence against the recent Tsumai. Current Biology 15(12), 1-5.

Dame, K.J. and Christian, R.R. 2006. Uncertainty and the use of network analysis for ecosystem based fisheries management. Fisheries 31, 331-341.

Daniel, P.A., Robertson, A.I. 1990.Epibenthos community data from mangrove and associated nearshore habitat from tropical northeast Queensland. Australian Institute of Marine Science. AIMS Report No. 3.

Darnell, R.M. 1962. Trophic spectrum of an estuarine community, based on studies of Lake Pontchartrain, Louisiana. Ecology 42, 553-558.

Daskalov, G.M. 2002. Overfishing drives a trophic cascade in a Black Sea. Marine Ecology Progress Series 225, 53-63.

Day, J.W., Hall, C.A.S., Kemp, W.M., Yanez-Arancibia, A. 1989. Estuarine Ecology. John Wiley and Sons, New York, USA.

Defeo, O. 1998. Testing hypotheses on recruitment, growth and mortality in exploited bivalves: an experimental perspective. In: Jamieson, J.G., Campbell, A. (eds.), Proceedings of the North Pacific Symposium on Invertebrate Stock Assessment and Management. Canadian Journal of Fisheries and Aquatic Sciences 125, 257–264.

Delos, R.M.R. 1995. Geoecology of Laguna de Bay, Philippines: long term alterations of a tropical-aquatic ecosystem 1820-1992. Dissertation zur Erlangung des Doktorgrades der Naturwissenschaften im Fachbereich Geowisswnschaften der Universität Hamburg.

Department of Fisheries (Thailand) 1997. Fisheries Statistics of Thailand 1995, No.5/1997. Fisheries Economics Devision, Department of Fisheries, Ministry of Agriculture and Cooperatives, Bangkok, Thailand.

Department of Fisheries (Thailand) 1997. The important notifications of Agriculture and Co-operatives regarding Thailand's fisheries. Division of Law and Treaties, Department of Fisheries, Thailand. September 197. (in Thai)

Department of Fisheries (Thailand) 2005. The Marine Fisheries Statistics 2002 Base on the Sample Survey. No. 34/2004. Fishery Information Technology Center, Department of Fisheries, Ministry of Agriculture and Cooperatives, Bangkok.

Department of Fisheries (Thailand) 2006. Statistics on fisheries production 2004. Ministry of Agriculture and Cooperatives, Bangkok.

De Sylva, D.P. 1985. Mektonic food webs in estuaries. In: Yañez-Arancibia, A. (ed.), Fish Community Ecology in Estuaries and Coastal Lagoons: Towards Ecosystem Integration. UNAM Press, Mexico, pp. 233-246.

Dittmar, T., Lara, R.J. 2001. Driving forces behind nutrient and organic matter dynamics in a mangrove tidal creek in North Brazil. Estuarine, Coastal and Shelf Science 52, 249-259.

Dixon, J.A., Sherman, P.B. 1991. Economics of protected areas. Ambio 20(2), 68-74.

Duarte, C.M., Cebrian, J. 1996. The fate of marine autotrophic production. Limnology and Oceanography 41, 1758-1766.

Duke, N.C., 1992. Mangrove floristic and biogeography. In: Robertson, A.I., Alongi, D.M.(eds.), Coastal and Estuarine Studies: Tropical Mangrove Ecosystems. Washington, D.C., American Geophysical Union, pp. 63-100.

Duke, N.C. 1995. Genetic diversity, distributional barrier and rafting continent-more thoughts on the evolution of mangroves. Hydrobiologia 295, 167-181.

Dulyapurk, V., Taparhudee, W., Yoonpundh, R., Jumnongsong, S. 2007. Mangrove ecosystems, communities and conflict: developing knowledge-based approaches to reconcile multiple demands. Final Report of Work Package 1: Multispecies Situation Appraisal of Mangrove Ecosystems in Thailand. Contract Number: FP6-003697.

Durongdej, S. 2000. Land use changes in coastal areas of Thailand. Proceedings of the APN/SURVAS/LOICZ Joint Conference of Climate Change and Adaptation in the Asia-Pacific Region, Japan.

Eaimsa-ard, M. and Amornchairojkul, S. 1997. The marine fisheries of Thailand, with emphasis on the Gulf of Thailand trawl fishery. In: Silvestre, G.T., Pauly, D. (eds.), Status and Management of the Tropical Coastal Fisheries in Asia. ICLARM Conf. Proc. 53, pp.85-95.

Eckman, S. 1967. Zoogeography of the sea. Sidgwick & Jackson, London.

Ecosystem Principles Advisory Panel 1996. Ecosystem-based fishery management. A report to congress by the Ecosystem Principles Advisory Panel. As mandated by the Sustainable Fisheries Act amendments to the Magnuson-Stevens Fishery Conservation and Management Act 1996.

Elliott, M., Hemingway, K.L., Costello, M.J., Duhamel, S., Hostens, H., Labropoulou, M., Marshall, S., Winkler, H. 2002. Links between fish and other trophic levels. In: Elliott, M., Hemingway, K.L. (eds.), Fish in Estuaries. Blackwell Science, Oxford. 636p.

Ellison, A.M. 2008. Managing mangroves with benthic biodiversity in mind: Moving beyond roving banditry. Journal of Sea Research 59, 2-15.

Essington, T., 2001. Precautionary approach in fisheries management: The devil is in the details. Trends in Ecology and Evolution 13, 121-122.

FAO 1995. World Fisheries: Problems and Prospects, 21st Session. FAO Fisheries Department, Rome.

FAO 2002. The state of world fisheries and aquaculture. FAO Fisheries Department, Rome.

FAO 2005. "Thailand Capture" Capture Fisheries Production. FAO database 2005.

FAO 2007. The world's mangroves. A thematic study prepared in the framework of the Global Forest Resources Assessment 2005. FAO Forestry Paper 153. FAO, Rome.

FAO 2008. Global study of shrimp fisheries. FAO Fisheries technical Paper 475. FAO, Rome.

FAO Corporate Document Repository 2001. Report of a bio-economic modeling workshop and a policy dialogue meeting on the Thai demersal fisheries in the Gulf of Thailand, Hua Hin, Thailand, 31 May-9 June 2000. Field Report F-16. Fishcode/Management-GCP/INT/648/NOR. FAO/Norway Government Cooperative Program. FAO, Rome.

FAO Species Identification Guide for Fishery Purposes 2001. The Living Marine Resources of the Western Central Pacific. FAO, Rome.

FAO, UNEP. 1980. The present state of mangrove ecossytems in Southeast Asia and the impact of pollution. South China Sea Fisheries and Coordinating Program Manial, March 1980.

FAO, UNEP. 1981. Tropical forest resources assessment project, Forest Resources of Tropical Asia. 475 p.

FAO, UNEP. 1989. Shrimp fisheries in the Bay of Bengal Programme: Marine Fishery Resources Management. FAO, UNEP. 27 p.

Faunce, C.H., Serafy, J.E. 2006. Mangroves as fish habitat: 50 years of field studies. Marine Ecology Progress Series 318, 1-18.

Fetahi, T., Mengistou, S. 2007. Trophic analysis of Lake Awassa (Ethiopia) using mass-balance Ecopath model. Ecological Modelling 201, 398-408.

Field, J.G., Wuff, F. 1989. Network analysis in marine ecology: an assessment. In: Wuff, F., Field, J.G., Mann, K.H. (eds.), Network Analysis in Marine Ecology: Methods and Applications, Coastal and Estuarine Studies. Springer-Verlag, Berlin, pp. 261-282.

Field, J.G., Wuff, F., Mann, K.H. 1989. The need to analyse ecological networks. In: Wuff, F., Field, J.G., Mann, K.H. (eds.), Network Analysis in Marine Ecology: Methods and Applications, Coastal and Estuarine Studies. Springer-Verlag, Berlin, pp.3-12.

Field, C.B., Osborn, J.G., Hoffmann, L.L., Polsenberg, J.F., Ackerly, D.D., Berry, J.A., Bjorkman, O., Held, Z., Matson, P.A., Mooney, H.A. 1998. Mangrove biodiversity and ecosystem function. Global Ecology and Biogeography Letters 7, 3-14.

Finn, J.T. 1976. Measures of ecosystem structure and function derived from analysis. Journal of Theorethical Biology 56, 363-380.

Flewwelling, P., Hosch, G. 2006. Review of the state of world marine capture fisheries: Indian Ocean. Ed Cassandra de Young. FAO, Rome.

Fogarty, M.P., Murawski, S.A. 1998. Large-scale disturbance and the structure of marine systems: fishery impacts on George Bank. Ecological and Applications 8, S6-S22.

Francis, M.P., Morrison, M.A., Leathwick, J., Walsh, C., Middleton, C. 2005. Predictive models of small fish presence and abundance in northern New Zealand harbours. Estuarine, Coastal and Shelf Science 64, 419-435.

Freire, K.M.F., Christensen, A., Pauly, D. 2008. Description of the East Brazil large marine ecosystem using trophic model. Scientia Marina 72 (3), 477-491.

Frith, D.W.R., Tantanasiriwong, B.H. 1976. Zonation and abundance of macrofauna on a mangrove shore, Phuket Islands, Southern Thailand. Phuket Marine Biological Center Research Bulletin 10, 1–34.

Froese, R., Pauly, D. (eds.). 2004. FishBase. World Wide Web electronic publication version www.fishbase.org

Fry, B., Smith, T.J. 2002. Stable isotope studies of red mangroves and filter feeders from the Shark River Estuary, Florida. Bulletin of Marine Science 70, 871-890.

Funge-Smith, S. 2005. Tsunami fisheries impact–Thailand. Asia-Pacific Fishery Commission Briefing Note Tsunami Impact on Fisheries in Thailand.

Gaedke, U., 1995. A comparison of whole-community and ecosystem approaches (biomass size distributions, food web analysis, network analysis, simulation models) to study the structure, function and regulation of pelagic food webs. Journal of Plankton Research 6, 1273-1305.

Garces, L.R., Man, A., Ahmad, A.T., Mohamad-Norizam, M., Silvestre, G.T. 2003. A trophic model of the coastal fisheries ecosystem off the west coast of Sabah and SaraWak, Malaysia. In: Silvestre, G., Garces, L., Stobutzki, I., Ahmed, M., Valmonte-Santos, R.A., Luna, C., Lachica-Alino, L., Munro, P., Christensen, V., Pauly, D. (eds.), Assessment, Management and Future Directions for Coastal Fisheries in Asian Countries. WorldFish Center Conference Proceedings 67, pp. 333-352.

Gerrodette, T., Dayton, P.K., Macinko, S., Fogarty, M.J. 2002. Precautionary management of marine fisheries: Moving beyond burden of proof. Bulletin of Marine Science 70, 657-668.

Gibson, R.N., Robb, L., Burrows, M.T., Ansell, A.D. 1996. Tidal, diel and longer term changes in the distribution of fishes on a Scottish sandy beach. Marine Ecology Progress Series 130, 1–17.

Giesen, W., Wulffraat, Zieren, M, Scholten, L. 2007. Mangrove Guildbook for Southeast Asia. FAO and Wetlands International, Dharmasarn Co., Ltd. 186 p.

Gillbert, D., Sundby, B., Gobeil, C., Mucci, A., Tremblay, G.H. 2005. A seventy-two-year record of diminishing deep-water oxygen in the St. Lawrence estuary: The northwest Atlantic connection. Limnology and Oceanography 50: 1654-1666.

Giri, C., Zhu, Z., Tieszen, L.L., Singh, A., Gillette, S., Kelmelis, A. 2008. Mangrove forest distribution and dynamics (1975-2005) of the tsunami-affected region Asia. Journal of Biogeography 35, 519-528.

Grima, A. P. L., Berkes, F. 1989. Access, rights to use and management. In: Berkes, F. (ed.), Common Property Resources Ecology and Community Based Sustainable Development. Belhaven Press, London.

Gulland, J.A. 1988. Fish population dynamics, 2nd ed. Wiley, London, UK, 422 p.

Hajisamae, S., Chou, L.M. 2003. Do shallow water habitats of impacted coastal strait serve as nursery grounds for fish fishes? Estuarine, Coastal and Shelf Science, 56(2), 281-290.

Hajisamae, S., Chou, L.M., Ibrahim, S. 2003. Feeding habits and trophic organization of the fish community in shallow waters of an impacted tropical habitat. Estuarine, Coastal and Shelf Science 58, 89-98.

Hajisamae, S., Chou, L.M., Ibrahim, S. 2004. Feeding habits and trophic relationships of fishes utilizing and impacted coastal habitat, Singapore. Hydrobiologia 520, 61-71.

Hajisamae, S., Yeesin, P., Chaimongkol, S. 2006. Habitat utilization by fishes in a shallow, semi-enclosed estuarine bay in southern Gulf of Thailand. Estuarine, Coastal and Shelf Science 68, 647-655.

Hajisamae,S., Yeesin, P., Ibrahim,S., and Sirimontraporn, P. 1999. Abundance and diversity of juvenile fishes in Saiburi estuary, Gulf of Thailand. Songklanakarin. Journal of Science and Technology 21, 265-275.

Hall, S.J. 1999. The effects of fishing on marine ecosystems and communities. Blackwell Science, Oxford, England.

Halliday, I.A., Young, W.R. 1996. Density, biomass and species composition of fish in a subtropical *Rhizophora stylosa* mangrove forest. Marine and Freshwater Research 47, 609-615.

Havanond, S. 1997. Mangrove forest conservation in Thailand. Biological Bulletin. NTNU 32(2), 97-102.

Haywood, M.D.E., Heales, D.S., Kenyon, R.A., Loneragan, N.R., Vance, D.J. 1998. Predation of juvenile tiger prawns in a tropical Australian estuary. Marine Ecology Progress Series 162, 201-214.

Heymans, J.J., Baird, D. 2000. Network analysis of the northern Benguela ecosystem by means of NETWRK and ECOPATH. Ecological Modelling 131, 97-119.

Heymans, J.J., Ulanowicz, R.E., Bondavalli, C. 2002. Network analysis of the South Florida Everglades graminoid marches and comparison with nearby cypress ecosystem. Ecological Modelling 149, 5-23.

Higashi, M., Patten, C.B., Burns, P.T. 1993. Network trophic dynamics: The models of energy utilization in ecosystems. Ecological Modelling 66, 1-42.

Hindell, J.S., Jenkins, G.P., 2004. Spatial and temporal variability in the assemblage structure of fishes associated with mangroves (*Avicennia marina*) and intertidal mudflats in temperate Australian embayments. Marine Biology 144, 385-389.

Hofmann, E.E., Powell, T.M., 1998. Environmental variability effect on marine fisheries: Four case histories. Ecological Applications 8, S23-S32.

Hossain, Md. Z., Lin, C.K. 2001. Diversified uses of abandoned shrimp ponds. ITCZM Monograph Series 2001(5), 1-23.

Huitric, M., Folke, C., Kautsky, N. 2002. Development and government policies of the shrimp farming industry in Thailand in relation to mangrove ecosystems. Ecological Economics 40, 441-455.

Hungspreugs, M., Dharmvanij, S., Utoomprurkporn, W., Yuangthong, C., Sompongchaiyakul, P. 1987. The Chemistry of Mae Klong River, Sri Nakarind Dam and Khao Laem Dam. Report submitted to the National Environment Board, Thailand, November, 1987. (in Thai)

Hunter, M.D., Price, P.W. 1992. Playing chutes and ladders: heterogeneity and the relative roles of bottom-up and top-down forces in natural communities. Ecology 73(3), 725-732.

Hynes, H.B.N. 1950. The food of freshwater sticklebacks (*Gasterosteus aculeatus* and *Pygosteus pungitius*), with a review of method used in studies of the food of fishes. Journal of Animal Ecology 19, 36-58.

Ikejima, K., Tongnunui, P., Medej, T., Taniuchi, T. 2003. Juveniles and small fishes in a mangrove estuary in Trang province, Thailand: Season and habitat differences. Estuarine, Coastal and Shelf Science 56, 447-457.

Intong, S., Chotiyaputta, C., Anukul, T. 1993. Otter board trawl surveyed in the inner Gulf of Thailand. In: Chullasorn, S., Chotiyaputta, C. (eds.). Fishing status of Thailand. Proceeding of Regional Workshop on Responsible Fishing, Bangkok, Thailand, 24-27 June. 450 p.

International Union for Conservation of Nature and Natural Resources (IUCN) 2007. Policies and emerging issues in coastal management in Thailand. The World Conservasion Union, Thailand Programme.

Jacobi, M.C., Schaeffer-Novelli,Y. 1990. Oil spills in mangroves: a conceptual model based on long-term field observation. Ecological Modelling 52, 53-59.

Janekarn, V. 1993. A review of larval fish distribution and abundance in the Andamar Sea, Thailand. Phuket Mar. Biol. Cent. Spec. Publ. 12, 123-130.

Janekarn, V., Boonruang, P. 1986. Composition and occurrence of fish larvae in mangrove areas along the east coast of Phuket Island, western Peninsular, Thailand. Phuket Marine Biological Center Research Bulletin 44, 1-22.

