The Environment in Asia Pacific Harbours

Edited by

Eric Wolanski

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FOREWORD

In the USA, Asia and Europe, as well as worldwide, trade is growing rapidly and much of it depends on shipping. This is leading to the development of mega-cities and mega-harbours. The marine environment is degrading. Is increasing trade ecologically sustainable? This book addresses this question through harbours in the Asia Pacific region, including Tokyo Bay, the Pearl Estuary, Hong Kong, Shanghai, Ho Chi Minh City, Manila Bay, Jakarta Bay, Bangkok, Singapore, Klang, Pearl Harbour, and Darwin. Much of the world trade goes through these harbours. This book demonstrates, through the writing of eminent scientists in each of these countries, the oceanography and ecosystem science necessary to understand how these urbanised marine ecosystems function. It offers science-based solutions to achieve ecologically sustainable development. These lessons are important not only for the Asia Pacific Region, including Australia, but also worldwide.

The book is a wake-up call that all the countries in the Asia Pacific are facing the same, serious socio-economic and environmental problems with varying scales. Each of these countries addresses these issues differently. This book shows that we have much to learn from each other to ensure that development does not need to be at the cost of the environment. I commend this book for its comprehensive coverage of the links between oceanography, ecosystem processes, and socio-economic issues. I hope it will create constructive discussion and awareness of the potential pitfalls and possibilities for the Asia Pacific region and the need for integration our efforts to deal with these issues.

This book by Eric Wolanski, a leading scientist at the Australian Institute of Marine Science should be taken seriously by all governments throughout the region.

The Right Honourable Malcolm Fraser, A.C., C.H.
Former Prime Minister of Australia

PREFACE

We live in a world that is increasingly dependent on international trade and transport. Measured both by volume and by value, most imports and exports travel by sea. Ports and harbours are the essential gateways through which all this marine traffic must pass. Expansion is leading to the development of mega-cities and mega-harbours. Inevitably, these are under further pressures to expand, and to work more efficiently. At the same time there is increased awareness of the need for maintaining healthy marine environments in and around these busy coastal areas. In many cases, these marine environments are degrading. Coastal managers and politicians are asking whether, and if so how, increasing trade can be balanced with ecologically sustainable environments.

This book addresses this challenge by presenting a series of studies of harbours in the Asia Pacific region, including Tokyo Bay, the Pearl Estuary, Hong Kong, Shanghai, Ho Chi Minh City, Manila Bay, Jakarta Bay, Bangkok, Singapore, Klang, Pearl Harbour, and Darwin. Much of the world trade goes through these harbours. Each individual harbour has its own special circumstances. Nevertheless, internationally there is much to be learned by exchange of information on existing management practices in different ports, and within different coastal areas.

These detailed examples demonstrate, through the writing and insights of eminent scientists in several countries, the oceanography and ecosystem science necessary to understand how these urbanised marine ecosystems function. The book offers science-based solutions to achieve ecologically sustainable development. These lessons are fundamentally important for the Asia Pacific Region, but they will also substantially inform similar analyses of port and harbour management and practices worldwide.

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CHAPTER 1

INCREASING TRADE AND URBANISATION OF THE ASIA PACIFIC COAST

ERIC WOLANSKI

1. INTRODUCTION

Is increasing trade environmentally sustainable? My introduction to this question started in 1969 on a road trip from Princeton to New York City. The bus went along estuaries that were heavily industrialised and polluted; the water was black and litter was everywhere to be seen in intertidal areas. My professors at Princeton University told me that this was normal but that the pollution footprint of the harbours was small because people lived elsewhere. In other words, harbours and urbanisation did not simultaneously degrade the marine environment at the same sites.

Over the last twenty years I developed a scientific collaboration with leading marine scientists in the Asia Pacific region. It became apparent to us that the footprints of harbours and urbanisation have now merged. Environmental degradation is exacerbated in this region where urbanization has already reached unprecedented levels in the coastal zone where mega-cities and mega-harbours have developed and are still rapidly growing. This generates dramatic, complex, and dynamic human-induced changes in coastal ecosystems, where environmental degradation is significant and growing. This degradation ranges from the loss of biodiversity to the ecosystem collapse and toxic algae blooms.

On a scientific level, the challenges are enormous to understand the cumulative impacts on the aquatic ecosystems of urbanization and harbour development and operations. Yet science is needed to provide long-lasting solutions to environmental degradation. When the idea was first mooted to produce this book, some prospective authors expressed their concerns that the issues were politically sensitive. Yet all the authors produced thorough, well researched chapters openly detailing the environmental and socio-economic issues facing their waters.

What emerged is this book demonstrating the different solutions and pitfalls, successes and failures in a large number of ports and harbours in the Asia Pacific Region shown in Figure 1, and this is based on process-oriented ecosystem science and it is aimed at management. The study sites include Tokyo Bay, the Pearl Estuary, Hong Kong, Shanghai, Ho Chi Minh City, Manila Bay, Jakarta Bay,

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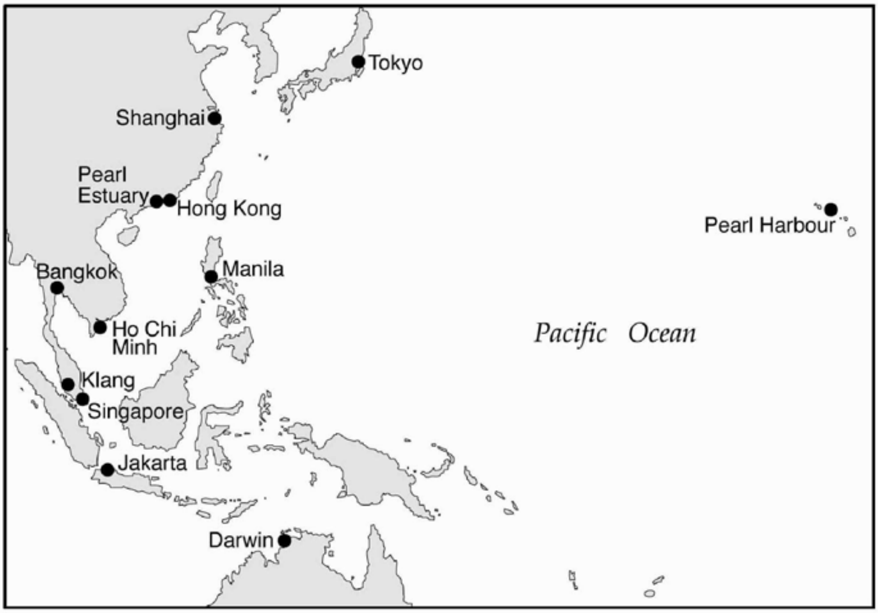

Figure 1. *A location map of harbours described in this book.*

Bangkok, Singapore, Klang, Pearl Harbor, and Darwin. About seventy million people live on their shores, and 600 million people in their watershed. The authors of these chapters are eminent, leading scientists in their respective countries; they bring an immense body of knowledge and experience. They demonstrate the range of multi-disciplinary sciences and the experience that is needed to enable ecologically sustainable development.

This book demonstrates how these urbanised water bodies function as ecosystems, by focusing on the links between physical and biological processes. In most of these environments many life history stages are planktonic. As a result, physical transports and processes play a key role in the transport of nutrients and pollutants, while biological and chemical processes determine the ecosystem community response to the stresses. Thus physical processes exert a profound influence on biological processes and their combination determines the ecological services – and the public health risks - that these waters deliver to the human population on its shore.

By bringing together these leading scientists, this book highlights the many similarities and dissimilarities brought upon by different scientific and management approaches, governance and socio-economic issues and different levels of development. Ecosystem management is shown to have developed rapidly – with local convergences and divergences in approaches, techniques and support funding. These experts provide hints of long-term solutions to enable socio-economic developments while maintaining, or restoring, the vital ecosystem services provided by the estuarine and coastal waters and ensuring public health.

2. CASE STUDIES

2.1. Tokyo Bay

Chapter 2 by Keita Furukawa and Tomonari Okada describes the environmental issues in Tokyo Bay. The bay is an enclosed, most heavily populated, and densely utilized bay in Japan. It is vulnerable to environmental degradation. A new government initiative was initiated in 2001 to promote environmental restoration in coastal zones. The Tokyo Bay Restoration Plan was adopted in March 2003 to clarify objectives and action plans. Sharing a good understanding of the scientific and engineering background of wetland restoration projects with stakeholders and their involvement of them are keys to achieving the wise use of the coastal environment.

Chapter 3 by Hirofumi Hinata and Keita Furukawa shows that the ecosystem health of Tokyo Bay depends on the presence of tidal flats – most of which have been destroyed by harbours. To undertake the task of identifying where and how to restore tidal flats as an important coastal ecosystem, the "Asari Project" was initiated to determine the existence of an ecological network in Tokyo Bay. The project maps the distribution patterns of the Asari short-necked clam larvae. The series of distribution of D-shaped larvae (juvenile Asari clam larvae typically 100 μm in size) and umbo larvae (fully grown Asari clam larvae typically 170-180 um in size) show successive advection by currents in the bay. This suggests the existence of an ecological network. The quantification of this network was made possible by numerical modelling that supports the existence of an ecological network.

In chapter 4, Keisuke Nakayama demonstrates that the oceanography is a dominant process determining ecosystem health. Both long-term (e.g. seasonal) and short-term (e.g. days to weeks) processes control the water circulation in Tokyo Bay. Two dominant processes that control the water circulation in Tokyo Bay, namely the fortnightly modulation of the estuarine circulation, and the change over several days of the horizontal circulation around the head of Tokyo Bay. In chapter 5, Hinata demonstrates that the intrusion of oceanic water in Tokyo Bay has a significant impact on the environmental health of the bay. This intrusion occurs as a series of events, with the dominant ways being a classical, two-layer, estuarine circulation, and a three-layer flow structure. These have quite different environmental consequences in terms of the net fluxes of heat, nutrients and suspended matter.

2.2. Shanghai and the Changjiang Estuary

In chapter 6, Jianrong ZHU and colleagues describe the impacts of the deep waterway project on the Changjiang Estuary. The improved 3-D ECOM model was applied to simulate the impact of the project on saltwater intrusion. The saltwater intrusion after the deep waterway project has been distinctly decreased in the North

Channel and at the lower section of the North Passage, while it has worsened at the upper section of the project site and in the South Passage. The resulting estuarine dynamics processes are discussed. The impacts of the deep waterway project on planktons and benthos are also analysed. It is suggested that the degradation of biodiversity in the estuary is due to the cumulative impacts of a number of human activities including eutrophication from agriculture and sewage discharge, land reclamation, and estuarine engineering projects.

In chapter 7, Jing ZHANG and colleagues describe how anthropogenic perturbations over the last five decades have changed the landscape and hydrographic features over the Changjiang (Yangtze River) drainage Basin, following rapid population increases. Coastal ecosystems responded by changes in food-web structure and function. In the delta region, human being influence has been identified over a broad context of studies, either for natural science and social activities. Relative high levels of organic pollutants (e.g. DDTs, PCBs and PAHs) have been found in the river channel affected by the directed waste drainage, while in the coastal areas pollutants in bottom sediments show a gradient of decrease. Core sediment samples in the Changjiang delta region show the maximum of pollutant concentrations in 1970s and provide with evidence of relief of pollution effect after 1990s. Reclamation in the delta region has induced rapid adjustment of tidal flat and affects the landscape of wetland, shown by the profiles from salt marsh to bare-flat. Although the nutrient concentration is rather high in the Changjiang, the reported harmful algal bloom events take place in the area of offshore waters, i.e. 50-100 km off the river mouth, owing to high turbidity in the delta region. Constructions of harbour and jetty have caused eternal loss of habitats in the Changjiang Delta Region, which is of critical importance in recruitment of fishery species (e.g. spawning ground), resulting in unsustainability of living resources. While human settlement in the Changjiang Delta Region is expected to be continued in the near future, the conflicts of shortage of land surface area and industrialization is becoming one of the bottlenecks for the sustainability of social society. While reclamation of wetlands in the coastal area is required to maintain the progress of urbanization in coastal region of China (e.g. Shanghai), the consequence is the loss of habitats for the wildlife, spawn and hatch grounds for marine living resources, which change the community structure and put those species of economic values in danger or extinction.

Pollution by trace metals and synthetic organic materials can cause the problems at molecular level and induce the genetic diseases, which in turn alter the strategy at genetic level and change the whole system via food-chain. An increased discharge from the Changjiang into the East China Sea has caused frequent HABs (i.e. harmful algal blooms) in coastal waters further offshore from delta region, and a serious hypoxia problem has recently reported in literature, which induces the concern of public society. Dredging in the Changjiang Estuary for navigation and construction of jetty in the main channel have been reported to further destroy the fishing ground and growing filed, and block the migratory route of traditional economic species in the region. In that case, the Changjiang Estuary is losing its traditional values at ecosystem level, and the role of providing multiple functions of the delta region is turning to the simplified service system, e.g. land area for settlement and water-way for transportation and trade for economics.

The on-going engineering constructions in the Changjiang Delta Region are expected to further modify the habitats and affect community structure of wildlife, which can affect the whole ecosystem via food-web interactions, unless proper management is made to protect the habitats and to improve water environment at ecosystem level.

2.3. Pearl River estuary

The Pearl River Estuary is densely distributed with ports. Guangzhou, Shenzhen and Hong Kong are the three principal ports in the region. The Guangzhou Port has 140 berths and 22 mooring areas. General cargo ship of 13.5 m draught and container ship of 4th and 5th generation's ships can navigate through during the flood tides. The Shenzhen Port has 102 berths with capacity over 500 tons, in which 23 are above 10,000 t. Shenzhen is the fourth largest container transportation base in the world. Wastewater discharge in the Pearl River Delta is about 31.6×10^8 m^3 yearly, mainly from the rapidly developing cities on the Chinese mainland.

Mingjiang ZHOU and colleagues in chapter 8, Lixian DONG and colleagues in chapter 9, and Yian LI and colleagues in chapter 10, describe the geographic setting and the physical oceanography of the Pearl River estuary. The river discharges into the South China Sea (SCS) through eight distributaries, locally called "the eight Gates". The four eastern gates discharge their waters into the "Lingdingyan", called here the "Pearl River Estuary" (PRE). The PRE has two deep channels used for shipping, the western channel and the eastern channel. The average yearly discharge of the Pearl River is around 330×10^9 m^3. The annual sediment load is about 88.72×10^6 t. The estuary is poorly flushed and environmental degradation is severe.

The Pearl River Delta is one of the most populated areas in the Chinese mainland with a population over 28 million and a density of 674 people km^{-2}. The delta's per capita GDP exceeded US$ 3600 in 2000. In 1999, the exported goods from the Pearl River delta valued at US$ 67 billion.

This estuarine ecosystem is profoundly degraded. Total pollutant load and some organic pollutant loads carried to the Pearl River Estuary (PRE) from the Pearl River, as well as concentrations of dissolved heavy metals, oil and grease in the PRE, are presented. The most prominent characteristic of the water quality of the PRE is the high nitrate to phosphate ratio. In the upper estuary, this ratio (N:P) can be up to 200:1 in the surface layer. About 300 phytoplankton species have been recorded for the PRE, including diatoms, dinoflagellates, and other species, There is a rapid increase of the frequency of harmful algal blooms (HABs) in this area over the last two decades. Most of these HAB events happened in Hong Kong waters and in bays further east of Hong Kong and in Shenzhen Bay at the east side of the middle PRE.

The fast human population growth and the rapid expansion of the mariculture industry have exerted much stress on the coastal environment off the Pearl River delta. As in elsewhere of China, there is a rapid increase of the frequency of harmful algal bloom (HAB) in this area over the last two decades. Most of these

HAB events happened in Hong Kong waters and in bays further east of Hong Kong. However, outbreaks of HAB in the PRE itself also happened more frequently in recent years, especially in Shenzhen Bay at the east side of the middle PRE. HABs in PRE are mostly caused by dinoflagellate species, accounting for 72.7% of the total events. The high concentration of inorganic nitrogen, i.e., eutrophication, is the direct cause for the increase of HAB frequency in the PRE. However, phosphate and turbidity are both important factors influencing the HAB occurrence. Nutrients are high in the northern part of the estuary, but the chlorophyll a concentration is generally low there because of the high turbidity. There are two relatively high concentration of chlorophyll a in the estuary, one near the Shenzhen Bay and the other southwest of Hong Kong. The concentration in the two areas is in general relatively low. In fact, HAB events in the PRE often occur at these two sites. However, relationship between the HAB and the environmental conditions in the PRE has not yet been established.

2.4. Hong Kong

In chapter 11, Nora TAM describes the findings of pollutions studies in mangroves. Mangroves are important inter-tidal wetlands found along the coastlines of tropical and subtropical regions. Historically, many urban and industrial centers were established adjacent to estuaries fringed by mangroves, and mangrove swamps have been used as convenient dumping sites for waste generated from human activities. In recent years, rapid development in coastal areas, urbanization and industrialization have led to significant increases in anthropogenic inputs of pollutants to the mangrove ecosystems in Hong Kong and mainland China. Elevated concentrations of nutrients, heavy metals and toxic organic pollutants have been recorded in mangrove sediments, and the degree of accumulation is found to be related to the sources of pollution and characteristics of the sediments. Mangrove sediments act as pollutant sinks, and mangrove plants that are specially adapted to stressed environments are able to tolerate pollution to a certain extent. It seems that mangrove ecosystems have a considerable capacity to withstand domestic, poultry and industrial discharges, their sediments, plants and microorganisms are capable of retaining and transforming pollutants. This chapter reviews the concentrations of nutrients, heavy metals and persistent toxic organic pollutants in mangrove sediments in Hong Kong and mainland China, identify the sources of pollution, evaluate the impacts, and explore the potential of employing mangrove wetlands for waste and wastewater treatment.

In chapter 12, Kwok-Leung PUN describes the characteristics of the Hong Kong coastal environment. The increase in pollution loads entering the harbour causes eutrophication of the coastal waters leading to the increase in red tide incidents affecting many fish culture zones in Hong Kong. The large-scale development projects involving a massive reclamation and export of effluent may also impact on the fish culture zones and the other water-sensitive receivers. The feasibility and planning of the projects relies on the application of water quality models to undertake impact assessment. Relevant water quality standards and approach for water quality model validation are presented. The experience in applying water quality models for prediction of water quality impacts associated with land reclamation, bridge construction, and export of effluent projects in Hong Kong is described. This chapter presented the general characteristics of the Hong Kong coastal environments and the potential impacts associated with large-scale developments. The approach for water quality model validation and application of the validated models for prediction of water quality changes in reclamation, bridge construction and export of effluent projects are described in this chapter. Dredging and filling activities are of major concern in the reclamation and bridge construction projects. Export of effluent from sewage treatment works to less sensitive water bodies on one hand reduces the eutrophication in the originally affected water body and on the other hand introduces additional loads into the receiving water body. Through the application of water quality modelling, the potential impacts and benefits of the project can be examined providing accurate data for decision making on the feasibility of the development project.

In chapter 13, Joseph LEE and collaborators describe the ecosystem dynamics of Hong Kong coastal waters. Hong Kong is a mega-city with a population of 6.7 million; it is situated at the mouth of the Pearl River Estuary in southern China. The marine resources in the 1800 km^2 of coastal waters are intensively utilized. The water quality has been deteriorating as a result of the high nutrient loads from the rapidly urbanised and industrialised Pearl River Delta (PRD) region, and significant sewage discharges into Victoria Harbour. Hong Kong waters are relatively unique because of the frequent occurrence of harmful algal blooms, and the complexity and richness in eutrophication dynamics within a relatively small area. Viewed against the increasing nutrient loads, it is surprising that the eutrophication impacts on Hong Kong waters are not worse than they are. This paper gives an overview of the temporal and spatial dynamics of algal blooms. The tidal current and salinity structure in the partially-mixed estuary are elucidated using numerical results from a calibrated three-dimensional hydrodynamic model. Physical and biological interactions are discussed in relation to seasonal phytoplankton ecology as well as episodic events. It is shown that key factors that govern the eutrophication dynamics in Hong Kong waters and Victoria Harbour include: tidal mixing, wind, stratification and vertical stability, flushing rate, nutrient and light limitation.

2.5. Pearl Harbor

In chapter 14 Steve Coles describe the environmental setting of Pearl Harbor. This is the largest and most enclosed harbour or embayment in the Hawaiian Islands and consists of three main coves that receive most of the freshwater runoff from central O‘ahu. Traditionally the harbour was highly utilized for fish pond culture, but water quality degraded through the nineteenth and much of twentieth century. The harbour has been controlled by the U.S. Navy since the annexation of Hawai‘i in 1898, and enlargement of the entrance channel and construction of the Pearl Harbor Navy Base began in 1910-11. Development of the Navy base and the surrounding urban areas through the next 60 years resulted in elimination of most of the fishponds, hardening of shorelines by construction of piers and dry docks, disposal of sewage and other pollutants, sedimentation from land runoff, and periodic dredging to remove accumulated sediments. Along with changes to the marine community that can be assumed to have resulted from these stressors, opening the harbour to Navy shipping provided a vector for the introduction of nonindigenous species from ships and other vessels such as barges and floating dry docks. Studies evaluating the occurrence and estimated time of arrival of introduced species indicated that approximately 23% of the identifiable marine biota in 1996 were composed of known or suspected introduced species. Introduced species have been increasing in the harbour since 1900, with decadal peaks in new introductions relative to total newly reported taxa apparently occurring during wartime periods in 1910-20 and 1940-50 when ship traffic in the harbour would been maximal. These findings are compared with information available for other harbours in Hawai‘i and the tropical Pacific, and a need is indicated for similar studies of introduced marine species in harbours elsewhere in the Asia-Pacific region.

2.6. Bangkok and the upper Gulf of Thailand

In chapter 15 Suphat Vongvisessomjai provides information on the physical environment and the oceanographic conditions is required to understand the physical processes in the gulf and the coastal zone for purpose of infrastructure development including port construction and coastal management. This information includes (A) the tracks and strength of cyclones, (B) winds and waves, and (C) tides and tidal currents including flushing time of pollutants. These conditions are included in the design parameters for the construction of three important ports of Thailand.

In chapter 16 Gullaya Wattayakorn presents an overview of the environmental problems of the Gulf of Thailand from the rapid population and economic growth. The available information indicates that the main degradation issues in the Gulf of Thailand are overexploitation of fisheries, loss of habitat and marine pollution. Among these, loss of habitat is considered a serious problem throughout the Gulf. Mangrove destruction is the most obvious and has probably had the greatest loss. Coral reefs are subjected to a number of disturbances, which are often irreversible. Seagrass beds are destroyed by fishing gear and by sediment from bad agricultural and engineering practices, forest clearing and runoff from cities. In addition, the increased frequency of red tides in recent years may be an indication of the changing conditions of coastal aquatic environment. There is a need to accentuate and develop compatible and sustainable policies in resource management to retain the systems indefinitely and also to enhance continuous economic returns.