Janekarn, V., Sawangarreruk, S. 1987. Occurrence of fish larva along the west coast of Thailand (Phuket, Satul). Proceeding of the Seminar on Fisheries. Department of Fisheries, Thailand.

Janekitkarn, S., Premcharoen, S. and Vitheesawat, P. 1999. The estuarine fish of the Tha Chin River, Samut Sakorn, Thailand. In: Séret, B., Sire, J.-Y. (eds.). Proceedings of the 5th Indo-Pacific Fish Conference, Nouméa, 1997. Soc. Fr. Ichthyol., Paris, pp. 57-64.

Janekitkosol, W., Somchanakit, H., Eiamsa-ard, M., Supongpan, M. 2003. Strategic review of the fishery situation in Thailand. In: Silvestre, G., Garces, L., Stobutzki, I., Ahmed, M., Valmonte-Santos, R.A., Luna, C., Lachica-Alino, L., Munro, P., Christensen, V., Pauly, D. (eds.). Assessment, Management and Future Directions for Coastal Fisheries in Asian Countries. WorldFish Center Conference Proceedings 67, pp. 915-956.

Janjua, M.Y. 2007. Application of Ecopath with Eosim on trophic network of Lake Annecy with reference to fish community. Internalship Researh Report Mater 2 Research, Biodiversity and Ecosystem Functioning, Universite Blaise Pascal.

Jennings, S., Kaiser, M.J. 1998. The effects of fishing on marine ecosystems. Advances in Marine Biology 46, 28-46.

Jepsen, D.B., Winemiller, K.O. 2002. Structure of trophic river food webs revealed by stable isotope ratios. Oikos 96, 46-55.

Johnson, J. C., Borgatti, S.P., Luczkovich, J.J., Everett, M.G. 2001. Network role analysis in the study of food webs: An application of regular role coloration. Journal of Social Structure, JoSS: Volumn 2.

Jørgensen, S. E. 1994. Fundamental of ecological modelling. 2nd edition. Developments in Environmental Modelling 19. Elseviar Science B.V. Molenwerf P.O. Box 211, 1000 VE Amsterdam, The Netherlands. ISBN: 0-444-81572-4. 628 p.

Juntarachote, K. 1984. Study on economic return of small scale capture fisheries and coastal aquaculture. Current Technical Paper No. 30. Training Department, Southeast Asian Fisheries Development Center.

Jutagate, T., Mattson, N.S., Moreau, J., Sricharoendham, B., Kumsri, M., 2002. Ecosystem in Sirinthorn and Nam Hgum Reservoirs: a comparison. Kasetsart University Fisheries Research Bulletin 24, 1-14.

Kavanagh, P. Newlands, N., Christensen, V., Pauly, D. 2004. Automated parameter optimization for Ecopath ecosystem models. Ecological Modelling 172, 141-149.

Kellecher, K. 2005. Discards in the world's marine fisheries- an update. FAO Fisheries Technical Paper 470. Rome, FAO.

Kennish, M.J. 1994. Practical handbook of marine science. 2nd ed. CRC Press. Boca Raton. 566 p.

Kennish, M.J. 2002. Environmental threats and environmental futures of estuaries. Environmental Conservation 29 (1), 78–107.

Khemakorn, P., Kongprom, A., Dechboon, W., Supongpan, M. 2005. Trash fish: The link between capture fisheries and aquaculture in Thailand. In: Regional Workshop on low Value and "Trash Fish" in the Asia-Pacific Region. Hanoi, Viet Nam, 7-9 June 2005.

Khongchai, N., Vibunpant, S., Eiamsa-ard, M., Supongpan, M. 2003. Preliminary analysis of demersal fish assemblages in coastal waters of the Gulf of Thailand, In: Silvestre, G., Garces, L.,Stobutzki, I., Ahmed, M., Valmonte-Santos, R.A., Luna, C., Lachica-Alino, L., Munro, P., Christensen, V., Pauly, D. (eds.). Assessment, Management and Future Directions for Coastal Fisheries in Asian Countries. WorldFish Center Conference Proceedings 67. pp. 249-261.

Kimani, E.N., Mwatha, K.G., Wakwabi, E.O., Ntiba, J.M., Okoth, B.K. 1996. Fishes of the shallow tropical mangrove forest, Gazi, Kenya. Marine and Freshwater Research 47, 857-868.

Kitchell, J.F., Eby, L.A., He, X., Schindler, D.E., Wright, R.A. 2004. Predator-prey dynamics in an ecosystem context. Journal of Fish Biology 45(Supplement A), 209-226.

Knadler, C.E., Jr. 2008. Models of a predator-prey relationship in a closed habitat. In: Mason, S.J., Hill, R.R., Mönch, L., Rose, O., Jefferson, T., Fowler, J.W. (eds.). Proceeding of the 2008 Winter Simulation Conference.

Kongprom, A., Khemakorn, P., Eiamsa-ard, M., Supongpan, M. 2003. Status of demersal fishery resources in the Gulf of Thailand. In: Silvestre, G., Garces, L.,Stobutzki, I., Ahmed, M., Valmonte-Santos, R.A., Luna, C., Lachica-Alino, L., Munro, P., Christensen, V., Pauly, D. (eds.). Assessment, Management and Future Directions for Coastal Fisheries in Asian Countries. WorldFish Center Conference Proceedings 67, pp. 137-152.

Kongsangchai, J. 1994. Conservation of mangroves into other uses in Thailand. Proceeding of the workshop on International tropical Timber Organization Project, Yokohama, Japan.

Kongsangchai, J., Prayoonsitti, T. 1990. Vertebrate species in the mangrove forests of Thailand (excluding fish). RFD Report. Bangkok. 37p.

Kröger, S., Law, R. J. 2005. Sensing the sea. Trends in Biotechnology 23, 250-256.

Laegdsgaard, P., Johnson, C. 1995. Mangrove habitats as nurseries: Unique assemblages of juvenile fish in subtropical mangroves in eastern Australia. Marine Ecology Progress Series 126, 67-81.

Laegdsgaard, P., Johnson, C., 2001.Why do juvenile fish utilize mangrove habitats? Journal of Experimental Marine Biology and Ecology 257, 229-253.

Lai, H.C. 1984. A review of oil spills with special reference to mangrove environment. In: Lai, H.C., Feng, M.C. (eds.), Fate and Effects of Oil in the Mangrove environment. University of Sains, Malaysia, Penang, pp. 5-19.

Landesmann, L. 1994. Negative impacts of coastal aquaculture development. World Aquaculture 25(2), 12-17.

Laroche, J., Baran, E., Rasoanandrasana, N.B. 1997. Temporal patterns in a fish assemblage of a semiarid mangrove zone in Madagascar. Journal of Fish Biology 51, 3-20.

Layman, C.A., Silliman, B.R., 2002. Preliminary survey and diet analysis of juvenile fishes of an estuarine creek on Andros island, Bahamas. Bulletin of Marine Science 70 (1), 199-210.

Lear, R., Turner, T. 1977. Mangroves of Australia. University of Queensland Press.

Lekagul, B., McNeely, J.A. 1977. Mammals of Thailand. Association for the Conservation of Wildlife, Bangkok.

Leloup, F. A., Desroy, N., Le Mao, P., Pauly, D., Le Pape, O. 2008. Interactions between a natural food web, shellfish farming and exotic species : The case of Bay of Mont Saint Michel (France). Estuarine, Coastal and Shelf Science 76, 111-120.

Le Pape, O., Menesguen, A. 1997. Hydronamic prevention of eutrophication in the Bay of Brest (France). A modelling approach. Journal of Marine Systems 12, 171-186.

Le Pape, O., Jean, F., Menesguen, A. 1999. Pelagic and benthic model coupling, application to the Bay of Brest, a semi-enclosed zone of western Europe. Marine Ecology Progress Series 189, 135-147.

Lewmanomont, K. 1976. Algal flora of mangrove forests. Proceedings of the First Thai National Seminar on Mangrove Ecology, Phuket, 10–15 January 1976. National Research Council of Thailand, Part 2, Vol. 1: 202–215 (in Thai, English abstract)

Ley, J.A., McIvor, C.C., Mantague, C.L. 1994. Food habitats of mangrove fishes: a comparison along estuarine gradients in northeastern Florida Bay. Bulletin of Marine Science 54(3), 881-899.

Lijie, D., Shiyu, L., Yu, L., Tao, J., Failler, P. 2008. An application of the Ecopath with Ecosim model to the Pearl river delta coastal sea. Acta Scientiarum Naturalium (under review).

Lin, H.J., Shao, K.T. 1999. Seasonal and diel changes in a subtropical mangrove fish assemblage. Bulletin of Marine Science 65, 775-794.

Lin, H.J., Shao, K.T., Jan, R. Q., Hsieh, H. L., Chen, C. P., Hsieh, L.Y., Hsiao, Y. T. 2007. A trophic model for the Danshuei River Estuary, a hyponic estuary in northern Taiwan. Marine Pollution Bulletin 54, 1789-1800.

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

Link, J.S. 2002. What dose ecosystem-based fisheries management mean? Fisheries 27(4), 18-21.

Little, M.C., Reay, P.J., Grove, S.J. 1988. Distribution gradients of ichthyoplankton in an east African mangrove creek. Estuarine, Coastal and Shelf Science 26, 669-677.

Little, M.C., Reay, P.J., Grove, S.J. 1988. The fish community of an East African mangrove creek. Journal of Fish Biology 32, 729-747.

Livingston, R.J. 2002. Trophic Organization in Coastal Systems. CRC Press, Boca Raton, FL, USA.

Loneragen, N.R., Bunn, S.E., Kellaway, D.M. 1997. Are mangroves and seagrasses sources of organic carbon for penaeid prawns in a tropical Australian estuary? A multiple stable-isotope study. Marine Biology 130, 289-300.

Loneragen, N.R., Potter, I.C., Lenanton, R.C.J., Caputi, N. 1986. Spatial and seasonal differences in the fish fauna of the shallows in a large Australian estuary. Marine Biology 92, 575-586.

Longonje, S. 2008. The Cameroon forest ecosystem: Ecological and environmental dimensions. PhD thesis, Environment Department, The University of York.

Lotka, A. J. 1925. Elements of physical biology. Baltimore: Williams & Wilkins Co.

Ludwig, D., Hilborn, R., Walters, C. 1993. Uncertainty, resource exploitation, and conservation: Lesson from history. Science 260, 17-36.

Lugo, A.E., Snedaker, S.C. 1974. The ecology of mangroves. Annual Review of Ecology and Systematics 5, 39-64.

Lugo, A.E., Brown, S., Brinson, M.M. 1990. Concepts in wetland ecology. In: Lugo, A.E., Brinson, M., Brown, S. (eds.). Ecosystem of the world. 15 Forested Wetlands. Amaterdam: Elevier Science, pp. 53-85.

Macintosh, D.J. 1996. Mangrove and coastal aquaculture; doing something positive for the environment. Aquaculture Asia 1(2), 3-8.

Macintosh, D.J., Phillips, M.J. 1992. Environmental issues in shrimp farming, In: De Saram, H., Singh, T. (eds.). Shrimp '92: Proceeding of the 3rd Global Conference on Shrimp Industry (Hong Kong), pp. 118-145.

Macintosh, D.J., Ashton, E.C., Havanon, S. 2002. Mangrove rehabilitation and intertidal biodiversity: A study in the Ranong mangrove ecosystem, Thailand. Estuarine, Coastal and Shelf Science 55, 331-345.

Manickchand-Heileman, S., Mendoza-Hill, J., Kong, A.L., Arocha, F., 2004. A trophic model for exploring possible ecosystem impacts of fishing in the Gulf of Paria, between Venezuela and Trinidad. Ecological Modelling 172, 307-322.

Manson, F.J., Loneragan, N.R., Skilleter, G.A., Phinn, S.R. 2005b. An evaluation of the evidence for linkages between mangroves and fisheries: A synthesis of the literature and identification of research directions. Oceanography and Marine Biology Annual Review 43, 483-513.

Manson, F.J., Loneragan, N.R., Harch, B.D., Skilleter, G.A., Williams L. 2005a. A broad-scale analysis of links between coastal fisheries production and mangrove extent: A case study for northeastern Australia. Fisheries Research 74, 60-85.

Marchant, R. 1982. Seasonal variation in the macroinvertebrate fauna of billabongs along Magela Creek, Northern Territory. Australian Journal of Marine and Freshwater Research 33, 329-342.

Marie-Bozec, Y., Gascue, D., Kulbicki, M. 2004. Trophic model of lagoonal communities in a large open atoll (Uvea, Loyalty islands, New Caledonia). Aquatic Living Resources 17, 151-162.

Mbabazi, D.,Orach-Meza, F.L., Makanga, B., Hecky, R.E., Balirwa, J.S., Ogutu-Ohwayo, R., Verburg, P., Namulemo, G., Muhumuza, E., Luyiga, J. 2004. Trophic structure and energy flow in fish communities of two lakes of the Lake Victoria basin. Uganda Journal of Agricultural Sciences 9, 348-359.

McCann, K.S. 2000. The diversity-stability debate. Nature 405, 228-233.

McIsaac, G.F., David, M.B., Gertner, G.Z., and Goolsby, D.A. 2001. Eutrophication: Nitrate flux in the Mississippi River. Nature 414, 166-167.

Meemeskul, Y. 1982. Species and size composition of trash fish from otter board trawl monitoring survey by Pramong 2 and Pramong 9 in the Gulf of Thailand, 1980. In: Chullasorn, S., Chotiyaputta, C. (eds.). Fishing status of Thailand. Proceeding of Regional Workshop on Responsible Fishing, Bangkok, Thailand, 24-27 June 1980. 450 p.

Menasveta, P. 1976. Ecology of fish population in the coastal area of Bang Pra, Chonburi. Proceeding of the First Thai National Seminar on Mangrove Ecology. Phuket Marine Biological Centre. 10-15 January A76. Part 2 Vol. 2. National Research Council of Thailand, pp. 346-357.

Metcalf, S.J., Dambacher, J.M., Hobday, A.J., Lyle, J.M. 2008. Importance of trophic information, simplification and aggregation error in ecosystem models. Marine Ecology Progress Series 360, 25-36.

Meteorological Department 1987. Statistics on the Climate of Thailand in Period 30 year (1956-1985). Ministry of Communication. Bangkok, Thailand.

Micheli, F., 1999.Eutrophication, fisheries, and consumer-resource dynamics in marine pelagic ecosystems. Science 285, 1396-1398.

Miles, D.H., Kokpol, U., Chittawong, V., Tip-Pyang, S., Tunsuwan, K., Nguyen, C. 1998. Mangrove forests- The importance of conservation as a bioresource for ecosystem diversity and utilization as a source of chemical constituents with potential medicinal and agricultural value. Pure and Applied Chemistry 70(11), 2113-2121.

Minello, T.J., Able, K.W., Weinstein, M P., Hays, C.G. 2003. Salt marshes as nurseries for nekton: testing hypotheses on density, growth and survival through meta-analysis. Marine Ecology Progress Series 246, 39-59.

Mitsch, W.J., Gosselink, J.G. 2000. Wetland. 3rd Edition. John Wiley and Sons, Inc.

Mohamed, K.S., Zacharia, P.U., Muthiah, C., Abdurahiman, K.P., Nayak, T.H. 2005. A trophic model of the Arabian sea ecosystem off Karnataka and simulation of fishery yields for its multigear fisheries. CMFRI Publication. 65 p.

Monkolprasit, S. 1983. Fish in mangrove and adjacent area. UNDP/UNESCO regional Project Training and Research Pilot Programme on the Mangrove Ecosystems of Asia and Oceania Ras/79/062/E110/13: 1st Training Course-Introduction to Mangrove Ecosystem, 16 p. Thailand, 2-30 March 1953. Bangkok: UNESCO.

Monkolprasit, S. 1994. Fish composition and food habits in mangrove forests at Phang-Nga Bay and Ban Don Bay, Thailand. Kasetsart University Fishery Research Bulletin. 20, 1-21.

Morissette, L. 2007. Complexity, cost and quality of ecosystem models and their impact on resilience: a comparative analysis, with emphasis on marine mammals and the Gulf of St. Lawrence. PhD thesis, Zoology, University of British Columbia, Vancouver BC, Canada.

Mustafa, M.G. 2003. Trophic model of the coastal ecosystem in the water of Bangladesh, Bay of Bengal. In: Silvestre, G., Garces, L., Stobutzki, I., Ahmed, M., Valmonte-Santos, R.A., Luna, C., Lachica-Alino, L., Munro, P., Christensen, V., Pauly, D. (eds.), Assessment, Management and Future Directions for Coastal Fisheries in Asian Countries. WorldFish Center Conference Proceedings 67, pp. 263-280.

Nabhitabhata, J. 1982. Ecological studies of birds in mangrove forests, Songkhla Lake, Thailand. 4th National Seminar on Mangrove Ecosystems. 7-11 July 1982, Surat Thani, Thailand. (in Thai)

Nabhitabhata, J. Chan-ard, T., Chuaynkern, Y. 2004. Amphibians and Reptiles in Thailand. Office of the Environmental Policy and Plannings, Bangkok.