2.7. Ho Chi Minh City

In chapter 17 Nguyen Huu Nhan describes the environmental setting of Ho Chi Minh City (HCMC), including the Thi Vai–Vung Tau (TV-VT); this area had, has, and will have for the foreseeable future the largest harbour network in Vietnam. This network plays a very important role in Vietnam’s social and economic development. All shipping routes operate in very sensitive regions such as wetlands, mangroves, urban centers with high population density, and industrial zones. The environment faces serious challenges. At present, the environment in harbours of HCMC suffers from pollution caused by domestic, industrial, aquaculture and shipping route wastes; all these wastes are almost untreated before discharge. The pollution status of water and bottom sediment in Sai Gon and Thi Vai harbour groups is alarming. The risk and frequency of accidents with severe environmental impact are high. Urgently needed remedial measures include (1) the construction of treatment facilities for wastes from domestic and industrial areas, from ships, and aquaculture activities; (2) cleaning polluted bottom sediments in HCMC, Go Dau and Thi Vai harbours; (3) stopping the destruction of the Can Gio Mangrove Reserve and the decline of its biological resources by human activities and shipping routes; (4) to set up effective tools to prevent and respond to environment pollution accidents.

2.8. Manila Bay

In chapters 18 and 19, Gil Jacinto and colleagues describe the environmental setting and issues of Manila Bay. The bay has a surface area of 1,700 km^2 with a 190 km coastline that opens to the South China Sea. There are seven rivers surrounding the bay with two of the biggest rivers contributing 70% of the freshwater runoff. It has a drainage area of 17,000 km^2 with approximately 16 million in population. More than 3,000 years ago, Manila Bay was connected to Laguna Lake, which in recent years have separated except for a small interaction through the Pasig River. Shoreline positions of the bay have changed due to land reclamation, conversion of mangrove and mudflat areas into fishponds, soil erosion, siltation, and sea level rise. Estimates of sedimentation for different parts of the bay have ranged from 0.6 to 9 cm yr^{-1}. Wet and dry periods are the two pronounced seasons in the bay, and the prevailing wind patterns are the NE monsoon, Trades, and SE monsoon. Tides are mixed diurnal with an average range of 1.2 m during spring and 0.4 m during neap. Freshwater discharge, tides and winds affect the circulation patters in Manila Bay. Manila Bay is the most strategically and economically important body of water in the Philippines. It has a significant socio-economic role for Metro Manila and the surrounding provinces that share its long coastline. It is recognized under the Manila Bay Declaration in 2001 as a source of food, employment and income for the people, the local and international gateway of the country to promote tourism and recreation. It is also important because of its cultural and historical heritage. It is major fishing ground and being almost completely landlocked, Manila Bay is considered one of the world's great harbours. The biggest shipping ports, ferry terminals, fish port and yachting marina are found in the bay. An average of 30,000 ships arrive and depart from these ports annually to transport passengers, manufactured goods and raw materials. The beaches along parts of the bay encourage tourism in the area. However, the bay has undergone significant changes in its water and sediment chemistry, linked heavily to the increase in population and other human activities. The bay has become a receiving site for domestic and industrial sewage. Levels of total and fecal coliform, dissolved oxygen, oil and grease, and nutrients are the indicators of the decline in water quality. Heavy metals such as Cd, Hg, Zn, Pb, and Cr; and total polyaromatic hydrocarbons in the sediments showed localized contamination.

Fisheries resources have measurably declined from the 1940's onward, primarily due to over-fishing or over-collection. There has been a decline in trawl catch per unit effort or CPUE (kg h^{-1}) from 46 in 1947 to 10 in 1993. The demersal biomass decreased from 4.61 mt km^{-2} or 8,290 tons in 1947 to 0.47 mt km^{-2} or 840 tons in 1993. In some areas the poor management of shellfisheries resulted to unstable production of commercially valuable mussels and oysters, disappearance of the windowpane oyster and contamination of shellfish particularly with fecal coliforms.

Manila Bay has a wide range of environmental problems that need to be addressed - from land-based and sea-based sources of pollution to harmful algal blooms, subsidence and groundwater extraction, overexploitation of fishery resources, and habitat conversion and degradation. However, there are reasons to be optimistic. There is greater accountability expected of public officials vis-a-vis environmental laws, significant and increasing infrastructure investments to treat and reduce domestic sewage discharges into the bay, the implementation of the Manila Bay Environmental Management Project, and the adoption of Integrated Coastal Management by local government units and communities around Manila Bay. Time will tell if the political will to implement these measures is sufficient to allow the bay to revert to be a clean, safe, wholesome, and productive ecosystem for the present and future generations.

2.9. Port Klang

In chapter 20 Choon-Weng Lee and Chui-Wei Bong analyse the carbon flux through bacteria in Port Klang waters, a eutrophic tropical environment from September 2004 until February 2005. Water quality was poor due to low dissolved oxygen (DO) concentration (<200 μM), and high total suspended solids (TSS) (>260 mg l^{-1}). TSS was mainly inorganic in nature, with particulate organic matter <5% of TSS. Two episodes of hypoxia (DO<125 μM) were observed in early December 2004 and February 2005. Based on marine water quality data collected by the Department of Environment of Malaysia, the water quality at and around Port Klang deteriorated from 1990 to 2003. Over 10 years (1994–2003), TSS increased 132 mg l^{-1}, and DO decreased by 48 μM. The NO_3:PO_4 ratio was low, ranging from 0.05 to 0.38, suggesting nitrogen limitation for the phytoplankton. Gross primary production (GPP) correlated significantly with NO_3 (R^2=0.867, n=5, $p<0.05$). The low NO_3 concentration in February 2005 could have limited GPP, and indirectly triggered hypoxia. GPP correlated with community respiration (R^2=0.956, n=5, $p<0.01$) except in February 2005 when there was uncoupling between primary production and heterotrophy. Heterotrophic metabolism was probably supported by other sources of allochthonous organic matter (e.g. the Klang River) during this period.

2.10. Singapore

In chapter 21 Karina Yew-Hoong GIN and colleagues present an overview of the phytoplankton composition in Singapore coastal waters and their relationships with nutrient and environmental conditions. A variety of techniques were employed to determine the structure of the phytoplankton community, including sophisticated methods such as flow cytometry and high performance liquid chromatography (HPLC) but also traditional methods, such as microscopy and extracted chlorophyll measurements. Using data collected in the last six years, waters in the Johor Strait were found to be more eutrophic than the Singapore Strait, with chlorophyll levels reaching as high as 60 $\mu g\,l^{-1}$, consistent with the higher nutrient concentrations measured in the Johor Strait. Nutrient enrichment tests showed that Singapore waters are generally nitrogen limited although for the Johor Strait, nutrient limitation can also switch to phosphorus. The size structure of phytoplankton in the Johor Strait is skewed to larger microplankton whereas for the Singapore Strait, smaller pico- and nano-plankton dominate. Compared to the Singapore Strait, algal blooms are a frequent occurrence in the Johor Strait although there are few documented cases of harmful algal blooms (HAB). To help alleviate eutrophication problems, an integrated approach is needed where control measures are used in conjunction with monitoring and numerical modeling.

In chapter 22 Loke Ming Chou describe the marine habitats of Singapore waters. Shipping activities and port infrastructure development are intense. Most of the country's limited marine territory is under port authority. Coastal reclamation, seabed dredging to deepen shipping lanes or extract sand deposits, and the dumping of dredged spoils at sea all add further impacts to marine biodiversity. Natural habitats are reduced in extent or degraded, while newly created ones favour biological communities of different species composition. Species eliminations are highly localised and species distribution patterns are altered in response to changing environmental conditions, especially below 6 m depth. Overall, complete species extinction from Singapore waters is limited and less than expected considering the scale, variety and intensity of impacts. Abundance, however, is reduced with many species now less common or rare. The lower than expected rate of species richness decline is attributed to habitat resilience from the originally rich biodiversity and strong marine pollution control measures. Port development and marine biodiversity are not mutually exclusive. Further reduction of impact effects on marine biodiversity is possible provided that some attention is focused on marine conservation.

In chapter 23 Eng Soon CHAN and colleagues present an overview of the physical oceanography of Singapore coastal waters. Flows in this domain are driven mainly by tides and seasonal net pressure gradients. The interaction of tidal streams from the Java Sea, the Malacca Strait and the South China Sea is complex and the transport of discharges such as oil spills into the domain is typically trans-boundary. Whilst the physical processes are important in the prediction of oil spill trajectories and dilution, the coupling of these processes to the biology and chemistry of the water body is important in determining the fate and environmental impact of spills. A comprehensive oil spill-food chain model, which combines a multi-phase oil spill model with a simple food-chain model, has been applied to the Evoikos-Orapin Global oil spill in the Singapore Strait. The results of the simulation showed that the concentrations of anthracene in phytoplankton, zooplankton, small fish, large fish and benthic invertebrates one day after the spill were below the lethal toxicity levels (LC_{50}). Fertilization experiments to examine the bioremediation capabilities of tropical sands demonstrated that nutrient addition was able to significantly accelerate the natural degradation process. For the more persistent organics in the marine environment, baseline studies showed that concentrations of polycyclic aromatic hydrocarbons (PAHs) measured in Singapore's coastal waters were generally higher than levels reported elsewhere, whereas organochlorine pesticides (OCPs) and polychlorinated biphenyls (PCBs) were generally lower than the reported levels for other Asian countries.

2.11. Jakarta Bay

In chapter 24 Dietriech Bengen and colleagues describe the 400-year long environmental legacy of Jakarta Bay. In that period Jakarta Bay rose to international prominence as a leading harbour in Southeast Asia. During the rapid economic growth during the l980s the port facilities were rapidly expanded and large areas of the bay foreshore and wetlands were reclaimed to accommodate industrial and urban development. The resulting ecological and social costs are high. Jakarta Bay is perhaps the most polluted harbour in Asia and has very little intact fisheries. What fisheries still exist has a non-original structure and comprises opportunistic species that can exist in the now heavily polluted waters. The combination of inappropriate past development and continuing lack of coordinated bay governance has created a legacy of severe environmental damage – several of the former "thousand islands" that comprised a small archipelago of coral cays in the outer arc of the bay have disappeared due to sand mining and those that remain are among the most threatened coral reefs in Asia. The authors describe how, through a combination of neglect and ignorance, Indonesian society now faces major challenges to sustain the economic, social and ecological values of the bay. There are growing calls to reduce pollution, protect traditional fishing communities, restore coral reefs and mangrove systems and optimize bay use for industrial, urban, tourism and conservation uses. However, efforts to address bay management on a more integrated and long-term basis are only now emerging and will require a far greater level of political commitment if current degrading influences are to be reversed and an effective and comprehensive management regime introduced.

2.12. Darwin

In chapter 25 David McKinnon and colleagues describe the environmental setting and health of Darwin harbour. This is Australia's only tropical harbour and is Australia's closest port to South East Asia. The environment within Darwin Harbour is still relatively pristine, with extensive mangrove forests fringing the harbour and a diverse range of ecosystems within the harbour itself, including some coral-dominated communities. The harbour is macro-tidal (7.8 m tidal range) and characterised by high turbidity. Standing stocks of nutrients are low (e.g. NO_3 ~0.3 μM in the dry season, 1.5 μM in the wet season). In spite of light limitation and low nutrient status, water column primary production is high - net primary production exceeds 2 g C m^{-2} d^{-1} in the wet season. Current anthropogenic inputs into the harbour amount to <2% of the nutrient demand necessary to sustain primary production by phytoplankton and mangroves. Ambitious plans to expand the port, including construction and dredging activities, combined with a large fish catch by recreational fishers present emerging threats to the ecosystem health of Darwin Harbour.