NACA, 1996. A survey of water pollution source from coastal aquaculture. Draft final report Network of Aquaculture Centres in Asia-Pacific, Bangkok, Thailand. 183 pp.(exc. Appendices).

Nagelkerken, I., Faunce, C.H. 2007. Colonisation of artificial mangroves by reef fishes in a marine sea space. Estuarine, Coastal and Shelf Science 75, 417-422.

Nagelkerken, I., van der Velde, G. 2004a. Are Carribbean mangroves important feeding grounds for juvenile reef fish from adjacent seagrass beds? Marine Ecology Progress Series 274, 143-151.

Nagelkerken, I., van der Velde, G. 2004b. Relative importance of interlinked mangroves and seagrass beds as feeding habitats for juvenile reef fish on a Caribbean island. Marine Ecology Progress Series 274, 153-159.

Nagelkerken, I., van der Velde, G., Gorissen, M. W., Meijer, G. J., van't Hof, T., den Hartog, C. 2000. Importance of mangroves, seagrass beds and the shallow coral reef as a nursery for important coral reef fishes, using a visual census technique. Estuarine, Coastal and Shelf Science 51, 31-44.

Nagelkerken, I., Kleijnen, S., Klop, T., van den Brand, R., de la Moriniere, E.C., van der Velde, G. 2001. Dependence of Carribean reef fishes on mangroves and seagrass beds as nursery habitats: A comparison of fish faunas between bays with and without mangroves/seagrass beds. Marine Ecology Progress Series 214, 225-235.

Nagelkerken, I., Roberts, C.M., van der Velde, G., Dorenbosch, M., van Riel, M.C., de la Morinere. E.C., Nienhuis, P.H. 2002. How important are mangroves and seagrass beds for coral-reef fish? The nursery hypothesis tested on an island scale. Marine Ecology Progress Series 244, 299-305.

Nakthon, N. 1992. Marine territory of Thailand and neighbouring countries. Hydrograph Department, Royal Thai Navy.

Naiyanetr, P. 1979. The uca crabs in Thailand. 3rd National Seminar on the Mangrove Forest Resources. Prince of Songkla University, 22–25 April 1979. (abstract)

National Economic and Social Development Board (NESDB) 1997. Public disasters management in Thailand. NESDB, UNEP, and Songkhla Makarin University, Bangkok. (in Thai)

National Research Council of Thailand (NRCT) 1999. Sustaining Marine Fisheries, Report of committee on ecosystem management for sustainable marine fisheries, ocean studies board, 168 p.

National Research Council of Thailand (NRCT) 2002. Bibliography of mangrove research in CDROM 2002. (in Thai).

National Statistic Office and Department of Fisheries 1997. 1995 Marine fishery cencus, coastal zone 3. Bangkok: Karnsasana.

Naval Hydrographic Department 1980. Bathymetric map series of the Gulf of Thailand (between 1921-1973). Royal Thai Navy, Hydrographic Department, Thailand.

Nee, S. 1990. Community construction. Trends in Ecololgy and Evolution 5(10), 337-339.

Neelasri, K. 1981. Analysis of the observed current during the inter-monsoon period. In: the 2nd Seminar on Water Quality and Living Resources in Thai Water, pp. 57-63.

Neira, F.J., Potter, I.C., 1992. The ichthyoplankton of a seasonally closed estuary in temperate Australia. Does an extended period of opening influence species composition? Journal of Fish Biology 41, 935-953.

Newton, G.M. 1996. Estuarine ichthyoplankton ecology in relation to hydrology and zooplankton dynamics in a salt-wedge estuary. Marine and Freshwater Research 47, 99-111.

Nilvanich., K. 1999. Community structure of prawns in Tha Chin Mangrove Estuary, Samut Sakhon Province. Master thesis, Department of Marine Science, Faculty of Science, Chulalongkorn University, Thailand.

Niquil, N., Arias-Gonzalez, J.E., Delesalle, B. Ulanowicz, R.E. 1999. Characterization of the planktonic food web of Takapoto Atoll Lagoon, using network analysis. Oecologia 118, 232-241.

NOAA, 2007. Ecopath modeling: precursor to an ecosystem approach to fisheries management. Retrieved March 1st, 2007, from http://www.celebrating200years.noaa.gov.

Nurhakim, S. 2003. Marine fisheries resources of the north coast of Central Java, Indonesia: An ecosystem analysis. In: Silvestre, G., Garces, L., Stobutzki, I., Ahmed, M., Valmonte-Santos, R.A., Luna, C., Lachica-Alino, L., Munro, P., Christensen, V., Pauly, D. (eds.), Assessment, Management and Future Directions for Coastal Fisheries in Asian Countries. WorldFish Center Conference Proceedings 67, pp. 299-312.

Odum, W.E. 1969. The strategy of ecosystem development. Science 104, 262-270.

Odum, W.E. 1971. Fundamentals of ecology. W.B. Saunders Co., Philadelphia.

Odum, W.E., Heald, E.J. 1972. Trophic analyses of an estuarine mangrove community. Bulletin of Marine Science 22, 671-737.

OEPP (Office of the Environmental Policy and Planning) Ministry of Science and Technology, Thailand 1994. The environmental management of coastal aquaculture: A case study of shrimp culture in Samut Sakhon and Chantaburi provinces. Network of Aquaculture Centres in Asia-Pacific, Bangkok, Thailand.

Okey, T. A. 2004. Shifted community states in four marine ecosystems: some potential mechanisms. Ph.D. thesis, Department of Zoology, The Univeristy of British Columbia. 173 p.

Oksanen, L. 1983. Trophic exploitation and Artic phytobiomass patterns. The American Naturalist 122, 45-52.

Oksanen, L., Fretwell, S.D., Arruda, J., Niemela, P. 1981. Exploitation ecosystems in gradients of primary productivity. The American Naturalist 118, 240-261.

Omori, M. 1977. Distribution of warm ephiplanktonic shrimps of the genera *Lucifer* and *Acetes* (Macura, Penaeidea and Sergestidae). In: Proceeding of the symposium on warm water zooplankton (Goa: Special publication NIO), pp. 1-12.

Ong, J.E. 1995. The ecology of mangrove conservation and management. Hydrobiologia 295, 343-351.

Oosterveer, P., Kamolsiripichaiporn, S., Rasiah, R. 2006. The "Greening" of industry and development in Southeast Asia: Perspectives on industrial transformation and environmental regulation: Introduction. Environment, Development and Sustainability 8(2), 217-227.

Ortiz, M. 2001. Holistic modelling of a subtidal benthic ecosystem of northern Chile (Tongoy bay), to improve the knowledge and understanding of its structure and function: assessing the effects of intensive fisheries upon different invertebrates and algal species. PhD thesis, Universität Bremen, Deutschland.

Ortiz, M., Woff, M. 2002. Dynamical simulation of mass-balance trophic models for benthic communities of north-central Chile: assessment of resilience time under alternative management scenarios. Ecological Modelling 148, 277-291.

OSB (Ocean Studies Board) 2006. Dynamic changes in marine ecosystems: Fishings, foodwebs, and future options. Committee on Ecosystem Effects of Fishings: Phase II-Assessments of the Extent of Change and the Implications for Policy. Ocean Studies Board, Division on earth and Life Studies, National Research Council of the National Academies. The National Academies Press, Washington, D.C.

Pahl-Wostl, C. 1993. Food webs and ecological networks across temporal and spatial scales. Oikos 66, 415-432.

Panjarat, S. 2008. Sustainable fisheries in the Andaman sea coast of Thailand. The United Nations-Nippon Foundation Fellowship Programme 2007-2008. Division for Ocean Affairs and the Law of the Sea, Office of Legal Affairs, The United Nations, New York.

Patten, C.B. 1995. Network integration of ecological extremal priciples: exergy, emergy, power, ascendency and indirect effects. Ecological Modelling 79, 7-89.

Patten, C.B., Higashi, M., Burns, P.T. 1990.Trophic dynamics in ecosystems networks: significance of cycles and storage. Ecological Modelling 52, 1-28.

Pauly, D. 1979. Theory and management of tropical multispecies stocks: a review, with emphasis on the southeast Asian demersal fisheries. ICLARM studies and reviews, 1. International Center for Living Aquatic Resources Management: Manila, Philippines. iv, 35 p.

Pauly, D. 1988. Fisheries research and the demersal fisheries of southest Asia. In: Gulland, J.A. (ed.), Fish population dynamics, 2nd ed. Wiley/Interscience, Chichester, UK. pp.329-348.

Pauly, D. 1996. Fleet-operational, economic and cultural determinants of by-catch uses in southeast Asia. In: Solving By-catch: Considerations for Today and Tomorrow.Report No. 96-03. Sea Grant College Program, University of Alaska, Fairbanks, AK, pp.285-288.

Pauly, D., Christensen,V., Waters, C.J. 2000. Ecopath, Ecosim and Ecospace as tools for evaluating ecosystem impacts of fisheries. ICES Journal of Marine Science 57, 607-709.

Pauly, D., Christensen,V., Dalsgaard, J., Froese, R., Torres Jr, F. 1998. Fishing down marine food web. Science 279, 860-863.

Pauly, D., Christensen, V., Guénette, S., Pitcher, T.J., Sumaila, U.R., Walters, C.J., Watson, R., Zeller, D. 2002. Towards sustainability in world fisheries. Nature 418(8), 689-695.

Peebua, P., Kruatrachue, M., Pokethitiyook, P., Kosiyachinda, P. 2006. Histological effects of contaminated sediments in Mae Klong River Tributaries, Thailand, on Nile tilapia, *Oreochromis niloticus.* ScienceAsia 32, 143-150.

Perez-Ruzafa, A., Quispe-Becerra, J.I., Garcia-Charton, J.a., Marcos, C. 2004. Composition, structure and distribution of the ichthyoplankton in a Mediteranean coastal lagoon. Journal of Fish Biology 64, 202-218.

Persson, L., Diehl, S., Johansson, L., Andersson, G., Hamrin, S.F., 1992. Trophic interaction in temperate lake ecosystems: a test of food chain theory. The American Naturalist 140(1), 59-84.

Phapavasit, N. 1995. Factors maintaining biodiversity of mangrove forest in Thailand. In: Khemnark, C. (ed.), Ecology and Management of Mangrove Restoration and

Regeneration in East and Southeast Asia. Proceeding of the ECOTONE IV, 18-20 January 1995, Wang Tai Hotel, Thailand.

Piet, G.L., Piet, J.S., Guruge, W.A.H.P., Viverberg, J., Van Densen, W.L.T. 1999. Resource partitioning along three niche dimensions in a size-structured tropical fish assemblages. Canadian Journal of Fisheries and Aquatic Sciences 56, 1241-1254.

Pimm, S.L., Kitching, R.L. 1987. The determinations of food chain lengths. Oikos 50, 302-307.

Piumsomboon, A. 2000. Mae Klong River.http://data.ecology.su.se/mnode/Asia /Thailand/maeklongriver/maeklong

Piyakarnchana, T. 1989. Yield dynamics as an index of biomass shifts in the Gulf of Thailand. In: Sherman, K., Alexander, L.M. (eds.). Biomass Yields and Geography of Large Marine Ecosystems. AAAS Symposium 111, Westview Press, Inc., Boulder, CO., pp.95-142.

Plagányi, É.E., Butterworth, D.S. 2004. A critical look at the potential of Ecopath with Ecosim to assist in practical fisheries management. In: Shannon, L.J., Cochrane, K.L., Pillar, S.C. (eds.) Ecosystem approaches to fisheries in the southern Benguela. African Journal of Marine Science 26, 261-287.

Plathong, S., 1998. Status of mangrove forests in Southern Thailand. Wetlands International Thailand Porject No. 5. Hat Yai, Thailand.

Plathong, S., Plathong J. 2004. Past and Present Threats on Mangrove Ecosystem in Peninsular Thailand. Coastal Biodiversity in Mangrove Ecosystems: UNU-INWEH-UNESCO International Training Course. Centre of Advanced Studies, Annamalai University.

Plathong, S., Sitthirach, N. 1998. Traditional and current uses of mangrove forests in Southern Thailand. Wetlands International Thailand Porject No. 3. Hat Yai, Thailand.

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

Polovina, J.J., and Own, M.D., 1983. ECOPATH: A user's manual and program listing. National Marine Fisheries Services, NOAA, Honolulu Adm. Rep. H-83-23.

Pomeroy, R. 2003. Marine protected area: an ecosystem-based fisheries management tool. Wrack Lines 3(1), 6-14.

Pomeroy, R. 2005. Ecosystem-based fisheries management. Sea Grant Conecticut. Fisheries Fact Sheet.Publication Number CTSG-05-01.

Poovachiranon, S. and Satapoomin, U. 1994. Occurrence of fish fauna associated in mangrove-seagrass habitats during the wet season, Phuket, Thailand. In: S. Sudara, S., Wilkinson, C.R., Chou, L.M. (eds.), Proceeding of the third ASEAN-Australia symposium on living coastal resources, Vol. 2: Research paper. Bangkok, Thailand; Chulalongkorn University, pp.539-550.

Porteus, E. 2008. Natural coastline defense: Mangrove forest in Southeast Asia. World Resources Institute, Washington D.C.

Potaros, M. 1995. Country report on Thailand. In: FAO/NACA 1995, Regional Study and Workshop on the Environmental Assessment and Management of Aquaculture Development (TCP/RAS/2253) Annex II-16. N.A.C.A. Environment and Aquaculture Development Series No. 1. Network of Aquaculture Centres in Asia and the Pacific, Bangkok, Thailand, Annex II-16 pp.377-390.

Prayoonrat, P. 2004. A survey of insects in the mangrove forest at the mouth of the Bangpakong River in Thailand. Asian Journal of Biology Education 2, 81-85.

Premcharoen, S., Janekitkarn, S., Vitheesawat, P. 1997. A preliminary study of the species composition of fish in mangrove forests at Tha Chin Estuary, Changwat Samut Sakhon. 23rd Congress on Science and Technology of Thailand. 20-22 October 1997. Chiang Mai, Thailand.

Raffaelli, D. 1992. Conservation of Scottish estuaries. Proceeding of the Royal Society of Edinburgh 100B, 55-76.

Raffaelli, D., Hawkins, S. 1996. Intertidal ecology. Chapman & Hall.London 356 p.

Ramírez, A.R., Lopez, M.I., Szelistowski, W.A. 1990. Composition and abundance of ichthyoplankton in a Gulf of Nicoya mangrove estuary. Revista de Biologia Tropical 38, 463-466.

Ramsar Convention Bureau 2001.Thailand designates five new Ramsar sites. http://www.ramsar.org/wn/w.n.thailand_5sites.htm

Randall, J.E. 1967. Food habit of reef fishes of the West Indies. Studies in Tropical Oceanography 5, 665-847.

Ray, S., Ulanowicz, R.E., Majee, N.C., Roy A.B. 2000. Network analysis of a benthic food web model of a partly reclaimed island in the Sundarban mangrove ecosystem. Indian Journal of Biological Systems 8, 263–278.

Rehage, J.S., Loftus, W.F. 2007. Seasonal fish community variation in headwater mangrove creeks in the southwestern Everglades: An examination of their role as dry-down refuges. Bulletin of Marine Science 80(3), 625-645.

Reusch, T.B.H., Hughes, A.R. 2006. The emerging role of genetic diversity for ecosystem functioning: Estuarine macrophytes as models. Perspective in Estuarine and Coastal Science 29(1), 159-164.

Rice, J.C. 2000. Evaluating fishery impacts using metrics of community structure. ICES Journal of Marine Science 57, 682-688.

Ricker, W.E. 1969. Food from the sea. In: Resources and Man. A study and recommendation by the Committee and Resources and Man, Division of Earth Sciences, National Academy of Science National Research Council. Freeman, San Francisco, pp. 87-108.

Rivera-Arriaga, E., Lara-Domínguez, A.L., Villalobos-Zapata, G.,Yáñez-Arancibia, A. 2003. Trophodynamic ecology of two critical habitats (seagrasses and mangroves) in Términos Lagoon, southern Gulf of Mexico. Mexico Fisheries Centre Research Report 11(6), 245-254.

Robertson, A.I., Blaber, S.J.M. 1992. Plankton, epibenthos and fish communities. In: Robertson, A.I., Alongi, D.M. (eds.), Tropical Mangrove Ecosystem. Washington DC: American Geophysical Union, pp.173-224.

Robertson, A.I., Duke, N.C. 1987. Mangroves as nursery sites: comparisons of the abundance and species composition of fish and crustaceans in mangroves and other nearshore habitats in tropical Australia. Marine Biology 96, 193-205.

Robertson, A.I., Duke, N.C. 1990a. Mangrove fish-communities in tropical Queensland, Australia: spatial and temporal patterns in densities, biomass and community structure. Marine Biology 104, 369-379.

Robertson, A.I., Duke, N.C. 1990b. Recruitment, growth and residence time of fishes in a tropical Australia mangrove ecosystem. Estuarine, Coastal and Shelf Science 31, 723-743.

Robinson, M.K. 1974. The physical oceanography of the Gulf of Thailand, Naga Expedition; Bathythermograph (BT) temperature observations in the Timor Sea, Naga Expedition, Cruise S11. Naga Report Volumn 3, Part1. La Jolla: University of California, Scripps Institution of Oceanography.

Robinson, L.A. Frid, C.L.J. 2003. Dynamic ecosystem models and the evaluation of ecosystem effects of fishing: can we make meaningful predictions? Aquatic Conservation: Marine and Freshwater Ecosystems 13, 5-20.

Rocha, G.R.A., Rossi-Wongtschowski, C.L.D.B., Pires-Vanin, A.M.S., Soares, L.S.H. 2007. Trophic model of São Sebastião Channel and continental shelf systems, SE Brazil. Pan-American Journal of Aquatic Sciences 2(2), 149-162.

Rönnbäck, P. 1999. The ecological basis for economic value of seafood production supported by mangrove ecosystems. Ecological Economics 29, 235-252.

Round, P.D. 2001. Waterfowl and their habitats in the Gulf of Thailand. BCST's Bulletin 18(4), 1-10.

Royal Forest Department 1997. Representative at the 10th National Mangrove Ecology Seminar, 25-27 August 1997, Hat Yai, Thailand. The National Research Councilof Thailand.

Rozas, L.P., Minello, T.J. 1998. Nekton use of salt marsh, seagrass, and nonvegetated habitats in a south Texas (USA) estuary. Bulletin of Marine Science 63(3), 481-501.

Ruenglertpanyakul, W., Attasat, S., Wamichponpan, P. 2004. Nutrient removal from shrimp farm effluent by aquatic plants. Water Science & Technology 50(6), 321-330.

Rybarczyk, H., Elkaim, B., Ochs, L., Loquet, N. 2003. Analysis of the trophic net work of a macrotidal ecosystem: The Bay of Somme (Eastern Channel). Estuarine, Coastal and Shelf Science 58, 405-421.

Ryder, R.A., Kerr, S.R., Taylor, W.W., Larkin, P.A. 1981. Community consequences of fish stock diversity. Canadian Journal of Fisheries and Aquatic Sciences 38, 1856-1866.

Sahavachrin, O., Boonkerd, T. 1976 Epiphytic flowering plants in mangrove forest. Proceedings of the First Thai National Seminar on Mangrove Ecology, Phuket, 10–15 January 1976. National Research Council of Thailand, Part 2, Vol. 1., 187–195. (in Thai, English abstract)

Sala, E., Aburto-Oropeza, O., Reza, M., Paredes, G., López-Lemus, L.G. 2004.Fishing down coastal food webs in the Gulf of California. Fisheries 29, 19-25.

Salini, J.P., Brewer, D.T., Blaber, S.J.M. 1998.Dietary studies on the predatory fishes of the Norman River Estuary, with particular reference to penaeid prawns. Estuarine, Coastal and Shelf Science 46, 837-847.

Sanchirico, J.N., Smith, M.D., Lipton, D.W. 2006. An approach to ecosystem-based fishery management. Discussion papers dp-06-40, Resource for the Future.

Sanders, M. 1995. Impacts of predator-prey relationships on harvesting strategies and management. Paper prepared for the Conference on the Sustainable Contribution of Fisheries to Food Security, Kyoto, Japan.

Sanvicente-Anoeve, L., Chiappa-Carrara, X., Ocana-Luna, A. 2002. Spatio-temporal variation of ichthyoplankton assemblaes in two lagoon systems of the Mexican Caribean. Bulletin of Marine Science 70(1), 19-32.

Sasekumar,A., Chong,V.C., Leh, M.U., D' Cruz, R. 1992. Mangroves as a habitat for fish and prawns. Hygrobiologia 247, 195-207.

Sasekumar,A., Chong,V.C., Lim,K.H., Singh, H. 1994.The fish community of Matang waters. In: Sudara, S., Wilkinson, C.R., Chou,L.M. (Eds.), Proceeding of the third ASEAN-Australia symposium on living coastal resources, Vol. 1: Status review. Bangkok, Thailand; Chulalongkorn University, pp.446-453.

SCB (South China Sea Programme) 1981. Report on the workshop on the biology and resources of penaeid shrimps in the South China Sea area. Part 2. SCB/GEN/81/30. 143 p.

Schoenly, K., Cohen, J.E. 1991. Temporal variation in food web structure: 16 empirical cases. Ecological Monographs 61(3), 267-298.

Seijo, J.C., Defeo, O. 1994. Externalidades positivas en pesquerías: la incidencia de interdependencias ecológicas. Jaina (México) 5(1), 18.

SHARP (Shastri Applied Research Project) 2004. Assessing management options to achieve sustainability in the shrimp-mangrove system in the Indian coastal zone of the Bay of Bengal. Working paper #GCP-JU-SHARP-1. A background paper on Environment Services and the Associated Economic Activities of the Srimp-

Mangrove System in the Indian Coastal Zone of the Bay of Bengal with Exphasis on the Sundarbans in West Bengal. 27 p.

Sikhantakasamit, B. 2001. Variations in copepod, cladocera and rotifer populations in mangrove swamp at Ban Klong Kone, Samut Songkhram Province. MSc thesis, Department of Marine Science, Faculty of Science, Chulalongkorn University, Thailand.

Silvestre, G., Selvanathan, S., Salleh, S., A.H.M. 1993. Preliminary trophic model of the coastal fisheries resources of Vrunei Darussalam, South China Sea. In: Christensen, V., Pauly, D. (eds), Trophic Models of Aquatic Ecosystems, ICLARM Conference Proceedings 26, International Center for Living Aquatic Resources Management, Manila, Philippines, pp.300-306.

Singh, H. R., Sasekumar, A. 1994. Distribution and abundance of marine catfish (Fam: Ariidae) in the Matang mangrove waters. In: Sudara, S., Wilkinson, C.R., Chou, L.M. (eds.), Proceeding of the third ASEAN-Australia symposium on living coastal resources, Vol. 2: Research paper. Chulalongkorn University, Bangkok, Thailand, pp.539-550.

Siripong, A. 1985. The characteristics of the tides in the Gulf of Thailand. Proceeding of the 5th National Seminar on Mangrove Ecology, 26-29 July 1985, Phuket, Thailand. National Reresearch Council fo Thailand, V.1: 1-15. (in Thai)

Sittilert, S. 1985. Species, quantities and distribution of benthic fauna in the Tachin River. MSc thesis, Kasetsart University, Thailand.

Sittilert, S., Yenbootra, S., Polysakdee, T. 1976. Survey of vertebrates (except fishes) in some mangrove areas. Proceedings of the 1st Thai National Seminar on Mangrove Ecology, Phuket, 10–15 January 1976. Part 2, Vol. 2, 317–331. (in Thai, English abstract)

Snedaker, S.C. 1989. Overview of ecology of mangroves and information needs for Florida Bay. Bulletin of Marine Science 44(1), 341-347.

Snidvong, K. 1982. Environmental problems and mangrove resource management in Thailand. Proceedings, NRCT-JSPS Rattanakosin Bicentennial Joint Seminar on Science and Mangrove Resources, 2-6 August 1982. Phuket, Thailand.

Songkhla Provincial Office 1998. The notification of Songkhla Province, July 28, 1999. (in Thai)

Sontirat, S. 1982. A study on morphological changes of fish larvae in brackish water. Report of Research Project, Kasetsart University, Bangkok.

Sosa-López, A., Mouillot, D., Chi, T.D., Ramos-Miranda, J. 2005. Ecological indicators based on fish biomass distribution along trophic levels: An application to the Terminos coastal lagoon, Mexico. Journal of Marine Science 62, 453-458.

Spach, H.L., Santos, C., Godefroid, R.S., Nardi, M., Cunha, F. 2004. A study of fish community structure in a tidal creek. Brazilian Journal of Biology 64(2), 1-16.

Sripanpaiboon, S. 1995. Species size composition and catch rate of trash fish by commercial trawler in the Gulf of Thailand. In: Chullasorn,S., Chotiyaputta, C. (eds.). Fishing Status of Thailand. Proceeding of Regional Workshop on Responsible Fishing, 24-27 June 1995, Bangkok, Thailand.

Stevenson, N.J. 1997. Disused shrimp ponds: Options for redevelopment of mangrove. Coastal Management 25(4), 423-425.

Stobutzki, I.C., Hall, S.J. 2005. Rebuilding coastal fisheries livelihoods after the Tsunami; Key lessons from past experience. NAGA, WorkdFish Center Newsletter 28(1&2), 6-12.

Stonsaovapak, W. 1982. A study of soil characteristics along toposequence in Mae Klong Basin study area. MSc thesis, Department of Agriculture, Kasetsart University, Thailand.

Straile, D. 2002. North Atlantic Oscillation syschronizes food –web interactions in central European lakes. Proceedings of the Royal Soceity of London B 269, 391-395.

Supongpan, M., Christensen, V., Walters, C., Pitcher, T. 2000. The use of ecosystem models to investigate multi-species management strategies for capture fisheries in the Gulf of Thailand. A paper presented for the workshop on the Use of Ecosystem Models to Investigate Multispecies Management Strategies for Capture Fisheries held by FAO/UBC, Canada, July 2000.

Sudara,S., Satumanatpan, S., Nateekanjanalarp, S. 1994. In: Sudara, S., Wilkinson, C.R., Chou, L.M. (eds.), Proceeding of the third ASEAN-Australia symposium on living coastal resources, Vol. 2: Research paper. Chulalongkorn University, Bangkok, Thailand, pp.551-560.

Taal, M.D. 1994. Flow and salt transport in mangrove swamps. Communication on Hydraulic and Geotechnical Engineering. Report No. 94-6. Faculty of Civil Engineering, Delft University of Technology.

Tabuchi, R. 2003. The rehabilitation of mangroves in Southeast Asia. In: Saxena , K.G., Liang, L., Kono, Y., Miyata, S.(eds.), Small-scale Livelihoods and Natural Resources Management in Marginal Areas: Case Studies in Monsoon Asia. Proceedings of the International Symposium, Tokyo, Japan, 29-30 October 2003.

Tanaka, K., Choo, P.S. 2000. Influence of nutrient outwelling from the mangrove swamp on the distribution of phytoplankton in the Matang mangrove estuary. Journal of Oceanography 56, 69-78.

Thailand Environment Foundation 1997. Development of an action plan to improve water quality in the Central River Basin, Thailand. Main Report, August 1997. Pollution Control Department, Ministry of Science, Technology and Environment, Thailand.

Thailand Environment Monitor 2000. The "Green" Environmental Agenda. The World Bank Group.

Thailand Environment Monitor 2006. Marine and Coastal Resources: Status and Trends. The World Bank Group.

Thayer, G.W., Colby, D.R., Hettler, W.F. 1987. Utilization of the red mangrove prop root habitat by fishes in South Florida. Marine Ecology Progress Series 35, 25-38.

Thollot, P., Kulbick, M., Harmelin-Vivien, M. 1999. Rėseaux trophiques et fonctionnement trophodynamique de l'ichtyofaune des mangroves de Nouvelle-Calėdonie. Acadėmie des sciences 322, 607-619.

Thomalla, F., Chadwick, M., Shaw, S., Miller, F. 2008. Cyclone Naris: What are the lessons from the 2004 Tsunami for Myanmar's recovery. Risk, Livelihoods and Vulnerability Programme-May 2008. Stockholm Environment Institute.

Tokrisana, R, Duangsawasdi, M. 1992. Thailand experience in fisheries management. FAO Fisheries Report 2(474), 532-537.

Tomlinson, P.B. 1986. The botany of mangroves. Cambridge University Press. Cambridge. 419 p.

To-on, J. 1999. Benthic macrofauna and distribution of fiddler crabs in mangrove forest, Tha Chin Estusry, Samut Sakhon Province. MSc thesis, Department of Marine Science, Faculty of Science, Chulalongkorn University, Thailand.

Ulanowicz, R.E. 1980. A hypothesis on the development of natural communities. Journal of Theoretical Biology 85, 223-245.

Ulanowicz, R.E. 1986. Growth and development: Ecosystem phenomenology. New-York, USA. Springer Verlag. 203 p.

Ulanowicz, R.E. 1995. The part whole relation in ecosystem. In: Patten, B.C., Jorgensen, S.E., Auerbach, S.I. (eds.), Complex Ecology. Prentice-Hall, New Jersey, pp.549-560.

Ulanowicz, R.E. 1997. Ecology, the ascendant perspective. Columbia University Press, New York.

Ulanowicz, R.E. 2005. Ecological network analysis: An escape from the machine, pp.201-207. In: A. Belgrano, U. Scharler, J. Dunne, R.E Ulanowicz (eds.). Aquatic Food Webs: An Ecosystem Approach. Oxford University Press. Oxford.

UNDP/UNESCO 1991. The integrated multidisciplinary survey and research programme of the Ranong mangrove ecosystem. UNDP/UNESCO Regional Project-Research and its application to the management of mangroves of Asia and the Pacific (RAS/86/120). Funny Publishing Limited Partnership. Bangkok. 183p.

UNEP 1985. Management and conservation of renewable marine resources. UNEP Regional Sea Reports and Studies No. 65. 85 p.

UNEP 1995. Global Biodiversity Assessment. Cambridge University Press, Cambridge.

UNEP 2004. Overfishing, a major threat to the global marine ecology. Environment Alert Bulletin. DEWA/GRID-Europe.

United Nations Thailand 2008. Tsunami in Thailand.

(http://www.un.or.th/tsunamiinthailand/Tsunami.html)

Upadhyay, V.P., Ranjan, R., Singh, J.S. 2002. Human-mangrove conflicts: The way out. Current Science 83(11), 1328-1336.

Vadhanakul, S., Meemeskul, Y., Pramokchutima, S. 1985. An analysis of demersal fish taken from otter trawl board trawling survey in the Gulf of Thailand, 1981. In: Chullasorn, S., Chotiyaputta, C. (eds.). Fishing Status of Thailand. Proceeding of Regional Workshop on Responsible Fishing, 24-27 June Bangkok, Thailand.

Vaivanijkul, P. 1976. A general survey of insects at Bangpoo. Proceedings of the First Thai National Seminar on Mangrove Ecology, Phuket, 10–15 January 1976. National Research Council of Thailand, Part 2, Vol. 1, 257–262.

Valiela, I., Bowen, J.L., York, J.K. 2001. Mangrove forests: One of the world's threatened major tropical environment. BioScience 51(10), 807-815.

Vasconcellos, M., Mackinson, S., Sloman, K., Pauly, D. 1997. The stability of trophic mass-balance models of marine ecosystems: a comparative analysis. Ecological Modelling 100, 125-134.

Vatanachai, S. 1979. Species and abundance of fish eggs and fish larvae of the mangrove swamp at Laem Pak Bia, Petchburi, 1978-1979. Report of the 3rd National Seminar on Mangrove Ecology. pp.442-460. NRCT, Bangkok, Thailand. (in Thai)

Veeravaitaya, N. 2007. Don Hoi Lot tidal flats: sustainable harvesting of razor clams. Wetland Ecosystem # 4: Tidal flats.

Vega-Cendejas, M.E. 2003. Trophic dynamics of a mangrove ecosystem in Celestun Lagoon, Yucatan Peninsular, Mexico. Fisheries Centre Research Report 11(6), 237-243.

Vega-Cendejas, M.E., Arreguín-Sáchez, F. 2001. Energy fluxes in a mangrove ecosystem from a coastal lagoon in Yucatan Peninsula. Ecological Modelling 137, 119–133.

Velasco, G., Castello, J.P. 2005. An ecotrophic model of southern Brasil continental shelf and fisheries scenarios for *Engraulis anchoita* (Pisces, Engraulididae). Atlântica, Rio Grande 27(1), 59-68.

Vibulsresth, S., Ketruangrots, C., Sriplung, N. 1975 Distribution of mangrove forest as revealed by the Earth Resources Technology Satellite (ERTS-1) Imagery. Technical Report No. 751003 National Research Council and Applied Scientific Research Corporation of Thailand, October 1975. 75 p.

Vibunpant, S., Khongchai, N., Send-eid, J., Eiamsa-ard, M., Supongpan, M. 2003. Trophic model of the coastal fisheries ecosystem in the Gulf of Thailand. In: Silvestre, G., Garces, L., Stobutzki, I., Ahmed, M., Valmonte-Santos, R.A., Luna, C., Lachica-Alino, L., Munro, P., Christensen, V., Pauly, D. (eds.), Assessment, Management and Future Directions for Coastal Fisheries in Asian Countries. WorldFish Center Conference Proceedings 67, pp.365-386.

Vidal, L., Basurto, M. 2003. A preliminary trophic model of Bahía de la Ascensión, Quintana Roo, Mexico. Fisheries Centre Research Reports 11(6), 255-264.

Vidthayanon, C., Premcharoen, S. 2001. Fish Diversity of the Lower Bang Pakong River, Central Thailand. Asian Wetland Symposium, 2001.27-30 August 2001. Penang, Malaysia. (abstract)

Vidthayanon, C., Premcharoen, S. 2002. The status of estuarine fish diversity in Thailand. Journal of Marine and Freshwater Research 53, 471-478.