In chapter 26 David Williams and colleagues describe the water circulation in Darwin Harbour, focusing on its flushing properties and the fate of sand and mud. It is shown that (1) the wet season runoff is important to flush the harbour and to redistribute fine sediment, (2) that the upper reaches of the harbour are very poorly flushed during the eight-month long dry season with a residence time estimated to be at least twenty days, (3) that the harbour traps most of the fine sediment from runoff and redirects it into mud banks in embayments and mangroves wetlands, (4) that eddies shed by the complex bathymetry maintain sand shoals, (5) the harbour is an inverse estuary that imports oceanic water during the dry season, and (6) the harbour is a stratified estuary for a few days to a few weeks during the wet season, and a vertically well-mixed estuary the rest of the time. These findings have profound implications on development strategies for ecologically sustainable development.

In chapter 27 with colleagues I propose a simple ecohydrology model that has been developed, calibrated, and applied to Darwin Harbour. The model integrates physical and biological processes in the estuary and it predicts the ecosystem health as determined by the following variables: salinity, nutrients, detritus, suspended particulate matter, phytoplankton (two size classes), zooplankton (two size classes), detritivores, zooplanktivorous fish and carnivorous fish. The model is used to assess to what degree the estuarine ecosystem health may degrade as a result of possible future human activities in the catchment, particularly land clearing, nutrient enrichment, and destruction of mangroves. The model is a tool that may enable an interaction between scientists, economists, the public, and decision makers to enable the ecologically sustainable development of Darwin Harbour catchment based on ecohydrological principles.

3. LESSONS FOR THE FUTURE

In chapter 28 I attempt to synthesise the results from these case studies and to answer the question: are increased trade and urbanisation of the coastal zone ecologically sustainable? I also propose a strategy for ecologically sustainable harbours and coastal urbanisation.

CHAPTER 2

TOKYO BAY: ITS ENVIRONMENTAL STATUS – PAST, PRESENT, AND FUTURE

KEITA FURUKAWA AND TOMONARI OKADA

1. INTRODUCTION

1.1. Natural conditions

Tokyo Bay is centrally located in Japan, between latitude 35 00'N and 35 40'N, and longitude 139°40' E and 140°05'E (Figure 1). This region has a temperate, humid climate. The lowest temperature is approximately 5 °C during January and February, and the highest temperature is approximately 30 °C during July and August. The months with the least precipitation are January and February. There is heavy precipitation during the rainy season in June, and during September and October, when typhoons often hit Japan. The monthly precipitation is approximately 50 mm and 150 mm in, respectively, January and September.

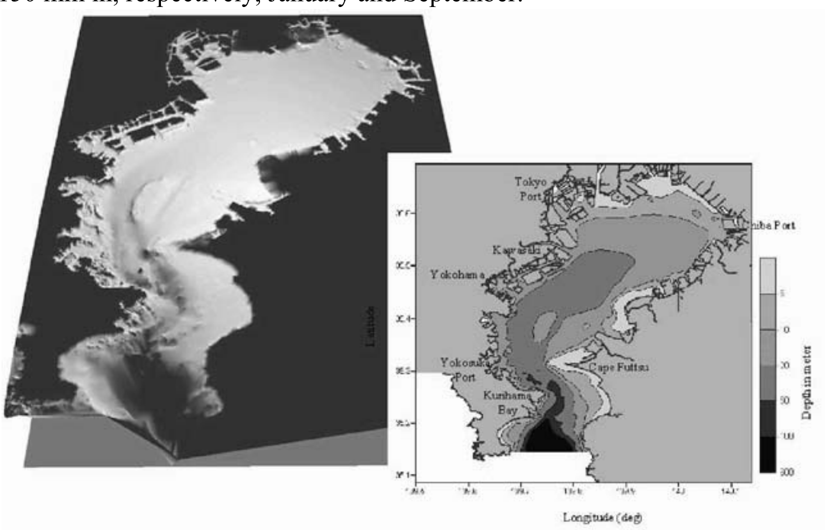

Figure 1. *Location map and bathymetry of Tokyo Bay.*

E. Wolanski (ed.), The Environment in Asia Pacific Harbours, 15–34.

The bay is the area north of the broken line connecting the eastern and western points of Suzaki and Kenzaki (see Figure 2a). The bay has an open interchange with the Pacific Ocean. The Kuroshio Current in the Pacific Ocean flows near the mouth of the bay.

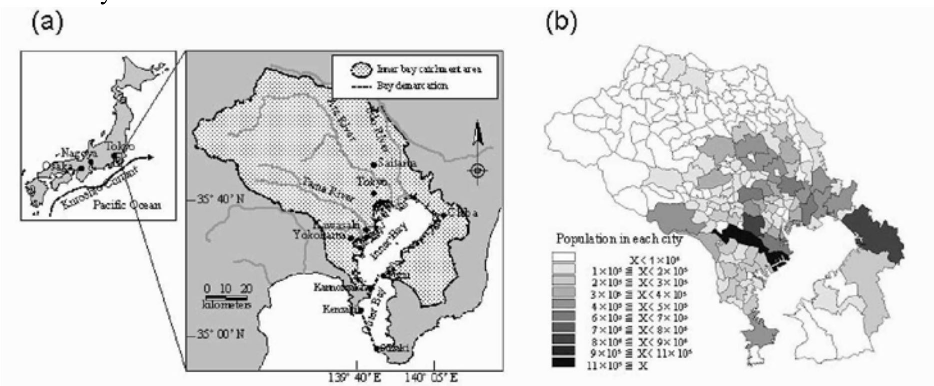

Figure 2. *Tokyo Bay catchment area. (a) location, (b) population in 2000.*

The bay has the shape of the letter "S." The narrowest width of the bay is 6 km on a line connecting the points of Futtu and Kannonzaki. Generally, the side to the North of that line is called the "inner bay", and the side to the South the "outer bay". In addition, the inner bay is often called simply "Tokyo Bay." In this chapter, "Tokyo Bay" refers to the inner bay. The inner bay has a length of 50 km and a width of 20 km. The average depth of the inner bay is 15 m, and its surface area is 960 km^2. The volume of the inner bay is 15 km^3. The catchment area of the inner bay is 7548 km^2. Its depth increases gradually from the head of the bay toward the mouth of the bay. The maximum depth of the inner bay is 50 m at its mouth. The seafloor is covered by silt or sand. The outer bay is deeper than the inner bay (Figure 1), with a maximum depth of approximately 600 m. The seafloor has a steep profile and is covered by rock or sand. The surface area of the combined inner bay and outer bay is 1380 km^2.

Rainwater falling in the catchment area of Tokyo Bay flows into the bay mainly through the Edo and Ara Rivers. Both rivers discharge into the head of the bay. The combined water discharge of both rivers accounts for approximately 50 % of all fresh water entering the bay. This water discharge forms a clear estuary circulation from the head of the bay toward its mouth.

The currents in the bay are caused mainly by tides, density gradients, wind stress, and the input of oceanic water.

The tides are typically semidiurnal, with tidal ranges of about 1.5 m during the spring tide and about 0.5 m during the neap tide. The maximum tidal current is about 1.2 m s^{-1} around Kannonzaki and about 0.2 m s^{-1} in the center of the bay. Generally, the residual current forms a clockwise circulation in the inner bay during summer.

1.2. Social background

Tokyo, the capital of Japan, is located in the catchment area of Tokyo Bay and has a population of 8 million people. Several other major cities are located in the inner bay catchment area: Yokohama, 3.5 million people; Kawasaki, 1.3 million people; Saitama, 1.0 million people; and Chiba, 0.9 million people (Figure 2 (b)). The inner bay catchment area has a total population of some 27.8 million.

The concentrations of population and industry in the catchment area of Tokyo Bay are very high according to estimates based on the population per surface area (= population in the catchment area/ surface area of the bay) and the total annual amount of freight at all ports in the bay. The population per surface area of Tokyo Bay is 29.0×10^3 people km^{-2}, while the population per surface area of San Francisco Bay and Chesapeake Bay in the United States of America are 8.1×10^3 people km^{-2} and 0.9×10^3 people km^{-2}, respectively (Table 1). In addition, the amount of annual shipping freight containers in Tokyo Bay is 5.8×10^6 TEU per year, while the amounts for San Francisco Bay and Chesapeake Bay are 1.9×10^6 and 1.6×10^6 TEU per year, respectively.

Table 1. *Comparison of the geographic and human use indicator between Tokyo Bay, Chesapeake Bay and San Francisco Bay. * Ogura (1993). * * International EMECS Center (2003). *** Degerlund (2005).*

	Tokyo Bay	San Francisco Bay	Chesapeake Bay
Surface area (km^2)	960*	1222**	18130**
Catchment area (10^3 km^2)	7.6*	156.0**	166.0**
Average depth (m)	15*	5**	6**
Population in the catchment area (in 2000) (10^6 people)	27.8	10.0**	15.7**
Population per catchment area (10^3 persons km^{-2})	3.7	0.06	0.1
Population per surface area (10^3 persons km^{-2})	29	8.1	0.9
Reclamation area (km^2)	249 (24.9 %)	240 (19.4 %)	12 (0.1 %)
Annual shipping freight containers (in 2000)(10^6 TEU)***	5.8	1.9	1.6

Some of the reasons behind the concentration of population and industry in the catchment area of Tokyo Bay include the following: (1) there are many suitable locations for ports in Tokyo Bay, where wave conditions are calm. (2) The catchment area of Tokyo Bay includes a wide plain (the Kanto plain) in Japan. (3)

The hinterlands of the bay are suitable for industry in Japan, which imports resources and exports products by sea. (4) The capital of Japan is located in the catchment area. (5) The climate of the catchment area is conducive to habitation.

The concentration of population and industry in the catchment area of Tokyo Bay has brought remarkable changes to its coastal area. Figure 3 illustrates the changes in the shape of the bay from 1920 to 2002. The reclamation of tidal flat areas and shallow water area increased rapidly from 1958 to 1976. Consequently, the surface area of Tokyo Bay decreased by 26% from 1900 to 2000. The present tidal flat area of Tokyo Bay is only 10 km^2, compared to 136 km^2 in 1900.

2. CHANGES IN THE ENVIRONMENT OF TOKYO BAY

How did the water quality of Tokyo Bay change over the past 100 years? One of the main factors in the deterioration of the water quality of Tokyo Bay is its conditions of eutrophication. The level of eutrophication in a semi-enclosed bay is determined by the balance between the volume of sea water exchange and the nutrient load. Organic matter accumulates in a bay when the nutrient load is greater than the volume of sea water exchange. The accumulated organic matter causes eutrophication in the bay. Therefore, this paper will firstly examine the volume of water exchange and the nutrient load in Tokyo Bay. Next, the changes in the typical phenomena of eutrophication and sediment conditions will be addressed. Moreover, the change in fish haul in Tokyo Bay will be examined as an index of ecological changes in the bay due to changes in water quality.