Villanueva, M.A., Lalèyè, P., Albaret, J.J., Laë, R., Tito de Morias, L., Moreau, J. 2006. Comparative analysis of trophic structure and interactions of two trophic lagoons. Ecological Modelling 197(3-4), 461-477.

Volterra, V. 1926. Variazioni e fluttuazioni del numero d'individui in specie animali conviventi. Mem. R. Accad. Naz. dei Lincei. Ser. VI 2, 31-113.

Walters, C., Christensen, V., Pauly, D. 1997. Structuring dynamic models of exploited ecosystems from trophic mass-balance assessments. Reviews in Fish Biology and Fisheries 7, 139-172.

Walters, C., Christensen, V., Pauly, D. 1999. Ecospace: prediction of mesoscale patterns in trophic relationships of exploited ecosystems, with emphasis on the impacts of marine protected areas. Ecosystems 2, 539-554.

Walters, C., Christensen, V., Pauly, D. 2002. Searching for optimum fishing strategies for fisheries development, recovery and sustainability. In: Pitcher, T.J., Cochrane, K. (eds.), The Use of Ecosystems Models to Investigate Multispecies Management Strategies for Capture Fisheries. Fisheries Centre Research Reports 10(2), University of British Columbia, Vancouver.

Walters, C., Christensen, V., Pauly, D., Kitchell J.F. 2000. Representing density dependent consequences of life history stragegies in aquatic ecosystems: EcoSim II. Ecosystems 3, 70-83.

Watson, R., Pauly, D. 2001.Systemaics distortions in the world fisheries catch trends. Nature 414, 534-536.

Wattayakorn, G., King, B., Wolanski, E., Suthanaruk, P. 1998. Seasonal dispersion of petroleum contaminants in the Gulf of Thailand. Continental Shelf Research 18, 641-659.

Weatherley, A.H. 1963. Notions of the niche and competition among animals, with special reference to freshwater fish. Nature, Lond 197, 14-17.

Webster, M. M., Hart, P.J.B. 2004. Substrate discrimination and preference in foraging fish. Animal Behavior 68, 1071-1077.

Weesakul, U., Lowanichchai, S. 2005. Rainfall forecast for agriculture water allocation planning in Thailand. Thammasat International Journal of Science and Technology 10(3), 18-27.

Weis, J.S., Weis, P. 2005. Use of intertidal mangrove and sea wall habitats by coral reef fishes in the Wakatobi Marine Park, Indonesia. The Raffles Bulletin of Zoology 53(1), 119-124.

Welcomme, R.L. 1985. River fisheries. FAO Fish. Tech. Pap. 262. FAO, Rome.

Well, F.E. 1983. An analysis of marine invertebrate distributions in a mangrove swamp in northwestern Australia. Bulletin of Marine Science 33(3), 736-744.

Werry, J., Lee, S.Y. 2005. Grapsid crabs mediate link between mangrove litter production and estuarine planktonic food chains. Marine Ecology Progress Series 293, 165-176.

Whitfield, A.K. 1998. Biology and ecology of fishes in Southern African estuaries. JLB Smith Institue of Ichthyology, Ichthyological Monograph No. 2 (Grahamstown, South Africa).

Whitfield, A.K. 1999. Ichthyofaunal assemblages in estuaries: a South African study. Reviews in Fish Biology and Fisheries 9(2), 151-186.

Willmann, R., 2005. Economic aspects in the relationship between capture fisheries and aquaculture. Region workshop on low value and "trash fish" in the Asia-Pacific region, Hanoi, Vietnam, 7-9 June 2005.

Wilson, J.G., Parkes, A. 1998. Network analysis of the energy flow through the Dublin Bay. Biology and Environment: Proceedings of the Royal Irish Academy 98B, 179-190.

Wilson, J. P., Sheaves, M. 2001. Short-term temporal variations in taxonomic composition and trophic structure of a tropical estuarine fish assemblage. Marine Biology 139, 787-796.

Wiriwuttikorn, T. 1996. Long-term variations of nutrients in the Upper Gulf of Thiland. MSc thesis, Department of Environmental Science, Chulalongkorn University, Thailand.

Wolanski, E., Ridd, P., 1986. Tidal mixing and trapping in mangrove swamp. Estuarine, Coastal and Shelf Science 23, 759-771.

Wolff, M., Koch, V., Isaac, V. 2000. A trophic model of the Caeté Mangrove Estuary (North Brazil) with considerations for the sustainable use of its resources. Estuarine Coastal and Shelf Science 50, 789-803.

Wooldridge, T. 1999. Estuarine zooplankton community structure and dynamics. In: Allanson, B.R., Baird, D. (eds), Estuaries of South Africa. Cambridge University Press, Cambridge, pp. 141-166.

Worrapimphong, K. 2005. Companion modelling for razor clam *Solen regularis* conservation at Don Hoi Lord, Samut Songkhram Province. PhD thesis, Department of Biology, Faculty of Science, Chulalongkorn University, Thailand.

Wright, J.M. 1986. The ecology of fish occurring in shallow water creeks of a Nigerian mangrove swamp. Journal of Fish Biology 29, 431-441.

Yáñez-Aranbicia, A., Nugent, Y.R.S. 1977. An. Centro Cienc. del Mar y Limnol. Univ. Nal. Auton. México. El papel ecológico de los peces en estuarios, y lagunas costeras 107-114. (1), 4

Yáñez-Arancibia, A., Lara-Domínguez, A.L., Rojas-Galaviz, J.L., Sánchez-Gil, P., Day, J.W., Madden, C.J. 1988. Seasonal biomass and diversity of estuarine fishes coupled with tropical habitat heterogeneity (southern Gulf of Mexico). Journal of Fish Biology 33, 191-200.

Yap, Y.N., Sasekumar, A., Chong, V.C. 1994. Sciaenid fishes of the Matang mangrove waters. In: Sudara, S., Wilkinson, C.R., Chou, L.M. (eds.), Proceeding of the third ASEAN-Australia symposium on living coastal resources, Vol. 2: Research paper. Chulalongkorn University, Bangkok, Thailand, pp.551-560.

Yoshida, T. 2006. Ecosystem models on the evolutionary time scale: a review of perspective. Paleontological Research 10(4), 375-385.

Zetina-Rejón, M.J., Arreguín-Sánchez, F., Chávez, E.A. 2001. Using an ecosystem modeling approach to assess the management of a Mexican coastal lagoon system. Ecosystem Modelling of a Coastal Lagoon 24, 88-96.

Appendix 1 Diet content (% volume) in fish stomachs in the Mae Klong Estuary during dry season

Chelon tade

Food contents	% Volume	% Proportion	Proportion
Detritus	1409.58	32.1456	0.3214
Diatom	1193.75	27.2235	0.2722
Blue green algae	122.92	2.8032	0.0280
Polychaete	1132.91	25.8361	0.2584
Fish eggs	20	0.4561	0.0046
Copepod	69.99	1.5961	0.0160
Amphipod	11.25	0.2565	0.0026
Nematode	15	0.3421	0.0034
Crustacea fragment	37.5	0.8552	0.0085
Mysid	16.67	0.3802	0.0038
Plant tissue	17.5	0.3991	0.0040
Animal tissue	20	0.4561	0.0046
Ceratium	55	1.2543	0.0125
Bacillaria	40	0.9122	0.0091
Pleurosigma	112.08	2.5562	0.0256
Coscinodiscus	41.25	0.9407	0.0094
Nitzschia	30	0.6841	0.0068
Dinophysis	33.34	0.7603	0.0076
Daphnia	6.25	0.1425	0.0014
Sum	**4384.99**	**99.99**	**0.9999**
No. of samples	**89**		

Liza subviridis

Food contents	% Volume	% Proportion	Proportion
Polychaete	62.5	1.0373	0.0104
Diatom	4000	66.3904	0.6639
Acetes fragment	75	1.2448	0.0124
Ostracod	100	1.6597	0.0166
Pleurosigma	165.41	2.7454	0.0274
Coscinodiscus	32.08	0.5324	0.0053
Nitzschia	42.49	0.7052	0.0070
Barnacle	54.99	0.9127	0.0091
Mollusc eggs	6.25	0.1037	0.0010
Nematode	5	0.0830	0.0008
Crustacea fragment	6.25	0.1037	0.0010
Detritus	1475	24.4814	0.2448
Sum	**6024.97**	**99.97**	**0.9997**
No. of samples	**92**		

Plotosus canius

Food contents	% Volume	% Proportion	Proportion
Shrimp	200	27.5862	0.2759
Bivalve	62.5	8.6207	0.0862
Fish unidentified	37.5	5.1724	0.0517
Bivalve fragment	75	10.3448	0.1034
Detritus	50	6.8965	0.0690
Crab	300	41.3793	0.4138
Sum	**725**	**100.00**	**1.0000**
No. of samples	**13**		

Arius macronotacanthus

Food contents	% Volume	% Proportion	Proportion
Bivalve	499.99	10.2250	0.1022
Fish eggs	358.75	7.3366	0.0734
Detritus	534.16	10.9238	0.1092
Bivalve larvae	189.99	3.8854	0.0388
Amphipod	592.2	12.1107	0.1211
Copepod	510.23	10.4344	0.1043
Crab	444.17	9.0835	0.0908
Shrimp	478.74	9.7904	0.0979
Ostracod	31.25	0.6391	0.0064
Stomatopod	20.5	0.4192	0.0042
Mysid	86.25	1.7638	0.0176
Fish unidentified	304.16	6.2202	0.0622
Bivalve fragment	120.83	2.4710	0.0247
Polychaete	198.75	4.0645	0.0406
Perividis	50	1.0225	0.0102
Tellina	52.55	1.0747	0.0107
Crab megalopa	21.72	0.4442	0.0044
Modiolus	15.47	0.3164	0.0032
Polynesoda	18.39	0.3761	0.0038
Balanus	5	0.1022	0.0010
Sipunculid	7.14	0.1460	0.0015
Philyra	12.5	0.2556	0.0025
Crab zoea	43.33	0.8861	0.0089
Shrimp larvae	156.66	3.2038	0.0320
Barnacle larvae	130.89	2.6767	0.0268

Euphausid	6.25	0.1278	0.0013
Sum	**4889.87**	**99.97**	**0.9997**
No. of samples	**101**		

Osteogeneiosus militaris

Food contents	**% Volume**	**% Proportion**	**Proportion**
Shrimp	25	100.00	1.0000
Sum	**25**	**100.00**	**1.0000**
No. of samples	**1**		

Ambassis gymnocephalus

Food contents	**% Volume**	**% Proportion**	**Proportion**
Detritus	550	26.7640	0.2676
Acetes	225	10.9489	0.1095
Fish eggs	187.5	9.1241	0.0912
Shrimp larvae	489	23.7956	0.2379
Copepod	208	10.1216	0.1012
Mysid	108	5.2555	0.0525
Amphipod	175	8.5158	0.0851
Polychaete	9	0.4379	0.0044
Crab megalopa	78.5	3.8199	0.0382
Crustacea fragment	25	1.2165	0.0122
Sum	**2055**	99.98	**0.9998**
No. of samples	**61**		

Eleuthronema tetradactylum

Food contents	**% Volume**	**% Proportion**	**Proportion**
Acetes	4010.83	77.6542	0.7765
Shrimp	823.33	15.9406	0.1594
Amphipod	20	0.3862	0.0039
Mysid	113.33	2.1942	0.0219
Detritus	130	2.5169	0.0252
Barnacle larvae	30	0.5808	0.0058
Polychaete	12.5	0.2420	0.0024
Tellina	25	0.4840	0.0048
Sum	**5164.99**	**99.99**	**0.9999**
No. of samples	**88**		

Stolephorus commersonii

Food contents	**% Volume**	**% Proportion**	**Proportion**
Mysid	1007.5	11.7356	0.1173
Acetes	6445	75.0731	0.7507
Shrimp	390	4.5428	0.0454
Modiolus	81.25	0.9464	0.0095
Detritus	22.5	0.2621	0.0026
Copepod	277.08	3.2275	0.0323
Amphipod	87.49	1.0191	0.0102
Monogenea	6.25	0.0728	0.0007
Crustacea fragment	6.25	0.0728	0.0007
Trematode	162.49	1.8927	0.0189
Nematode	12.5	0.1456	0.0014
Polynesoda	8.33	0.0970	0.0009
Bivalve	14.58	0.1698	0.0017
Crap megalopa	12.5	0.1456	0.0014
Salinator	6.25	0.0728	0.0007
Fish eggs	6.25	0.0728	0.0007
Squid	20	0.2330	0.0023
Shrimp larva	18.75	0.2184	0.0021
Sum	**8584.97**	**99.95**	**0.9995**
No. of samples	**133**		

Strongylura strongylura

Food contents	**% Volume**	**% Proportion**	**Proportion**
Insect	225	5.3892	0.0538
Acetes	650	15.5689	0.1557
Needlefish	175	4.1916	0.0419
Shrimp	75	1.7964	0.0180
Polychaete	125	2.9940	0.0299
Ant	25	0.5988	0.0060
Sardine	100	2.3952	0.0239
Stolephorus	100	2.3952	0.0239
Hermit crab	25	0.5988	0.0060
Shrimp larvae	25	0.5988	0.0060
Goby	100	2.3952	0.0239
Amphipod	25	0.5988	0.0060
Fish unidentified	2525	60.4790	0.6048
Sum	**4175**	**99.98**	**0.9998**
No. of samples	**68**		

Hyporhamphus (Hyporhamphus) limbatus

Food contents	% Volume	%Proportion	Proportion
Acetes	137.5	2.0014	0.0200
Shrimp	412.5	6.0044	0.0600
Fish unidentified	300	4.3668	0.0437
Ant	1007.5	14.6652	0.1466
Insect	4730	68.8501	0.6885
Polychaete	50	0.7278	0.0073
Detritus	137.5	2.0014	0.0200
Gastropod	20	0.2911	0.0029
Spider	75	1.0917	0.0109
Sum	**6870**	**99.99**	**0.9999**
No. of samples	**113**		

Rhynchorhamphus naga

Food contents	% Volume	% Proportion	Proportion
Ant	112.5	8.3333	0.0833
Insect	325	24.0741	0.2407
Shrimp	125	9.2592	0.0926
Fish eggs	62.5	4.6296	0.0463
Acetes	675	50	0.5
Amphipod	25	1.8518	0.0185
Barnacle larvae	25	1.8518	0.0185
Sum	**1,350**	**99.99**	**0.9999**
No. of samples	**28**		

Hypoatherina valenciennei

Food contents	% Volume	% Proportion	Proportion
Acetes	965	36.5530	0.3655
Detritus	590	22.3485	0.2235
Copepod	45	1.7045	0.0170
Stomatopod	150	5.6818	0.0568
Shrimp	412.5	15.625	0.1562
Fish eggs	350	13.2576	0.1326
Fish larvae	12.5	0.4735	0.0047
Barnacle	100	3.7879	0.0379
Crab larvae	15	0.5682	0.0057
Sum	**2,640**	**99.99**	**0.9999**
No. of samples	**59**		

Leiognathus decorus

Food contents	% Volume	% Proportion	Proportion
Polychaete	50	1.3889	0.0139
Ribbon worm	2900	80.5555	0.8055
Shrimp	650	18.0555	0.1805
Sum	**3600**	**99.99**	**0.9999**
No. of samples	**45**		

Secutor insidiator

Food contents	% Volume	%Proportion	Proportion
Copepod	25	100.00	1.0000
Sum	**25**	**100.00**	**1.0000**
No. of samples	**1**		

Secutor ruconius

Food contents	% Volume	%Proportion	Proportion
Copepod	50	100.00	1.0000
Sum	**50**	**100.00**	**1.0000**
No. of samples	**2**		

Aspericorvina jubata

Food contents	% Volume	% Proportion	Proportion
Acetes	1107.5	43.0097	0.4301
Polychaete	17	0.6602	0.0066
Shrimp	112.5	4.3689	0.0437
Bivalve	75	2.9126	0.0291
Crab	513	19.9223	0.1992
Fish unidentified	125	4.8544	0.0485
Crustacea	50	1.9417	0.0194
Diatom	58	2.2524	0.0225
Barnacle larvae	33	1.2815	0.0128
Bivalve	25	0.9709	0.0097
Animal tissue	425	16.5048	0.1650
Fish eggs	34	1.3204	0.0132
Sum	**2,575**	**99.98**	**0.9998**
No. of samples	**65**		

Dendrophysa russelli

Food contents	% Volume	% Proportion	Proportion
Crustacea fragment	454.16	56.7714	0.5677
Shrimp	50	6.2501	0.0625
Polychaete	8.33	1.0413	0.0104
Copepod	58.33	7.2914	0.0729
Amphipod	58.33	7.2914	0.0729
Crab	37.5	4.6876	0.0469
Crab fragment	12.5	1.5625	0.0156
Acetes	37.5	4.6876	0.0469
Nematode	12.5	1.5625	0.0156
Balanus	37.5	4.6876	0.0469
Crab zoea	8.33	1.0413	0.0104
Tellina	25	3.1251	0.0312
Sum	**799.98**	**99.99**	**0.9999**
No. of samples	**31**		