3. SEA WATER EXCHANGE

The present residence time of sea water in the inner Tokyo Bay has decreased in comparison with that in the past. The residence time of sea water for the inner bay in 2002 was 0.5 month during summer and 1.5 months during winter (Takao et al., 2004). In contrast, the residence time of sea water in the 1960s was 1 month during summer and 3 months during winter (Unoki and Kishino, 1977). Numerical simulations have indicated that the reduction in the residence time of sea water was caused mainly by an increase of fresh water input due to human population importing water from other catchments and the decrease in the surface area of the bay. The mechanisms of the reduction in the residence time of sea water are the following: (1) Estuary circulation due to fresh water input is the main cause of sea water exchange in Tokyo Bay (Unoki, 1998). As a result, the increase in fresh water input promotes estuary circulation and enhances sea water exchange. (2) The decrease in water surface area has caused a reduction in the tidal speeds. The reduction in tidal currents weakens vertical mixing and strengthens stratification in Tokyo Bay. Consequently, estuary circulation is promoted, and sea water exchange is enhanced.

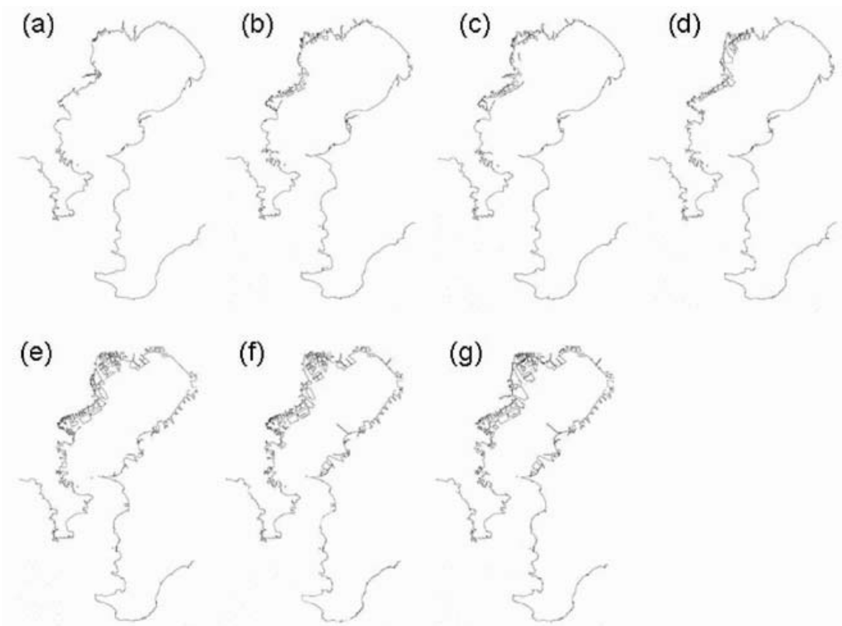

Figure 3. *Changes in Tokyo Bay coastline. (a) 1920, (b) 1940, (c) 1958, (d) 1969, (e) 1976, (f) 1990, and (g) 2002.*

The fresh water input into Tokyo Bay increased by approximately 50 $m^3\ s^{-1}$ from 1960 to 1970: it was approximately 350 $m^3\ s^{-1}$ before 1960 and 400 $m^3\ s^{-1}$ after 1970. The reason for the increase in fresh water input was the increase in the imported water mass from neighboring catchments during the period from 1960 to 1970. The imported water mass was used as city water and industrial water in the catchment area of Tokyo Bay. Reclamation in Tokyo Bay drastically decreased the surface area of the bay from 1960 to 2000, to 80% of the former area (i.e. 1960). The decrease in surface area caused an 11% decrease in the tidal range of M_2 (Unoki and Konishi, 1999). In addition, the tidal current at the mouth of the outer bay decreased by 20% from 1968 to 1983 (Yanagi and Onishi, 1999).

4. NUTRIENT LOAD

The nutrient load (represented by the parameter Chemical Oxygen Demand or COD) in Tokyo Bay reached its peak in the 1980's but decreased markedly thereafter (Figure 4). The nutrient load was 100×10^3 kg d^{-1} until 1940. It increased in proportion to the economic development of Japan after 1940 and reached 580×10^3 kg d^{-1} in the 1980s. After that, it decreased to 320×10^3 kg d^{-1} in the 2000s. The reasons for the decrease in nutrient load from the 1980s to the 2000s are the following: (1) The nutrient load from domestic waste decreased because of increased sewage treatment, (2) The nutrient load from factories decreased because of the obligation to treat liquid waste.

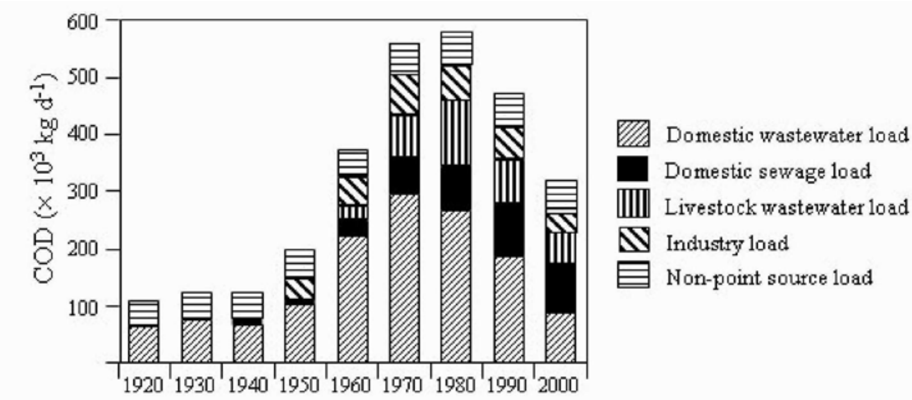

Figure 4. *Changes in nutrient load into Tokyo Bay.*

4.1. Eutrophication

Both the residence time of sea water and the nutrient load into the bay have decreased, yet the condition of eutrophication remains. The COD value currently remains at 3-5 mg l^{-1}, while it was about 6 mg l^{-1} in 1960s. In addition, the typical phenomena of eutrophication such as "red tide," "anoxic water," and "blue tide (aoshio)," still occur in Tokyo Bay. The sediment condition is sludge. These conditions will be described in greater detail.

4.2. Red tide

The red tide condition is a phenomenon of phytoplankton bloom. Its definition depends on the species of phytoplankton. Generally, the criteria for a red tide condition in Tokyo Bay are (1) less than 1.5 m of transparency; (2) at least 10^3 cells ml^{-1} of phytoplankton density for large species, or 10^4 cells ml^{-1} for small species; and (3) at least 50 µg m^{-1} of chlorophyll *a* (Nomura, 1998).

The red tide condition was an uncommon phenomenon between the 1900 and 1910 in Tokyo Bay (Okamura, 1907; Asakura, 1907), but when present, the main species of phytoplankton were flagellates such as the genera *Gynodinim*, *Pouchetia*, and *Peridinium* (Nomura, 1998). There are little data available regarding the red tide condition occurrence and constituent species prior to 1950. Nomura (1998) gave a historical review of red tide phytoplankton species and occurrences in Tokyo Bay. Beginning in the 1950's, the species of phytoplankton present in the red tide condition diversified. Red tide conditions with diatoms such as *S. costatum* began to occur (Nomura, 1998). After the 1970's, red tide conditions with *S. costatum* appeared throughout the year (Furota, 1980). The red tide condition with *S. costatum* became frequent after the 1980's. Very highly concentrated blooms, however, were not composed of diatom species but flagellates, such as *H. akashio* and the genus *Prorocetrum* (Nomura, 1998).

The frequency of the red tide condition in Tokyo Bay was less than 5 days per year before the 1940's. It gradually increased from the 1950's to the 1980's, reaching 20 days per year during the 1980's. The frequency has remained at 15-20 days per year since then.

4.3. Anoxic water

The frequency of the anoxic water condition in the bottom layer of Tokyo Bay has almost remained constant at 3-4 months per year since the 1980's. The area where anoxic water exists usually extends over 50% of the inner bay in summer.

4.4. Blue tide (aoshio)

The anoxic water condition in the bottom layer enhances the release of sulfide. When the anoxic water with sulfide upwells to the surface layer, the oxygen in the surface layer oxidizes the sulfide to sulfur. Then, the color of the surface layer changes to blue-white. This is called a "blue tide (aoshio)". In addition, the sulfide is harmful to marine organisms. In Tokyo Bay, the blue tide condition occurs in the inner bay when strong north winds continue for several days during summer and autumn. The blue tide condition inflicts significant damage on benthos such as shellfish that cannot escape from the blue tide water.

The blue tide condition in Tokyo bay was first observed in the 1950s. The frequency of the blue tide condition reached its peak in the 1980's, at about 6 times per year. It has since decreased to a current level of about 4 times per year, each event persisting on average for 2-3 days duration.

5. BOTTOM SEDIMENT

The seafloor of the shoreward area in Tokyo Bay was covered by sand in the 1950's. The present seafloor of this region however, is covered by sludge with a moisture content (= weight of water / weight of sediment × 100) of over 200 (Figure 5). The reasons for the change in sediment conditions are the reclamation of sand areas and the accumulation of organic matter. Figure 5b shows the distribution of water content in Tokyo bay. Our field observations of 75 points in the bay indicated that a moisture content of over 200 corresponds to at least 8% ignition loss, 20 g cm^{-3} COD, 0.5 mg S g^{-1} of sulfate, 0.6 mg g^{-1} of total phosphorus, 15 mg g^{-1} of total organic carbon, and 1.5 mg g^{-1} of total nitrate, 0.5-1.3 g m^{-2} d^{-1} of sediment oxygen uptake rates, along with less than 0.001 mm of sediment grain diameter.

6. FISH CATCH

The total annual catch of fish was more than 1×10^5 tons in the 1950's. About 90% of the catch was shellfish. The catch started to decrease gradually after 1960, reaching less than 5×10^4 tons per year in the 1970's. The main factor in the overall decrease was the decrease in the shellfish catch. Since then, the catch has continued to decrease. In the 1990's, the total annual catch of fish was 4×10^4 tons per year, with a shellfish catch of 3×10^4 tons per year.

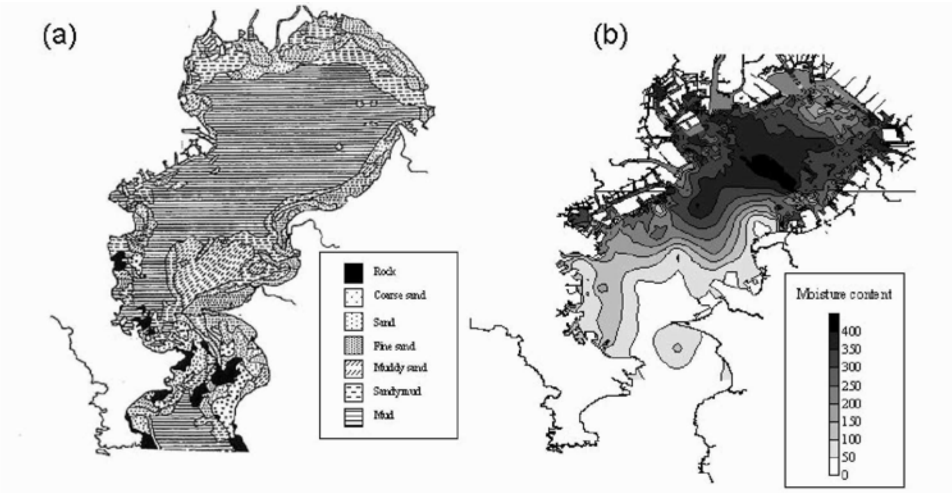

Figure 5. *Distribution of sediment in Tokyo Bay. (a) 1960's (Kaizuka, 1963), (b) 2000's.*

The factors causing the decrease in the shellfish catch are the following: (1) the decrease of habitat in the areas of tidal flats and shallow water regions because of reclamation; (2) the degradation of shellfish habitat environment due to anoxic water, red tide, and blue tide; (3) the competition with newly introduced species; (4) the disease; and (5) the destruction of the tidal flats network for benthic larvae transport.