Panna microdon

Food contents	% Volume	% Proportion	Proportion
Crab unknown	33.33	18.0172	0.1802
Shrimp	5	2.7028	0.0270
Copepod	33.33	18.0172	0.1802
Leucosia	33.33	18.0172	0.1802
Acetes	80	43.2456	0.4324
Sum	**184.99**	**100.00**	**1.0000**
No. of samples	**3**		

Pisodonophis boro

Food contents	% Volume	% Proportion	Proportion
Detritus	5	93.75	0.9375
Echiurid	75	6.25	0.0625
Sum	**80**	**100**	**1.0000**
No. of samples	**2**		

Cynoglossus lingua

Food contents	% Volume	% Proportion	Proportion
Detritus	25	20	0.2
Shrimp	75	60	0.6
Fish unidentified	25	20	0.2
Sum	**125**	**100.00**	**1.0000**
No. of samples	**3**		

Himantura imbricata

Food contents	% Volume	% Proportion	Proportion
Crab	54.17	43.3325	0.4333
Penaeus	12.5	9.9992	0.1
Acetes	29.17	23.3341	0.2333
Mysid	29.17	23.3341	0.2333
Sum	**125.01**	**99.99**	**0.9999**
No. of samples	**3**		

Dasyatis fluviorum

Food contents	% Volume	% Proportion	Proportion
Crab	37.5	13.6368	0.1364
Crab larvae	8.33	3.0292	0.0303
Acetes	175	63.6368	0.6364
Mysid	37.5	13.6368	0.1364
Copepod	8.33	3.0292	0.0303
Amphipod	8.33	3.0292	0.0303
Sum	**274.99**	**100.00**	**1.000**
No. of samples	**5**		

Escualosa thoracata

Food contents	% Volume	% Proportion	Proportion
Shrimp	1075	95.5555	0.9555
Copepod	50	4.4444	0.0444
Sum	**1125**	**99.99**	**0.9999**
No. of samples	**25**		

Escualosa elongata

Food contents	% Volume	% Proportion	Proportion
Bivalve	12.5	50	0.5
Copepod	12.5	50	0.5
Sum	**25**	**100.00**	**1.0000**
No. of samples	**1**		

Sillago sihama

Food contents	% Volume	% Proportion	Proportion
Shrimp	265	11.6228	0.1162
Polychaete	35	1.5351	0.0153
Acetes	1930	84.6491	0.8465
Mysid	12.5	0.5482	0.0055
Squid fragment	25	1.0965	0.0110
Amphipod	12.5	0.5482	0.0055
Sum	**2280**	**100.00**	**1.0000**
No. of samples	**34**		

Butis butis

Food contents	% Volume	%Proportion	Proportion
Shrimp	187.5	24.1935	0.2419
Detritus	262.5	33.8710	0.3387
Acetes	25	3.2258	0.0322
Goby	150	19.3548	0.1935
Mysid	37.5	4.8387	0.0484
Penaeus	112.5	14.5161	0.1452
Sum	**775**	**99.99**	**0.9999**
No. of samples	**21**		

Opiocara porocephala

Food contents	% Volume	% Proportion	Proportion
Isopod	50	14.2865	0.1429
Mysid	16.66	4.7603	0.0476
Acetes	266.66	76.1929	0.7619
Amphipod	16.66	4.7603	0.0476
Sum	**349.98**	**100.00**	**1.0000**
No. of samples	**9**		

Acentrogobius caninus

Food contents	% Volume	% Proportion	Proportion
Detritus	287.5	76.6667	0.7667
Fish eggs	12.5	3.3333	0.0333
Paracleistostoma	75	20.00	0.20
Sum	**375**	**100.00**	**1.0000**
No. of samples	**11**		

Favanigobius aliciae

Food contents	% Volume	%Proportion	Proportion
Detritus	25	100.00	1.0000
Sum	**25**	**100.00**	**1.0000**
No. of samples	**3**		

Pseudapocryptes lanceolatus

Food contents	% Volume	% Proportion	Proportion
Detritus	25	100.00	1.0000
Sum	**25**	**100.00**	**1.0000**
No. of samples	**1**		

Thryssa setirostis

Food contents	% Volume	% Proportion	Proportion
Acetes	25	100.00	1.0000
Sum	**25**	**100.00**	**1.0000**
No. of samples	**1**		

Sardinella fimbriata

Food contents	% Volume	% Proportion	Proportion
Zooplankton	25	100.00	1.0000
Sum	**25**	**100.00**	**1.0000**
No. of samples	**1**		

Sardinell albella

Food contents	% Volume	% Proportion	Proportion
Detritus	25	20.00	0.20
Green algae	25	20.00	0.20
Copepod	50	40.00	0.40
Shrimp	25	20.00	0.20
Sum	**125**	**100.00**	**1.0000**
No. of samples	**2**		

Gerres erythrourus

Food contents	% Volume	% Proportion	Proportion
Zooplankton	215	100.00	1.0000
Sum	**215**	**100.00**	**1.0000**
No. of samples	**4**		

Terapon theraps

Food contents	% Volume	% Proportion	Proportion
Fish	125	62.5	0.6250
Mysid	75	37.5	0.3750
Sum	**200**	**100.00**	1.0000
No. of samples	**4**		

Drepane punctata

Food contents	% Volume	% Proportion	Proportion
Polychaete	50	100.00	1.0000
Sum	**50**	**100.00**	**1.0000**
No. of samples	**1**		

Scatophagus argus

Food contents	% Volume	% Proportion	Proportion
Shrimp	25	100.00	1.0000
Sum	**25**	**100.00**	**1.0000**
No. of samples	**1**		

Appendix 1 (cont.) Hot season

Chelon tade

Food contents	% Volume	% Proportion	Proportion
Actinocyclus	7.5	0.0765	0.0007
Asterionella	2.5	0.0255	0.0002
Bacillaria	65.25	0.6658	0.0066
Cocconeis	2	0.0204	0.0002
Coscinodiscus	1699.035	17.3380	0.1733
Cymbella	8.75	0.0893	0.0009
Diploneis	9	0.0918	0.0009
Ditylum	2.5	0.0255	0.0002
Epithemia	37.5	0.3827	0.0038
Eucampid	1	0.0102	0.0001
Fragilaria	2	0.0204	0.0002
Gyrosigma	83.25	0.8495	0.0084
Lauderia	174.75	1.7832	0.0178
Mastogloia	3	0.0306	0.0003
Navicula	368.7	3.7624	0.0376
Nitzschia	622.75	6.3549	0.0635
Odontella	4.25	0.0434	0.0004
Pinnularia	25	0.2551	0.0025
Pleurosigma	1344.985	13.7251	0.1372
Rhizosolenia	15	0.1531	0.0015
Rhopaladia	25	0.2551	0.0025
Surirella	63.5	0.6480	0.0065
Thalassionema	10.25	0.1046	0.0010
Thalassiothrix	60.4	0.6163	0.0061
Other diatom	75	0.7653	0.0076
Foraminifera	631.76	6.4469	0.0645
Blue green algae	17.25	0.1760	0.0017
Crustacea nauplii	79.67	0.8130	0.0081
Bivalve larvae	4	0.0408	0.0004
Cirripedia larvae	1	0.0102	0.0001
Crab larvae	1	0.0102	0.0001
Polychaete	1502.25	15.3299	0.1533
Nematode	0.25	0.0025	0.00002
Amphipod	7.27	0.0742	0.0007
Copepod	25.53	0.2605	0.0026
Ostracod	13.75	0.1403	0.0014
Worm	1.75	0.0178	0.0001
Fish eggs	47	0.4796	0.0047
Detritus	1218.75	12.4369	0.1244
Animal tissue	744.85	7.6009	0.0760
Plant tissue	715.38	7.3002	0.0730
Bivalve fragment	0.9	0.0092	0.00009
Acetes tissue	74.25	0.7577	0.0076
Sum	**9,799.48**	**100.0000**	**1.0000**
No. of samples	**177**		

Liza subviridis

Food contents	% Volume	% Proportion	Proportion
Asteronellopsis	7.5	2.1132	0.0211
Cyclotella	4.16	1.1721	0.0117
Epithemia	7.5	2.1132	0.0211
Gyrosigma	12.5	3.5220	0.0352
Navicula	30	8.4528	0.0845
Pleurosigma	101.66	28.6439	0.2864
Thalassiosira	4.16	1.1721	0.0117
Dinophysis	4.16	1.1721	0.0117
Detritus	54.16	15.2602	0.1526
Crustacea unidentified	5	1.4088	0.0140
Insect	25	7.0440	0.0704
Plant tissue	68.7	19.3570	0.1935
Animal tissue	25	7.0440	0.0704
Nizschia	4.16	1.1721	0.0117
Copepod	1.25	0.3522	0.0035
Sum	**354.91**	**99.95**	**0.9995**
No. of samples	**10**		

Moolgooda seheli

Food contents	% Volume	% Proportion	Proportion
Coscinodiscus	19	25	0.25
Pleurosigma	19	25	0.25
Navicula	19	25	0.25
Nitzschia	19	25	0.25
Sum	**76**	**100.00**	**1.0000**
No.of samples	**1**		

Arius nenga

Food contents	% Volume	% Proportion	Proportion
Acetes	37.5	3.0992	0.0310
Crab	259	21.4049	0.2140
Mullet	75	6.1983	0.0619
Shrimp	37.5	3.0992	0.0310
Mysid	110	9.0909	0.0909
Nematode	15	1.2397	0.0124
Crustacea fragment	125	10.3306	0.1033
Shrimp fragment	147.5	12.1901	0.1219
Fish eggs	2.5	0.2066	0.0020
Crab fragment	65	5.3719	0.0537
Ambassis gymnocephalus	100	8.2645	0.0826
Acetes fragment	25	2.0661	0.0206

Leiognathus decorus	33	2.7273	0.0273
Escualosa	33	2.7273	0.0273
Polychaete	50	4.1322	0.0413
Fish unidentified	50	4.1322	0.0413
Fish fragment	45	3.7190	0.0371
Sum	**1210**	**99.96**	**0.9996**
No. of samples	**31**		

Ambassis gymnocephalus

Food contents	**% Volume**	**% Proportion**	**Proportion**
Acetes	312.5	5.7897	0.0578
Acetes tissue	117.5	2.1769	0.0218
Animal tissue	657.5	12.1816	0.1218
Crustacea tissue	1036.25	19.1987	0.1920
Crab megalopa	241.25	4.4697	0.0447
Copepod	1433	26.5493	0.2655
Amphipod	33	0.6114	0.0061
Crab zoea	433.75	8.0361	0.0804
Crustacea unidentified	97.5	1.8064	0.0181
Shimp larvae	37.5	0.6948	0.0069
Crab larvae	55	1.0190	0.0102
Shrimp fragment	291.5	5.4006	0.0540
Fish larvae	102.5	1.8990	0.0190
Mysid	12.5	0.2316	0.0023
Shrimp tissue	275	5.0949	0.0509
Fish eggs	1.25	0.0231	0.0002
Fish tissue	137.5	2.5475	0.0255
Stolephorus commersonii	50	0.9263	0.0093
Nematode	22.5	0.4168	0.0042
Polychaete	50	0.9263	0.0093
Sum	**5397.5**	**100.00**	**1.0000**
No. of samples	**244**		

Eleuthronema tetradactylum

Food contents	**% Volume**	**% Proportion**	**Proportion**
Mysid	3923.75	24.3371	0.2434
Acetes	4095.75	25.4039	0.2540
Copepod	52.5	0.3256	0.0032
Amphipod	17.5	0.1085	0.0011
Fish unidentified	30	0.1861	0.0019
Penaeus	905	5.6133	0.0561
Acetes	460	2.8531	0.0285
Crab megalopa	219.75	1.3630	0.0136
Shrimp	2650	16.4366	0.1644
Crustacea fragment	187.5	1.1629	0.0116
Crab larvae	796.25	4.7713	0.0477
Polychaete	67.5	0.4187	0.0042
Fish fragment	62.5	0.3876	0.0039
Crab zoea	1.5	0.0093	0.00009
Crustacea tissue	500	3.1012	0.0310
Animal tissue	70	0.4342	0.0043
Acetes tissue	725	4.4968	0.0450
Aspericorvina jubata	312.5	1.9383	0.0194
Polychaete fragment	12.5	0.0775	0.0008
Fish larvae	548.5	3.4021	0.0340
Shrimp larvae	484.5	3.0051	0.0300
Sum	**16122.5**	**99.82**	**0.9982**
No. of samples	**377**		

Stolephorus commersonii

Food contents	**% Volume**	**% Proportion**	**Proportion**
Mysid	1115	17.8550	0.1785
Ostracod	17.5	0.2802	0.0028
Copepod	300	4.8040	0.0480
Crab megalopa	117.5	1.8816	0.0188
Amphipod	15	0.2402	0.0024
Crab zoea	198.75	3.1827	0.0318
Acetes	1433.5	22.9553	0.2295
Fish larvae	100	1.6013	0.0160
Shrimp larva	95	1.5213	0.0152
Trematode	28.75	0.4604	0.0046
Crustacea fragment	425	6.8060	0.0680
Shrimp	310	4.9642	0.0496
Acetes tissue	975	15.6131	0.1561
Isopod	15	0.2402	0.0024
Animal tissue	260	4.1635	0.0416
Tellina	2.5	0.0400	0.0004
Modiolus	25	0.4003	0.0040
Lucifer	643.75	10.3086	0.1031
Fish unidentified	12.5	0.2002	0.0020
Stolephorus commersonii	60	0.9608	0.0096
Crab larvae tissue	35	0.5605	0.0056
Fish tissue	50	0.8007	0.0080
Bivalve	7.5	0.1201	0.0012
Gastropod	2.5	0.0400	0.0004
Sum	**6244.75**	**99.96**	**0.9996**
No. of samples	**184**		

Strongylura strongylura

Food contents	**% Volume**	**% Proportion**	**Proportion**
Stronglephorus commersonii	100	22.2222	0.2222
Acetes	150	33.3333	0.3333

Fish tissue	100	22.2222	0.2222
Animal tissue	75	16.6666	0.1666
Fish fragment	25	5.5555	0.0555
Sum	**450**	**99.98**	**0.9998**
No. of samples	**20**		

Hyporhamphus (Hyporhamphus) limbatus

Food contents	% Volume	% Proportion	Proportion
Detritus	50	1.7693	0.0177
Ant	97.5	3.4501	0.0345
Polychaete	52.5	1.8577	0.0186
Insect	1290	45.6475	0.4565
Crustacea fragment	625	22.1161	0.2212
Animal tissue	433.75	15.3485	0.1535
Ant tissue	200	7.0771	0.0708
Crab	10	0,3538	0.0035
Crab zoea	5	0.1769	0.0018
Crautacea unidentified	60	2.1231	0.0212
Fish larvae	1	0.0003	0.000003
Trematode	1.25	0.0442	0.0004
Sum	**2826**	**99.97**	**0.9997**
No. of samples	**90**		

Rhynchorhamphus naga

Food contents	% Volume	% Proportion	Proportion
Polychaete	50	14.2857	0.1428
Crab magalopa	50	14.2857	0.1428
Ant	75	21.4286	0.2143
Ant tissue	50	14.2857	0.1428
Animal tissue	75	21.4286	0.2143
Insect	50	14.2857	0.1428
Sum	**350**	**99.98**	**0.9998**
No. of samples	**12**		

Hypoatherina valenciennei

Food contents	% Volume	% Proportion	Proportion
Detritus	27	4.4850	0.0448
Animal tissue	575	95.5149	0.9551
Sum	**602**	**99.99**	**0.9999**
No. of samples	**40**		

Leiognathus decorus

Food contents	% Volume	% Proportion	Proportion
Animal tissue	695	26.4300	0.2643
Nemertean (*Pilidium* larvae)	245	8.3051	0.0830
Polychaete	25	0.8474	0.0085
Copepod	447.5	15.1695	0.1517
Amphipod	201.25	6.8220	0.0682
Mysid	1036.25	35.1271	0.3513
Shrimp	10	0.3390	0.0034
Panaeus	10	0.3390	0.0034
Acetes	115	3.8983	0.0390
Crab larvae	5	0.1695	0.0017
Crustacea unidentified	25	0.8474	0.0085
Nematode	110	0.8474	0.0085
Acetes tissue	25	0.8474	0.0085
Sum	**2,950**	**100.00**	**1.0000**
No. of samples	**81**		

Secutor incidiator

Food contents	% Volume	% Proportion	Proportion
Animal tissue	75	100.0000	1.0000
Sum	**75**	100.0000	1.0000
No. of samples	**12**	**100.00**	**1.0000**

Secutor ruconius

Food contents	% Volume	% Proportion	Proportion
Animal tissue	25	100.00	1.0000
Sum	**25**	100.00	1.0000
No. of samples	**15**	**100.00**	**1.0000**

Aspericorvina jubata

Food contents	% Volume	% Proportion	Proportion
Crustacea fragment	95	11.5151	0.1151
Crab megalopa	12.5	1.5151	0.0151
Trematode	15	1.8182	0.0182
Fish fragment	2.5	0.3030	0.0030
Detritus	2.5	0.3030	0.0030
Animal tissue	47.5	5.7576	0.0576
Acetes	125	15.1515	0.1515
Copepod	460	55.7576	0.5576
Polychaete tissue	25	3.0303	0.0303
Mysid	35	4.2424	0.0424

Nematode	5	0.6061	0.0060
Sum	**825**	**99.98**	**0.9998**
No. of samples	**26**		

Dendrophysa russelli

Food contents	% Volume	% Proportion	Proportion
Acetes	301.25	25.7369	0.2574
Crustacea fragment	127.5	10.8928	0.1089
Mysid	60.5	5.1687	0.0517
Copepod	327.5	27.9795	0.2798
Acetes tissue	250	21.3584	0.2136
Copepod tissue	25	2.1358	0.0213
Animal tissue	75	6.4075	0.0641
Trematode	3.75	0.3204	0.0032
Sum	**1170.5**	**100.00**	**1.0000**
No. of samples	**43**		

Plotosus canius

Food contents	% Volume	% Proportion	Proportion
Crab	25	11.1111	0.1111
Macrophthalmus	52.5	23.3333	0.2333
Pelecyora glouldi	62.5	27.7777	0.2777
Crab fragment	25	11.1111	0.1111
Shrimp fragment	25	11.1111	0.1111
Philyra	35	15.5555	0.1555
Sum	**225**	**99.98**	**0.9998**
No. of samples	**5**		

Himantura imbricata

Food contents	% Volume	% Proportion	Proportion
Amphipod	65	43.3333	0.4333
Acetes	17.5	11.6667	0.1167
Crab unidentified	67.5	45.0000	0.4500
Sum	**150**	**100.00**	**1.0000**
No. of samples	**3**		

Escualosa thoracata

Food contents	% Volume	% Proportion	Proportion
Copepod	450	53.8922	0.5389
Crustacea unidentified	100	11.9760	0.1198
Crab zoea	210.75	25.2395	0.2524
Crustacea fragment	22.5	2.6946	0.0269
Amphipod	2.5	0.2994	0.0030
Animal tissue	49.25	5.8982	0.0590
Sum	**835**	**100.00**	**1.0000**
No. of samples	**33**		

Escualosa elongata

Food contents	% Volume	% Proportion	Proportion
Crustacea tissue	175	77.7777	0.7777
Animal tissue	50	22.2222	0.2222
Sum	**225**	**99.99**	**0.9999**
No. of samples	**10**		

Anotostoma chacunda

Food contents	% Volume	% Proportion	Proportion
Foraminifera	15	30.0000	0.3000
Gyrosigma	10	20.0000	0.2000
Navicula	10	20.0000	0.2000
Pleurosigma	15	30.0000	0.3000
Sum	**50**	**100.00**	**1.0000**
No. of samples	**1**		

Herklotsichthys lispilonotus

Food contents	% Volume	% Proportion	Proportion
Detritus	25	10.8696	0.1087
Trematode	7.5	3.2609	0.0326
Animal tissue	197.5	85.8696	0.8587
Sum	**230**	**100.00**	**1.0000**
No. of samples	**14**		

Sardinella gibbosa

Food contents	% Volume	% Proportion	Proportion
Acetes	50	66.6666	0.6666
Animal tissue	25	33.3333	0.3333
Sum	**75**	**99.99**	**0.9999**
No. of samples	**2**		

Sardinella albella

Food contents	% Volume	% Proportion	Proportion
Zooplankton tissue	25	7.1428	0.0714
Animal tissue	75	21.3286	0.2143
Acetes tissue	250	71.4286	0.7123
Sum	**350**	**100.00**	**1.0000**
No. of samples	**9**		

Scatophagus argus

Food contents	% Volume	% Proportion	Proportion
Amphipod	75	18.7500	0.1875
Animal tissue	225	56.25	0.5625
Polychaete	25	6.25	0.0625
Algae	75	18.7500	0.1875
Sum	**400**	**100.00**	**1.0000**
No. of samples	**10**		

Scomberoides commersonnianus

Food contents	% Volume	% Proportion	Proportion
Fish larvae	45	60.0000	0.6000
Acetes	30	40.0000	0.4000
Sum	**75**	**100.00**	**1.0000**
No. of samples	**1**		

Terapon theraps

Food contents	% Volume	% Proportion	Proportion
Fish fragment	25	100.00	1.0000
Sum	**25**	**100.00**	**1.0000**
No. of samples	**1**		

Alepes djeddaba

Food contents	% Volume	% Proportion	Proportion
Crustacea tissue	175	63.6364	0.6363
Isopod	25	9.0909	0.0909
Copepod	75	27.2727	0.2727
Sum	**275**	**99.99**	**0.9999**
No. of samples	**9**		

Scomberoides tol

Food contents	% Volume	% Proportion	Proportion
Acetes	100	100.0000	1.0000
Sum	**100**	**100**	1.0000
No. of samples	**1**		

Appendix 1 (cont.) Rainy season

Chelon tade

Food contents	% Volume	% Proportion	Proportion
Amphipleura	11.11	0.0410	0.0004
Amphora	140.83	0.5199	0.0052
Anabaena	51.11	0.1887	0.0019
Anomocnema	2.5	0.0092	0.00009
Cocconeis	47.5	0.1754	0.0017
Coscinodiscus	1204.79	4.4480	0.0445
Cyclotella	428.47	1.5819	0.0158
Diploneis	41.83	0.1544	0.0015
Eutonia	156.79	0.5789	0.0058
Grammetophora	67.5	0.2492	0.0025
Urotrix	22.67	0.0837	0.0008
Skeletonema	16.67	0.0615	0.0006
Gyrosigma	742.45	2.7411	0.0274
Scenedesmus	33.33	0.1230	0.0012
Oscillatoria	36	0.1329	0.0013
Navicula	1337.03	4.9362	0.0494
Nitzschia	3716.84	13.7224	0.1372
Phagus	60.68	0.2240	0.0022
Pinnularia	528.66	1.9518	0.0195
Pleurosigma	2948.63	10.8862	0.1089
Pseudonitzschia	188.34	0.6953	0.0069
Rhizosolenia	2135.96	7.8858	0.0788
Surirella	41	0.1514	0.0015
Thalassionema	1452.96	5.3642	0.0536
Green algae	338.91	1.2512	0.0125
Polychaete	1462.33	5.3988	0.0540
Amphipod	12.5	0.0461	0.0005
Copepod	168.67	0.6227	0.0062
Ostracod	32.67	0.1206	0.0012
Flatworm	25	0.0923	0.0009
eggs	12.5	0.0461	0.0005
Detritus	8916.05	32.9176	0.3292
Plant tissue	595	2.1967	0.0220
Crustacea tissue	16	0.0591	0.0006
Gravel	92.7	0.3422	0.0034
Sum	**27085.98**	**99.96**	**0.9996**
No. of samples	**272**		

Liza subviridis

Food contents	% Volume	% Proportion	Proportion
Amphora	2.86	0.1682	0.0017
Pinnularia	16.67	0.9806	0.0098
Rhizosolenia	41.67	2.4511	0.0245
Grammatophora	2.86	0.1682	0.0017
Navicula	19.53	1.1488	0.0115
Pleurosigma	149.53	8.7957	0.0879
Nizschia	372.86	21.9324	0.2193
Thalassionema	26.67	1.5688	0.0157
Detritus	492.86	28.9911	0.2899
Polychaete	476.67	28.0387	0.2804
Crustacea tissue	2.86	0.1682	0.0017
Plant tissue	45	2.6470	0.0265
Copepod	50	2.9411	0.0294
Sum	**1700.04**	**100.00**	**1.0000**
No. of samples	**17**		

Arius macronotacanthus

Food contents	% Volume	% Proportion	Proportion
Acetes	682	1.3018	0.0130
Nereis	26138.3	49.8915	0.4989
Copepod	492.5	0.9401	0.0094
Amphipod	1155.5	2.2056	0.0220
Ostracod	120	0.2290	0.0023
Polychaete	2573.33	4.9118	0.0491
Xanthid crab	50	0.0954	0.0009
Hermit crab	510	0.9735	0.0097
Shrimp fragment	150	0.2863	0.0029
Shrimp	33.33	0.0636	0.0006
Isopod	160	0.3054	0.0030
Tellina .	6354.99	12.1301	0.1213
Penaeus	650	1.2407	0.0124
Crab	3310	6.3180	0.0632
Worm	12	0.0229	0.0002
Nematode	5	0.0095	0.0001
Crustacea fragment	2377.5	4.5380	0.0454
Bivalve	1422.49	2.7152	0.0271
Fish tissue	1110	2.1187	0.0212
Gastropod	183.33	0.3562	0.0036

Alpheur sp	90	0.1718	0.0017
Modiolus	641.66	1.2248	0.0122
Macrophthalamus	30	0.0573	0.0006
Morella	133.33	0.2545	0.0025
Polycyora	300	0.5726	0.0057
Sternaspis	100	0.1909	0.0019
Fish	380	0.7253	0.0072
Ant	30	0.0573	0.0006
Bryozoa	20	0.0382	0.0004
Algae	50	0.0954	0.0009
Gravel	20	0.0382	0.0004
Animal tissue	3105	5.9267	0.0593
Sum	**52390.26**	**99.25**	**0.9925**
No. of samples	**533**		

Arius sagor

Food contents	% Volume	% Proportion	Proportion
Crab	500	14.3472	0.1435
Neries	60	1.7217	0.0172
Amphipod	64	1.8364	0.0184
Copepod	11	0.3156	0.0031
Crustacea fragment	590	16.9297	0.1693
Tellina	70	2.0086	0.0201
Bivalve	195	5.5954	0.0560
Gastropod	15	0.4304	0.0043
Alpheus	90	2.5825	0.0258
Fish scale	100	2.8694	0.0287
Animal tissue	1790	51.3630	0.5136
Sum	**3485**	**100.00**	**1.0000**
No. of samples	**35**		

Cryptarius truncatus

Food contents	% Volume	% Proportion	Proportion
Nereis	40	40	0.40
Fish scale	60	60	0.60
Sum	**100**	**100.00**	**1.0000**
No. of samples	**1**		

Ketengus typus

Food contents	% Volume	% Proportion	Proportion
Animal tissue	100	4.1511	0.0415
Polychaete	350	14.5289	0.1550
Fish scale	1549	64.3008	0.6430
Copepod	90	3.7360	0.0370
Amphipod	10	0.4151	0.0041
Sternaspis	33.33	1.3836	0.0136
Nereis	243.33	10.1009	0.1010
Spionidae (Polychaete)	33.33	1.3836	0.0138
Sum	**2408.99**	**100.00**	**1.0000**
No. of samples	**23**		

Ambassis gymnocephalus

Food contents	% Volume	% Proportion	Proportion
Acetes	2895	7.4060	0.0741
Animal tissue	3350	8.57	0.0857
Crustacea tissue	3170	8.1095	0.0811
Crab megalopa	130	0.3326	0.0033
Copepod	7141	18.2681	0.1827
Amphipod	465	1.1896	0.0119
Crab zoea	20	0.0512	0.0005
Gastropod	60	0.1535	0.0015
Bivalve	139	0.3556	0.0035
Isopod	100	0.0256	0.0002
Insect	100	0.0256	0.0002
Alpheus	200	0.5116	0.0051
Shrimp tissue	390	0.9977	0.0100
Nereis	5520	14.1212	0.1412
Polychaete tissue	15410	39.8900	0.3989
Sum	**39090**	**99.99**	**0.9999**
No. of samples	**396**		

Ambassis nalua

Food contents	% Volume	% Proportion	Proportion
Nereis	100	100.00	1.0000
Sum	**100**	**100.00**	**1.0000**
No. of samples	**1**		

Eleuthronema tetradactylum

Food contents	% Volume	% Proportion	Proportion
Mysid	530	6.2353	0.0623
Acetes	3380	39.7647	0.3976
Copepod	50	0.5882	0.0059
Penaeus	2470	29.0588	0.2905
Alpheus	160	1.8823	0.0188
Crab	100	1.1765	0.0118
Shrimp	150	1.7647	0.0176
Polycyora	40	0.4706	0.0047
Polychaete	130	1.5294	0.0153
Nereis	420	4.9412	0.0494
Crustacea tissue	570	6.7059	0.0670
Animal tissue	500	5.8823	0.0588
Sum	**8,500**	**99.97**	**0.9997**
No. of samples	**164**		

Stolephorus commersonii

Food contents	% Volume	% Proportion	Proportion
Mysid	2890	14.4888	0.1449
Ostracod	60	0.2831	0.0028
Copepod	3412.33	16.0997	0.1610
Crab megalopa	25	0.1179	0.0012
Amphipod	180	0.8492	0.0085
Crab zoea	165	0.7785	0.0078
Acetes	3403.33	16.0572	0.1606
Animal tissue	300	1.4154	0.0141
Plant tissue	100	0.0418	0.0004
Crustacea fragment	2328.33	10.9853	0.1098
Shrimp	330	1.5570	0.0156
Penaeus	830	3.9160	0.0392
Goby	70	0.3303	0.0033
Animal tissue	180	0.8492	0.0085
Alpheus	170	0.8021	0.0080
Modiolus	120	0.5662	0.0057
Nereis	2020	9.5305	0.0953
Fish unidentified	150	0.7077	0.0071
Polychaete	2330	10.9932	0.1099
Crab	265	1.2503	0.0125
Gastropod larvae	100	0.0418	0.0004
Bivalve	1716	8.0962	0.0810
Gastropod	50	0.2359	0.0023
Sum	**21194.99**	**99.99**	**0.9999**
No. of samples	**126**		

Strongylura strongylura

Food contents	% Volume	% Proportion	Proportion
Penaeus	400	4.3956	0.0439
Acetes	300	3.2967	0.0330
Shrimp	130	1.4286	0.0143
Nereis	900	9.8901	0.0989
Polychaete	2170	23.8461	0.2385
Ant	100	1.0989	0.0110
Animal tissue	4050	44.5055	0.4450
Fish	700	7.6923	0.0769
Insect tissue	250	2.7472	0.0275
Crustacea tissue	100	1.0989	0.0110
Sum	**9,100**	**100.00**	**1.0000**
No. of samples	**98**		

Hyporhamphus (Hyporhamphus) limbatus

Food contents	% Volume	% Proportion	Proportion
Detritus	100	0.8403	0.0084
Amphipod	100	0.8403	0.0084
Ant	235	1.9748	0.0197
Polychaete	3303	27.7563	0.2776
Insect	1152	9.6807	0.0968
Nereis	2100	17.6470	0.1765
Crustacea fragment	100	0.8403	0.0084
Animal tissue	4710	39.5798	0.3958
Modiolus	100	0.8403	0.0084
Sum	**11900**	**100.00**	**1.0000**
No. of samples	**118**		

Rhynchorhamphus naga

Food contents	% Volume	% Proportion	Proportion
Acetes	100	4	0.0400
Animal tissue	130	5.2	0.0520
Insect	2270	90.8	0.9080
Sum	**2500**	**100.00**	**1.0000**
No. of samples	**25**		

Hypoatherina valenciennei

Food contents	% Volume	% Proportion	Proportion
Insect	1200	50	0.5000
Copepod	700	29.1666	0.2916
Acetes	100	4.1667	0.0417
Ant	300	12.5	0.1250
Animal tissue	100	4.1667	0.0417
Sum	**2400**	**100.00**	**1.0000**
No. of samples	**24**		

Leiognathus decorus

Food contents	% Volume	% Proportion	Proportion
Animal tissue	4579	43.5681	0.4357
Nemertean (Pilidium larvae)	100	0.9515	0.0095
Polychaete	140	1.3321	0.0133
Copepod	2501.33	23.7996	0.2380
Amphipod	859.66	8.1795	0.0818
Ostracod	50	0.4757	0.0047
Nereis	525	4.9952	0.0499
Bivalve	635	6.0419	0.0604
Tellina	380	3.6156	0.0361
Penaeus	100	0.9515	0.0095
Crustacea tissue	473.33	4.5036	0.0450
Trematode	33.33	0.3171	0.0032
Insect	33.33	0.3171	0.0032
Detritus	100	0.9515	0.0095
Sum	**10509.98**	**100.00**	**1.0000**
No. of samples	**109**		

Aspericorvina jubata

Food contents	% Volume	% Proportion	Proportion
Crustacea fragment	1390	10.2963	0.1030
Crab	400	2.9630	0.0296
Algae	20	0.1481	0.0015
Shrimp	210	1.5555	0.0155
Animal tissue	5090	37.7037	0.3770
Acetes	1265	9.3704	0.0937
Amphipod	435	3.2222	0.0322
Copepod	1360	10.0741	0.1007
Philyra	370	2.7408	0.0274
Leucosid	90	0.6667	0.0067
Tellina	100	0.7407	0.0074
Polychaete tissue	1180	8.7407	0.0874
Mysid	200	1.4815	0.0148
Penaeus	100	0.7407	0.0074
Modiolus	40	0.2963	0.0030
Bivalve	130	0.9630	0.0096
Gastropod	850	6.2963	0.0620
Fish unidentified	200	1.4815	0.0148
Plant tissue	70	0.5185	0.0052
Sum	**13500**	**99.89**	**0.9989**
No. of samples	**136**		