The area of tidal flats decreased about 90% over the last 100 years (Ogura, 1993). A internal zone biomass of 12.6×10^4 tons was lost as 126 km^2 of tidal flats were reclaimed (Furota, 1994). Furthermore, Kakino (1986) suggested that 3×10^4 tons of bivalve biomass were killed by the blue tide condition in September 1985.

The decrease in benthos biomass has caused a decrease in the purification capability of Tokyo Bay (Furota, 1994). This is considered to be the main reason why the frequencies of red tide and anoxic water conditions in Tokyo Bay have not decreased, even though the nutrient load has decreased.

7. FUTURE PLAN FOR ENVIRONMENTAL RESTORATION OF TOKYO BAY

As shown in above section, the environmental health of Tokyo Bay is susceptible to degradation because a large volume of water is entrapped in the enclosed area, the poor water exchange rate resulted in a long residence timescale, a large amount of nutrients is discharged, and bio-chemical activities makes further exacerbate the stresses. In this section, some restoration efforts are listed by describing the legislative condition, the history of environmental rehabilitation projects, and the Tokyo Bay Restoration Plan. It will highlight the possible future activities in Tokyo Bay for environmental restoration for the “Wise Use (RAMSAR, 2004)”.

7.1. Legislation on environment restoration in Japanese coastal line

There is now a strong public desire to conserve and restore the natural environment of the coastline and a trend towards public participation for managing those activities by coastline communities. National policy initiatives in this regard include the enactment of the Environment Basic Law (1993). The Basic Environment Law states our responsibility for future generations, stipulating the framework for environmental policies which ensure the benefit of a sound and rich natural environment for both a social structural development that would reduce as much as possible the load of socio-economic activities on the environment. Also of paramount importance is the need for international collaboration to achieve these goals. The focus is also directed towards the need for economic measures, the promotion of recycling programs, the designation of a public national holiday (Environment Day), the support for environmental education, and the importance of cooperation with private organizations. Based on this Law, the Basic Environment Plan was introduced in 1994 followed by revised in 2000 (Table 2).

Japanese coastlines are managed by several different organizations. After the enactment of the Environment Basic Law, the revision of the River Law (1997), the Coast Law (1999), the Port and Harbour Law (2000), and the Fisheries Basic Law (2001) were enacted. All these revised laws are clearly expressing a concern for environmental protection and restoration. For example, the Port and Harbour Law clarified the governmental commitment to environmental conservation in the port and harbour administration, stipulating that environmental considerations must be incorporated in port and harbour developments. The basic policies were modified to add descriptions about the conservation, the restoration, and the creation of favourable port environments as well as the interaction between human activities and natural ecosystem, which are referred to as "basic matters to consider regarding environmental conservation in the development, utilization and improvement of ports and harbours, along with the development of channel development". An another example, the Coast Law is adding coastal management measures from a comprehensive perspective. These include coastline protection, which is the fundamental objective of the old law; the improvement and conservation of coastal environments; and an appropriate use of coasts by the public. The Law was therefore designed to further promote comprehensive coastal conservation approaches with well-balanced views on disaster prevention, environment conservation and public use.

The goal to create a symbiosis between human activities and the natural ecosystem becomes a government goal. This has in turn led to a change in governmental approach towards nature conservation, in which ecological restoration and rehabilitation shall be actively promoted rather than just having its current status conservation. It may also include human assistance for the process of natural restoration. In order to make such a commitment to natural restoration to be handed down to future generations, there also is the need to establish procedures and a framework that can support the growing desire for commitment by the local residents who are increasingly aware of the environment.

Table 2. *Recent issues related to environmental restoration in Tokyo Bay port and harbour area.*

Year	**Environmental measures in ports and harbors**	**Other domestic measures in Japan**	**International issues**
1900s	Formulation of the policy "the creation a healthy and productive environment on water fronts"		
1970	Water Pollution Control Law was introduced		
1971			International cooperation for the conservation and wise use of wetlands was adopted in Ramsar
1972	Water Pollution Control Law was enacted		
1973		Law Concerning Special Measures for Conservation of the Environment of the Seto Inland Sea was enacted	
1975			International cooperation for the conservation and wise use of wetlands was come into force
1980			1st Meeting of the Conference of contracting parties (COP), Cagliari,
1987			3rd Meeting of the COP, Regina (Wise Use)
1988			"No net loss" declaration by President Bush, USA
1991			Earth Summit in Rio
1992			Adoption of Convention on Bio-diversity Habitat Directives is enacted, EU
1993		Conclusion of Convention on Biological Diversity Introduction of Basic Environment Law	

1994	Formulation of a new environmental port and harbor policy, "Eco-port" Eco-port model project started	Basic Environment Plan was authorized in the Cabinet	Effectuation of Convention on Bio-diversity
1995	Experiments on tidal frats started in Port and Harbor Research Institute	National Strategy on Bio-diversity of Japan	Establishment of International Coral Reef Initiative
1996	Formulation of the policy "Ports and harbors supporting mass transport and interaction"		Asia-Pacific Migratory Water bird Conservation Strategy
1997		Environmental Impact Assessment Law was enacted Revision of River Law	Adoption of Kyoto Protocol to the United Nations Framework Convention on Climate Change
1998		"Grand Design for the 21st Century" was formulated	
1999		Revision of Coast Law	7th Meeting of the COP, San Jose
2000	Revision of Port and Harbor Law Modification to the basic policies regarding port and harbor development A new port and harbor policy for the 21st century "Minato Vision connecting our lives to the sea and the world"	Six laws concerning water management and recycling was formulated	Adoption of Cartagena Protocol on Biosafety to the Convention of Bio-diversity Conference of contracting parties to the Convent on Bio-diversity Natura 2000 is enacted, EU
2001	Ministerial reformation Creation of the center of greenery along coastal areas	Prime Minister Koizumi's policy speech "Creation of a society with harmonious coexistence of nature and humans" Basic Environment Law was enacted	2nd Asia-Pacific migratory Water bird Conservation Strategy Millennium Ecosystem Assessment
2002	"Restoration of the sea" was included in urban restoration projects	National Bio-diversity Strategy of Japan was formulated Law Concerning Special measures for restoring Ariake and Yatsushiro Seas Law for the Promotion of Nature Restoration was introduced	Earth Summit in Johannesburg 8th Meeting of the COP, Valencia (ICZM)

The cabinet has advocated strategies in its documents "Creation of Cities and Land in a Coexisting Relationship with Nature (2001)", the "New Biodiversity

Strategy (2002)" and the "Law for the Promotion of nature Restoration (2003). Furthermore, the Urban Restoration Office of the cabinet enacted the tertiary decisions including the Restoration of the Sea under the Urban Restoration Project. This was followed up by the creation of Tokyo Bay Restoration Committee that includes central government agencies and local governments. The action plan to restore Tokyo Bay was enacted (2003). It gives specific goal of restoring Tokyo Bay to the municipal and public organizations by working together.

7.2. Restoration projects

Specific restoration project by the port and harbour bureau will be introduced as an example of history of environmental restoration. In this section "Restoration" has been taken as the collective expression for creating, altering, or improving wetland environments. This term therefore encompasses a number of terms used in previous works (PIANC, 2003):

- **Creation:** The conversion of a persistent non-wetland area into wetland through some activity of man.
- **Enhancement:** The alteration of existing wetlands to provide conditions that did not previously exist and that increase one or more user-defined values.
- **Reclamation:** The conversion of a water area or wetland area into a more terrestrial-based system or into a wetland above mean water level through some activity of man.
- **Regeneration:** Natural regrowth after disturbance.
- **Rehabilitation:** Human activity aiming at repairing damaged or blocked ecosystem functions.
- **Remediation:** The cleaning up of a polluted wetland site.

Efforts to conserve marine environment were represented by the Marine Environmental Improvement Project to clean-up floating garbage and oil (1974-), the "Sea-Blue" Project to improve seawater quality (1988-), and the "Eco-Port" project proposal (1994-).

7.3. 1970s: the marine environmental improvement project

Since 1960, Japan experienced rapid economic growth of economics. Unfortunately this growth was accompanied by the quasi-unregulated discharge of hazardous material (heavy metal and oil etc.) discharge in to bay. In 1974, the Water Pollution Control Law was enacted. The ministry of Environment has started to monitor water quality in public water (lakes, rivers and the sea).

The Marine Environmental Improvement Project was started under this initiative. The port and harbour bureau equipped Marine Environmental Improvement Vessels to collect floating garbage on a daily operational basis (Figure 6, Animation 1). At the same time, the vessel can be used for oil recovery when an oil spill occurs. It is a symptomatic measure for mitigating the degradation of the environment. Results are instantaneous and visible, nevertheless continual operation is required.

To support these activities, studies were undertaken of the water exchange mechanisms using hydraulic models. Large scale model tests using tanks, were conducted (Kaneko et.al., 1973) to estimate the transport and diffusion of materials in the bay. These demonstrated a close relation between bathymetric changes due to port construction and the flow and diffusion patterns.

In addition to hydraulic model studies, a numerical model was also developed. In the first stage of the development, the modelers attempted to calculate the steady-

Figure 6. *The marine environmental improvement vessel to collect floating garbage in daily operational basis in Tokyo Bay.*

Animation 1. *Equipped marine environmental improvement vessel to collect floating garbage.*

state, tidal residual currents in the bay (Kaneko et al., 1975). It took 5 hours to run a 4 tidal cycle model using a 2 km mesh model for Tokyo Bay. Thus, to speed up the calculations, ADI (Alternative Direction Implicit) scheme were well studied.

Nevertheless, the relationship between the flow field and the changes in bathymetry could not be checked by numerical models. To resolve this problem, there was the need for a combined approach using numerical and hydraulic modeling to study the steady state situation of current and material transport.

7.4. The "Sea-Blue" project

In 1980s, the hazardous material contamination problem seems to be under controlled. Nevertheless, the problems due to eutrophication, red tide (algal broom), odor problem, and oxygen depleted water caused by organic pollutants still remained. The "Sea-Blue" project was designed to tackle with this problem Study (Committee of Sea-Blue Technology, 1989). The "Sea-Blue" is a Japanese word expressing blue sea environment. The concepts of the "Sea-Blue" project were;

- Active good environment creation
- Close relation between land and sea
- Environment improvement in concern with use
- Enhance potential of self purification ability of sea
- Combination of techniques.

It aimed to enable water purification by putting together "Sea-Blue techniques" which were process oriented technologies to improve water quality and reduce environmental degradation. It is aiming to reduce source of environmental degradation. Nevertheless, the indices of environmental health were not directory related to ecosystem, but related to physical and chemical status such as flow and water quality. Furthermore, the plan did not take into account ecosystem sustainability. Thus, each measure must quantify its ability to enhance the water purification potential and its duration. The model used for such tests including considered nutrient cycling processes.

One example of such modeling is for "sand capping" techniques (Figure 7). A clean sand layer placed on top of the contaminated materials (capping) decreases completely or partially the re-entrainment of nutrients in the water column. Furthermore, the introduced sand layer gives a new habitat for organisms. Nevertheless, the new sand layer can become contaminated by detritus in the water column. Material transport in sand layer also diminishes the effect of capping.

A phosphorous cycling model was established to test the sand capping techniques (Horie and Hosokawa, 1985). The detail nutrient exchange process between the sand layer and the water column were modeled. A long term prediction using the model showed that the re-entrainment was suppressed by half for 20 years. Such studies gave a technical justification for implementing the plans in the field.
At this stage, a variety of models were developed for tackling specified problems those focused on specific areas and specific micro-processes.

7.5. 1990s: the "Eco Port" project

In the 1990s, the port and harbour bureau released the "Eco Port" plan. It extended the "Sea-Blue" project not only for water quality improvements, but also for the creation of ecosystem such as tidal flats and sea grass meadows. The goals of the projects are:

- Maintain or improve the water circulation and material cycling

- Take into account dominant processes of contamination
- Take into account the macro balance of material cycling
- DO (Dissolved oxygen) is important indicator of water quality
- Monitor the sediment quality and its change
- Monitor indicator species

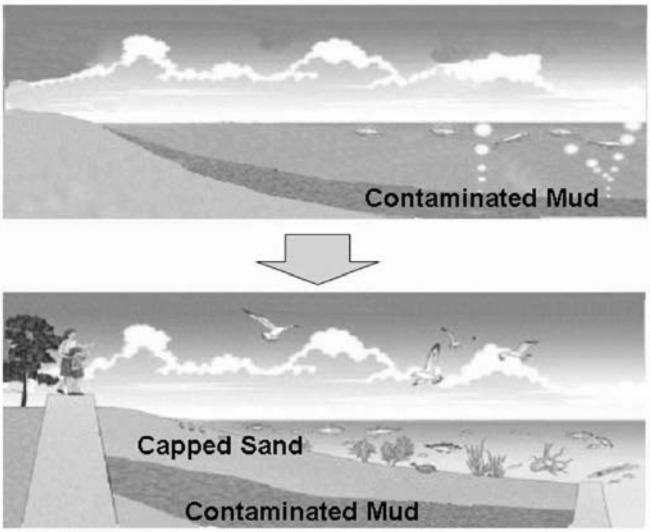

Figure 7. *Sand capping techniques in the "Sea-Blue" technologies.*

- Care for the monotonous ecosystem
- Take into account the shallow water, tidal flat and sea grass meadow ecosystems.

The difference between "Sea-Blue" and "Eco-Port" project is as follows. The "Sea-Blue" project provides a list of techniques, and it lets a planner choose. However, in the "Eco-Port" project, only the environmental concept is provided; a list of techniques is not provided. It not gives a list of techniques solely. This forces the planner to find the best way to do that, and to provide an analysis for this. On this point, the bureau has thus started to address the issues of ecosystem rehabilitation and restoration.

To support the project, the experimental facility shown in Figure 8 has been used since 1995. It consists of three water pools, each with an area of 20 m^2. Bottom mud was taken from the Banzu Tidal Flat in Tokyo Bay, thoroughly dried and vigorously mixed, then spread to a depth of 50 cm in the pools. Benthos was not observed in the accumulated dried mud. Fresh sea water was taken from Kurihama Bay and placed in the pools without any treatment. The water level was controlled to create tidal cycles in each pool. Biological succession and communities development were carefully monitored. It shows that the larger the body size (> 10^1-10^3 mm), the lower the population density (< 10^{-1}-10^9ind cm^{-2}). These ecotone experiments reproduced well the field observations on the Banzu tidal flat (Kuwae and Hosokawa, 2000; Kuwae et.al., 2004).

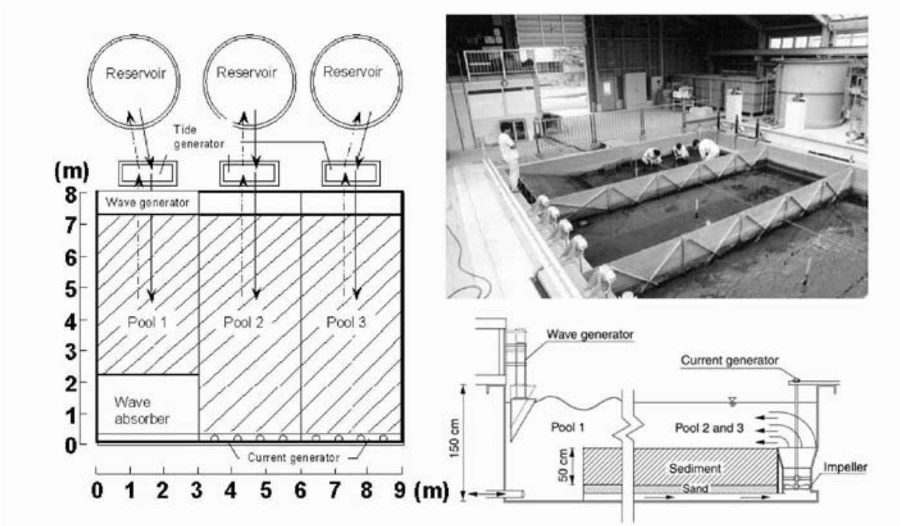

Figure 8. *The Intertidal Flat Experiment Facility (IFEF) at the Port and Airport Research Institute.*

Physical-biological interactions have been also studied in field. For example, the cluster analysis of macrofauna and sediment quality of a natural sandy tidal flat shows a strong relation between sediment quality and benthic community development (Furukawa et.al., 1999). Not only sediment quality, but also physical forces such as current and wave height distribution contribute to biological zonation in this area. These research findings support the idea of controlling ecosystems by controlling physical external forces and sediment quality by civil engineering measures.

8. TOKYO BAY RESTORATION PLAN

On 26 March 2003, the "Tokyo Bay Restoration Plan" was endorsed by the Council for Promotion of Tokyo Bay Restoration, which is formed of 11 central government bodies and 7 regional government bodies (4 prefectures and 3 cities). The plan consists of five parts: (1) overview of Tokyo Bay restoration planning, (2) present status of Tokyo Bay environment, (3) specific goal of Tokyo Bay restoration, (4) promotion of measures to achieve the goal, and (5) related activities.

8.1. Overview of Tokyo Bay restoration planning

The Tokyo Bay Restoration Planning initiative was initiated by a decision of the Japanese Cabinet in December 2001. Seven prefectures and cities surrounding the bay and related central government ministries formed a council to promote the restoration of Tokyo Bay.

The goal is the "Tokyo Bay restoration through water quality improvement". This goal is to be achieved by collaboration among the related bodies within ten years.

8.2. Specific goal of Tokyo Bay restoration

"Creation of a bay for the Tokyo metropolitan area that has amusement-based places, rich habitats with high biodiversity, and beautiful seashore areas" is the stated goal of Tokyo Bay restoration.

This goal does not have quantifiable target, such as "COD less than 5 mg L^{-1}". The goal is not prescriptive: it defines the condition required of a feature and not the actions or processes necessary to obtain or maintain that condition. The basic idea of the goal is that "it should be an expression of purpose".

The indicator for measuring achievements is the bottom layer concentration of dissolved oxygen (DO). The target level is in a "livable environment for benthos throughout the year". A low DO value is not a source material for an adverse condition, like nitrogen, phosphorus, and organic matter; rather, it is the result of an adverse condition. Thus, maintaining DO at a target level requires an integrated approach; simply controlling a specific effluent, for example, is not enough. The creation, restoration, and conservation of integrity of bay system within natural fluctuations are required.

The process for assessing the achievement of the goal is set as follows. Several priority implementation areas with specific targets have been set as points for monitoring (named as "appeal points"). Each point has a specific target image of restoration, as shown in Figure 9. It is thus possible to assess the project in terms of specific goals.

8.3. Promotion of measures to achieve the goal

The measures to achieve the goal are categorized into three areas: those that reduce the load imposed from land, those that promote environmental restoration offshore, and those that implement an environmental monitoring scheme.

The major actions for reducing the load from land are planning total load control, improving combined sewer systems, and suppressing runoff. These plans include wetland restoration to trap nutrients in run-off water, to control non-point source load, and to do a total cost control of the watershed. Not only does the plan list the tasks to be undertaken, but it also discusses the systems to be built.

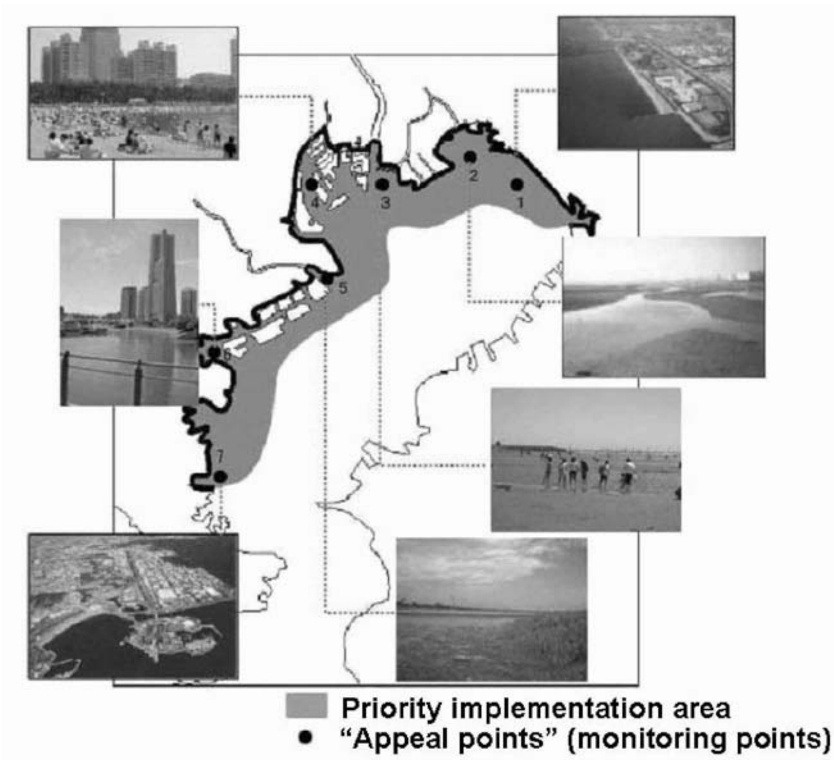

Figure 9. *Tokyo Bay Restoration Plan.*

The major action plans for promoting environmental restoration offshore are reducing the source of the internal load on the seawater and increasing the water purification capacity. These plans include dredging bottom sediment and replacing it with clean sand (sand capping), collecting floating garbage using specially designed vessels, and promoting clean-up campaigns with NGOs and the fishery industry. A network of ecosystems among habitats is an important concept in the restoration of wetlands in coastal areas.

A major action plan for implementing an environmental monitoring scheme in Tokyo Bay is the monitoring of bottom water DO levels. This includes the monitoring of the currents and water quality using moored facilities and vessels. Items to be considered include remote sensing, sharing of monitored data with stakeholders, provision of the results of the monitoring to the public, and public participation in the monitoring.

8.4. Related activities

Experimental actions are also designed to achieve the goal. Because of uncertainties about the natural environment condition, a step-by-step method (an adaptive approach) is needed to examine the techniques or systems to restore the environment. The environmental monitoring using high-frequency radar and knowledge exchange with other countries are examples of these activities.

9. PLANNING THE FUTURE REGIME

People living along a coastline are expected to use it wisely, and the wishes of the local people should be respected. Their support is indispensable for the many measures that require regular observation and long-term maintenance. Ways to determine the wishes of the local people, and ways to carry this out thorough management, must be established. Multi-disciplinary, scientific symposia for gathering experts in modeling, ecosystem assessment, observational techniques and data processing, and other specialized topics are needed to facilitate the sharing of knowledge. Systems for public participation in strategic planning are also needed. The plans should encourage continual improvements in observation methods, accumulation of observation results, development of reliable, user-friendly ecosystem models, analysis of environmental -restoration technologies, and maintaining the cooperation between scientists, local governments, and coastal residents.