Dendrophysa russelli

Food contents	% Volume	% Proportion	Proportion
Acetes	210	8.7137	0.0871
Crustacea fragment	300	12.4481	0.1245
Mysid	130	5.3942	0.0539
Copepod	100	4.1494	0.0415
Eunicidae	100	4.1494	0.0415
Amphipod	560	23.2365	0.2324
Nereis	200	8.2987	0.0830
Bivalve	70	2.9046	0.0290
Animal tissue	220	9.1286	0.0913
Penaeus	100	4.1494	0.0415
Crab	290	12.0332	0.1203
Fish unidentified	130	5.3942	0.0539
Sum	**2410**	**99.99**	**0.9999**
No. of samples	**24**		

Chrysochir aureus

Food contents	% Volume	% Proportion	Proportion
Polychaete	300	100	1.0000
Sum	**300**	**100.00**	**1.0000**
No. of samples	**3**		

Panna microdon

Food contents	% Volume	% Proportion	Proportion
Acetes	200	20	0.2000
Alpheus	100	10	0.1000
Shrimp	100	10	0.1000
Modiolus	40	4	0.0400
Polychaete	80	8	0.0800
Penaeus	100	10	0.1000
Bivalve	80	8	0.0800
Animal tissue	160	16	0.1600
Algae	140	14	0.1400
Sum	**1000**	**100.00**	**1.0000**
No. of samples	**10**		

Plotosus canius

Food contents	% Volume	% Proportion	Proportion
Crab	705	1.2877	0.0129
Copepod	100	0.1826	0.0018
Polychaete	120	0.2192	0.0022
Bivalve	535	0.9772	0.0098
Gastropod	150	0.2740	0.0027
Isopod	50	0.0913	0.0009
Nereis	51265	93.6347	0.9363
Modiolus	130	0.2374	0.0024
Xanthid crab	50	0.0913	0.0009
Grapsid crab	70	0.1278	0.0013
Alpleus	20	0.0365	0.0004
Shrimp	100	0.1826	0.0018
Crustacea tissue	420	0.7671	0.0077
Philyra	170	0.3105	0.0031
Amphipod	815	1.4886	0.0149
Insect	50	0.0913	0.0009
Sum	**54750**	**100.00**	**1.0000**
No. of samples	**56**		

Cynoglossus puncticeps

Food contents	% Volume	% Proportion	Proportion
Amphipod	550	78.5714	0.7857
Copepod	50	7.1428	0.0714
Acetes	100	14.2857	0.1428
Sum	**700**	**99.99**	**0.9999**
No. of samples	**7**		

Dasyatis fluviorum

Food contents	% Volume	% Proportion	Proportion
Crust. tissue	200	16.6667	0.1667
Amphipod	400	33.3333	0.3333
Gastropod larvae	100	8.3333	0.0833
Shrimp	100	8.3333	0.0833
Penaeus	80	6.6667	0.0667
Acetes	20	1.6667	0.0167
Crab	100	8.3333	0.0833
Animal tissue	200	16.6667	0.1667
Sum	**1200**	**100.00**	**1.0000**
No. of samples	**9**		

Escualosa thoracata

Food contents	% Volume	% Proportion	Proportion
Insect	100	100	1.0000
Sum	**100**	**100.00**	**1.0000**
No. of samples	**1**		

Escualosa elongata

Food contents	% Volume	% Proportion	Proportion
Mysid	100	5.4054	0.0540
Detritus	480	25.9459	0.2594
Copepod	400	21.6216	0.2162

Gastropod	335	18.1081	0.1811
Crab megalopa	15	0.8108	0.0081
Acetes tissue	250	13.5135	0.1351
Animal tissue	250	13.5135	0.1351
Plant tissue	20	1.0811	0.0108
Sum	**1,850**	**99.98**	**0.9998**
No. of samples	**16**		

Sillago sihama

Food contents	% Volume	% Proportion	Proportion
Crab	280	13.3333	0.3334
Copepod	40	1.9048	0.0190
Amphipod	105	5	0.0500
Crustacea tissue	525	25	0.2400
Shrimp	100	4.7619	0.0476
Crab megalopa	25	1.1905	0.0119
Polychaete	425	20.2381	0.2024
Nereis	330	15.7143	0.1571
Fish unidentified	70	3.3333	0.0333
Animal tissue	200	9.5238	0.0952
Sum	**2100**		
No. of samples	**21**		

Sardinella fimbriata

Food contents	% Volume	% Proportion	Proportion
Copepod	50	5.2631	0.0526
Nematode	100	10.5263	0.1053
Detritus	150	15.7895	0.1579
Gravel	150	10.5300	0.1053
Animal tissue	550	57.8947	0.5789
Sum	**950**	**100.00**	**1.0000**
No. of samples	**9**		

Sardinella lemuru

Food contents	% Volume	% Proportion	Proportion
Nereis	1190	49.5833	0.4958
Polychaete	685	28.5417	0.2854
Acetes	355	14.7917	0.1479

Crustacea tissue	90	3.75	0.0375
Amphipod	10	0.4167	0.0042
Isopod	20	0.8333	0.0083
Shrimp	50	2.0833	0.0208
Sum	**2400**	**99.99**	**0.9999**
No. of samples	**24**		

Setipinna taty

Food contents	% Volume	% Proportion	Proportion
Acetes	2440	53.0435	0.5304
Crustacea tissue	200	4.3478	0.0435
Shrimp	275	5.9783	0.0598
Copepod	1135	24.6739	0.2467
Amphipod	10	0.2174	0.0022
Nereis	470	10.2174	0.1022
Animal tissue	70	1.5217	0.0152
Sum	**4600**	**100.00**	**1.0000**
No. of samples	**46**		

Thryssa hamiltoni

Food contents	% Volume	% Proportion	Proportion
Polychaete	30	1.7647	0.0176
Shrimp	20	1.1765	0.0118
Acetes	1650	97.0588	0.9706
Sum	**1700**	**100.00**	**1.0000**
No. of samples	**16**		

Sardinella albella

Food contents	% Volume	% Proportion	Proportion
Copepod	100	100	1.0000
Sum	**100**	**100.00**	**1.0000**
No. of samples	**7**		

Gerres erythrourus

Food contents	% Volume	% Proportion	Proportion
Detritus	100	25.0006	0.2500
Nereis	95	23.7506	0.2375
Amphipod	5	1.2500	0.0125
Insect	100	25.0006	0.2500
Foraminifera	33.33	8.3327	0.0833
Copepod	33.33	8.3327	0.0833
Gastropod	33.33	8.3327	0.0833
Sum	**399.99**	**99.99**	**0.9999**
No. of samples	**4**		

Scatophagus argus

Food contents	% Volume	% Proportion	Proportion
Amphipod	50	0.8772	0.0088
Animal tissue	1470	25.7895	0.2579
Polychaete	1020	17.8947	0.1789
Nereis	790	13.8596	0.1386
Algae	1910	33.5088	0.3351
Leaves	410	7.1930	0.0719
Nematode	50	0.8772	0.0088
Sum	**5700**	**100.00**	**1.0000**
No. of samples	**57**		

Drepane punctata

Food contents	% Volume	% Proportion	Proportion
Crustacea tissue	100	50	0.5000
Polychaete	100	50	0.5000
Sum	**200**	**100.00**	**1.0000**
No. of samples	**2**		

Terapon theraps

Food contents	% Volume	% Proportion	Proportion
Insect	20	5	0.0500
Animal tissue	100	25	0.2500
Nereis	80	20	0.2000
Crab	100	25	0.2500
Fish egg	100	25	0.2500
Sum	**400**	**100.00**	**1.0000**
No. of samples	**4**		

Alepes djeddaba

Food contents	% Volume	% Proportion	Proportion
Fish unidentified	100	100	1.0000
Sum	**100**	**100.00**	**1.0000**
No. of samples	**1**		

Nibea albiflora

Food contents	% Volume	% Proportion	Proportion
Alpheus	100	100	1.0000
Sum	**100**	**100.00**	**1.0000**
No. of samples	**1**		

Opiocara porocephala

Food contents	% Volume	% Proportion	Proportion
Tellina	100	100	1.0000
Sum	**100**	**100.00**	**1.0000**
No. of samples	**1**		

Butis butis

Food contents	% Volume	% Proportion	Proportion
Isopod	100	33.3333	0.3333
Ostracod	100	33.3333	0.3333
Amphipod	100	33.3333	0.3333
Sum	**300**	**99.99**	**0.9999**
No. of samples	**3**		

Appendix 2 Compositions in diet matrix

Dry season

Rays

Ecological groups	Proportion
Sergestid shrimp	0.435
Shrimp	0.050
Crab	0.285
Zooplankton	0.230
Sum	**1.000**

Anchovies

Ecological groups	Proportion
Nekton	0.001
Shrimp	0.023
Sergestid shrimp	0.88
Benthic invertebrates	0.017
Zooplankton	0.082
Detritus	0.001
Sum	**1.000**

Sardines

Ecological groups	Proportion
Shrimp	0.289
Benthic invertebrates	0.125
Zooplankton	0.486
Phytoplankton	0.050
Detritus	0.050
Sum	**1.000**

Catfishes

Ecological groups	Proportion
Nekton	0.038
Shrimp	0.483
Crab	0.169
Benthic invertebrates	0.129
Zooplankton	0.122
Detritus	0.059
Sum	**1.000**

Mullets

Ecological groups	Proportion
Sergestid shrimp	0.007
Benthic invertebrates	0.141
Zooplankton	0.030
Phytoplankton	0.539
Detritus	0.283
Sum	**1.000**

Pelagics

Ecological groups	Proportion
Nekton	0.499
Shrimp	0.082
Sergestid shrimp	0.259
Crab	0.002
Benthic invertebrates	0.020
Zooplankton	0.077
Detritus	0.061
Sum	**1.000**

Perchlets

Ecological groups	Proportion
Shrimp	0.250
Sergestid shrimp	0.122
Benthic invertebrates	0.004
Zooplankton	0.356
Detritus	0.268
Sum	**1.000**

Ponyfishes

Ecological groups	Proportion
Shrimp	0.060
Benthic invertebrates	0.273
Zooplankton	0.667
Sum	**1.000**

Threadfin

Ecological groups	Proportion
Shrimp	0.160
Sergestid shrimp	0.776
Benthic invertebrates	0.007
Zooplankton	0.032
Detritus	0.025
Sum	**1.000**

Croakers

Ecological groups	Proportion
Nekton	0.017
Shrimp	0.295
Sergestid shrimp	0.303
Crab	0.164
Benthic invertebrates	0.084
Zooplankton	0.130
Phytoplankton	0.007
Sum	**1.000**

Benthics

Ecological groups	Proportion
Nekton	0.056
Shrimp	0.141
Sergestid shrimp	0.113
Crab	0.029
Benthic invertebrates	0.009
Zooplankton	0.046
Detritus	0.606
Sum	**1.0000**

Spot scat

Ecological groups	Proportion
Shrimp	1.000
Sum	**1.000**

Benthopelagics

Ecological groups	Proportion
Nekton	0.160
Shrimp	0.029
Sergestid shrimp	0.212
Benthic invertebrates	0.254
Zooplankton	0.346
Sum	**1.0000**

Appendix 2 (cont.)

Hot season

Rays

Ecological groups	Proportion
Sergestid shrimp	0.392
Crabs	0.392
Zooplankton	0.217
Sum	**1.000**

Anchovies

Ecological groups	Proportion
Nekton	0.031
Shrimp	0.057
Sergestid shrimp	0.461
Benthic invertebrates	0.063
Zooplankton	0.388
Sum	**1.000**

Sardines

Ecological groups	Proportion
Sergestid shrimp	0.245
Crab	0.036
Benthic invertebrates	0.127
Zooplankton	0.547
Phytoplankton	0.010
Detritus	0.035
Sum	**1.000**

Catfishes

Ecological groups	Proportion
Nekton	0.346
Shrimps	0.060
Sergestid shrimp	0.119
Crabs	0.120
Benthic inverterates	0.138
Zooplankton	0.201
Detritus	0.016
Sum	**1.000**

Mullets

Ecological groups	Proportion
Nekton	0.016
Sergestid shrimp	0.001
Benthic invertebrates	0.073
Zooplankton	0.039
Phytoplankton	0.583
Detritus	0.288
Sum	**1.000**

Pelagics

Ecological groups	Proportion
Nekton	0.458
Shrimps	0.121
Sergestid shrimp	0.167
Crab	0.002
Benthic invertebrate	0.081
Zooplankton	0.072
Detritus	0.099
Sum	**1.000**

Perchlets

Ecological groups	Proportion
Nekton	0.028
Shrimps	0.062
Sergestid shrimp	0.707
Benthic invertebrates	0.013
Zooplankton	0.190
Sum	**1.000**

Spot scat

Ecological groups	Proportion
Benthic invertebrates	0.05
Zooplankton	0.800
Phytoplankton	0.150
Sum	**1.000**

Benthics

Ecological groups	Proportion
Nekton	0.667
Sergestid shrimp	0.333
Sum	**1.000**

Ponyfishes

Ecological groups	Proportion
Shrimp	0.0002
Sergestid shrimp	0.0074
Benthic invertebrates	0.6182
Zooplankton	0.2951
Sum	**1.000**

Threadfin

Ecological groups	Proportion
Nekton	0.052
Shrimp	0.225
Sergestid shrimp	0.370
Benthic invertebrates	0.074
Zooplankton	0.279
Sum	**1.000**

Croakers

Ecological groups	Proportion
Nekton	0.060
Shrimps	0.110
Sergestid shrimp	0.349
Crab	0.002
Benthic invertebrates	0.011
Zooplankton	0.468
Sum	**1.000**

Benthopelagics

Ecological groups	Proportion
Nekton	0.243
Shrimps	0.094
Sergestid shrimps	0.465
Crab	0.018
Benthic invertebrates	0.138
Detritus	0.042
Sum	**1.000**

Appendix 2 (cont.)

Rainy season

Rays

Ecological groups	Proportion
Shrimp	0.317
Sergestid shrimp	0.017
Crabs	0.083
Zooplankton	0.583
Sum	**1.000**

Anchovies

Ecological groups	Proportion
Nekton	0.006
Shrimp	0.045
Sergestid shrimp	0.613
Crab	0.004
Benthic invertebrates	0.143
Zooplankton	0.189
Phytoplankton	0.0001
Sum	**1.000**

Sardines

Ecological groups	Proportion
Nekton	0.222
Sergestid shrimp	0.091
Crab	0.004
Benthic invertebrates	0.329
Zooplankton	0.269
Phytoplankton	0.002
Detritus	0.083
Sum	**1.000**

Catfishes

Ecological groups	Proportion
Nekton	0.272
Shrimps	0.123
Sergestid shrimp	0.013
Crab	0.037
Benthic invertebrates	0.500
Zooplankton	0.055
Sum	**1.000**

Mullets

Ecological groups	Proportion
Benthic invertebrates	0.168
Zooplankton	0.020
Phytoplankton	0.477
Detritus	0.335
Sum	**1.000**

Pelagics

Ecological groups	Proportion
Nekton	0.553
Shrimps	0.017
Sergestid shrimp	0.052
Benthic invertebrates	0.200
Zooplankton	0.176
Detritus	0.002
Sum	**1.000**

Perchlets

Ecological groups	Proportion
Nekton	0.002
Shrimps	0.007
Sergestid shrimp	0.120
Benthic invertebrates	0.772
Zooplankton	0.099
Sum	**1.000**

Spot scat

Ecological groups	Proportion
Benthic invertebrates	0.584
Zooplankton	0.009
Phytoplankton	0.407
Sum	**1.000**

Benthics

Ecological groups	Proportion
Zooplankton	0.8571
Benthic invertebrates	0.3333
Sergestid shrimp	0.0476
Sum	**1.000**

Ponyfishes

Ecological groups	Proportion
Nekton	0.005
Shrimp	0.054
Benthic invertebrates	0.608
Zooplankton	0.324
Detritus	0.009
Sum	**1.000**

Threadfin

Ecological groups	Proportion
Shrimp	0.085
Sergestid shrimp	0.708
Crab	0.012
Benthic invertebrates	0.128
Zooplankton	0.068
Sum	**1.000**

Croakers

Ecological groups	Proportion
Nekton	0.015
Shrimps	0.291
Sergestid shrimp	0.198
Crab	0.036
Benthic invertebrates	0.328
Zooplankton	0.103
Phytoplankton	0.029
Sum	**1.000**

Benthopelagics

Ecological groups	Proportion
Nekton	0.032
Shrimp	0.012
Crab	0.223
Benthic invertebrates	0.670
Detritus	0.063
Sum	**1.000**