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CHAPTER 3

ECOLOGICAL NETWORK LINKED BY THE PLANKTONIC LARVAE OF THE CLAM *RUDITAPES PHILIPPINARUM* IN TOKYO BAY

HIROFUMI HINATA AND KEITA FURUKAWA

1. INTRODUCTION

The National Institute for Land and Infrastructure Management (NILIM) has undertaken the task of identifying where and how to restore tidal flats as an important coastal ecosystem in estuaries and coastal seas. It initiated the "Asari Project" to determine the existence of an ecological network in Tokyo Bay.

Asari are short-necked clams that inhabit estuarine sandy tidal flats and shallow water regions (such as Haneda shallow, Sanmaizu, Sanbanse, Banzu tidal flats, and Futtsu shallow; Figure 1). They are an important fishery product in Japan. The average annual catch of Asari was 1.5×10^5 t in the 1980s; this declined to 0.75×10^5 t in the 1990s. Asari develop into a trochophore and then a veliger larva with a shell after hatching. Veliger larvae are classified into D-shaped larvae with a straight hinge line (typically 100 μm in size), umbo larvae, or fully grown larvae (typically 170-180 μm in size) according to the stage of development. Since Asari larvae have the planktonic stage: D-shaped larvae and umbo larvae, the lack of an ecological network can lead to a reduction in the catch of adult Asari. By identifying suitable places for restoring the network, we can identify the best places for restoring tidal flats.

Careful sampling was done to detect the pass of the network. Sixty-five sampling points were monitored at three-day intervals for three successive sampling times in summer and autumn. Asari larvae circulate over the entire bay, and there is a network of tidal flats along the west and east shores. The smallest larvae captured were only one or two days old, hence backtracking simulation could be used to identify their origin. The northwestern shore of the bay, and around Banzu and Futtsu tidal flats were identified as the source and as the answer to "where we should restore the tidal flats?" The northwestern shore of the bay overlap with the priority implementation areas of the Tokyo Bay restoration plan, hence this study supports on scientific grounds the implementation of the Tokyo Bay restoration plan.

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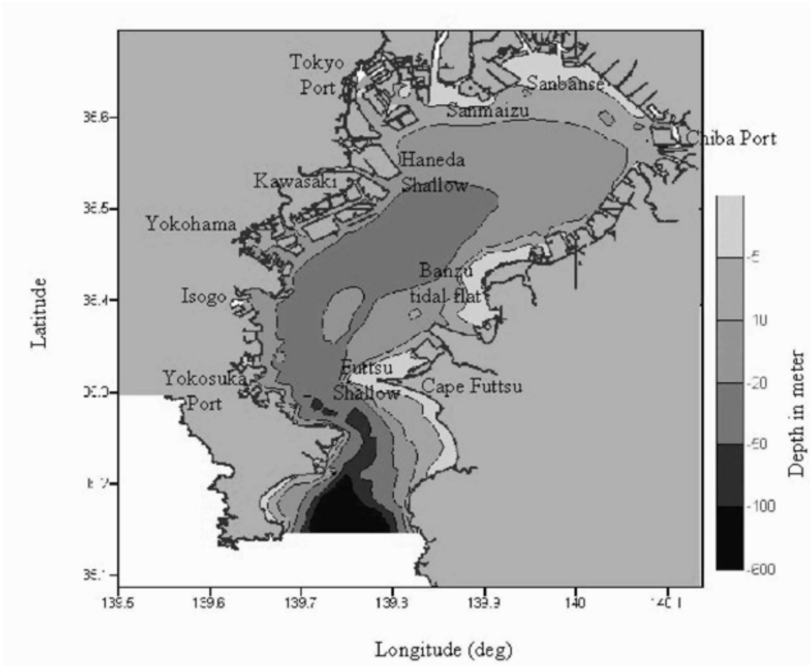


Figure 1. *Depth contour of Tokyo Bay, and possible Asari hatching areas in the Bay (Haneda shallow, Sanmaizu, Sanbanse, Banzu tidal flats, and Futtsu shallow).*

The "Wetland Restoration Project in Urban Coastal Zones" aims to integrate individual knowledge about habitat creation. It aims to answer the question, "how can we restore tidal flats?" The goal is to promote large-scale ecosystem experiments for establishing a firm scientific foundation for ecosystem restoration projects in urban coastal zones. Since April 2003, we have been making a preliminary study of habitat restoration using a tidal flat constructed in Osaka Bay. There is a pressing need to undertake activities that encourage collaboration between stakeholders, to establish optimal experiment plans, and to share knowledge and information.

2. THE SHELL LENGTH GROWTH RATES AND DISTRIBUTION OF PELAGIC LARVAE IN TOKYO BAY

Since it is necessary to understand advection process of the larvae in Tokyo Bay, which has been remained unknown, to construct artificial tidal flats in suitable areas or protect the clam’s larval supply areas in the bay, this study focuses on D-shaped and umbo larvae. Figure 2 shows the distributions of D-shaped larvae (shell length 90-130 μm) and umbo larvae (shell length 130-240 μm) in Tokyo Bay on the 2nd, 6th and 10th of August 2001. On August 2, numerous D-shaped larvae were found in the areas around Tokyo Port, the Banzu tidal flat and Cape Futtsu, while hardly any umbo larvae were found at all. Conversely, on August 6, hardly any D-shaped larvae were found, but numerous umbo larvae were in the central part of the bay. Then, on August 10, D-shaped larvae were found again in abundance around the bay mouth. Until now, nothing was known about the extent of pelagic (free-floating) short-necked clam (Ruditapes philippinarum) larvae in Tokyo Bay. However, the results of these observations have shown that these larvae spread out over the entire bay from their spawning grounds in the tidal flats and shallows.

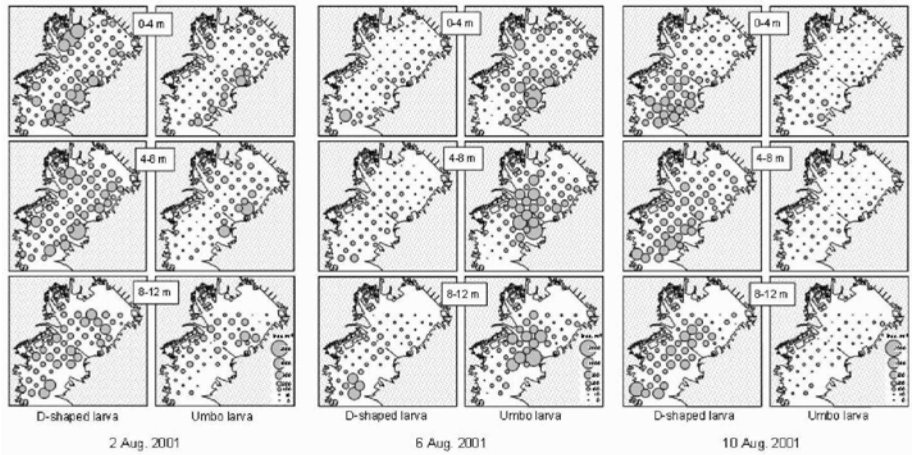


Figure 2. *Horizontal distributions of D-shaped larvae (shell length 90-130 μm) and umbo larvae (shell length 130-240 μm) at three depths in Tokyo Bay on 2, 6 and 10 August 2001 (after Kasuya et al., 2003).*

The pelagic larvae were dominated by different growth stages on different dates: the D-shaped stage on August 2 and 10, and the umbo stage on August 6. This suggests that the temporal variation in the density was affected more by a series of growth stages than by losses resulting from the flash out of the larvae to the outer sea, or from the death of populations due to a shortage of food. Accordingly, it is thought that the D-shaped larvae with a shell length of 100 μm, which were observed in the greatest numbers on August 2, had grown ,by August 6, into the umbo larvae with a shell length of 170-180 μm that were the most dominant on that date (Figure 3). Based on changes in the appearance density of these populations, which are thought to be one and the same, the rate of decrease of the population density over the 4-day period is estimated to be about 13%. Also, from the change of shell length, it is clear that the shell length of these larvae grows at a rate of 15-18 μm d^{-1} in Tokyo Bay during the summer. This growth rate is much higher than the value of 5.0-7.6 μm d^{-1} obtained in laboratory breeding experiments (Toba, 1992). Since this population was able to grow to a shell length of 210 μm or more by August 10, it appears that most of them would have progressed from pelagic to bottom-clinging behavior up to this point.

By working back from the shell length growth rates, it is possible to calculate when the larvae were spawned. In sea water held at 20 °C, short-necked clam eggs form D-shaped larvae with a shell length of 100 μm about 2 days after fertilization (Toba, 1987). Therefore, the larvae with a shell length of 100-120 μm that were

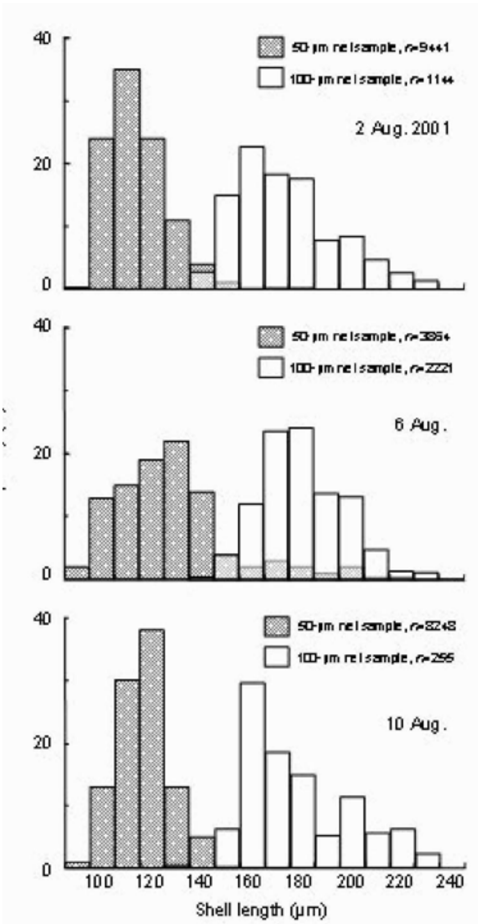


Figure 3. *Size frequency distributions of Ruditapes philippinarum larvae collected using a 50 μm and 100 μm nets on 2, 6 and 10 August 2001 (after Kasuya et al., 2003). n is the number of measured larvae.*

dominant on August 2 had existed as D-shaped larvae for the previous 0-2 days, and were fertilized 2-4 days earlier. It is thus estimated that these larvae had been released as eggs on or around July 30. It is thought that most of these larvae would have converted to a bottom-clinging lifestyle by August 10. Hence it appears that the pelagic stage of short-necked clam larvae in Tokyo Bay in summer lasts for approximately 10 days. Furthermore, it is estimated that the larvae with a shell length of 110-120 μm that were dominant on August 10 were released as eggs on or around August 6. For details of the sampling and method to identify the larvae, readers should be referred to the studies by Kasuya et al. (2003 and 2004).

3. A NUMERICAL CALCULATION OF THE ADVECTIVE PROCESS OF PELAGIC LARVAE

Next, numerical modeling was used to investigate the advective process of these pelagic short-necked clam larvae during the periods when clear results could not be obtained by the observation. Here, particular attention will be focused on the existence of networks between different areas of Tokyo Bay through the transport of these pelagic larvae.

3.1. Summary of the numerical model

3.1.1. Flow model

The Princeton Ocean Model (Blumberg and Mellor, 1987) was used as the current model in this study. The computational domain is shown in Figure 4.

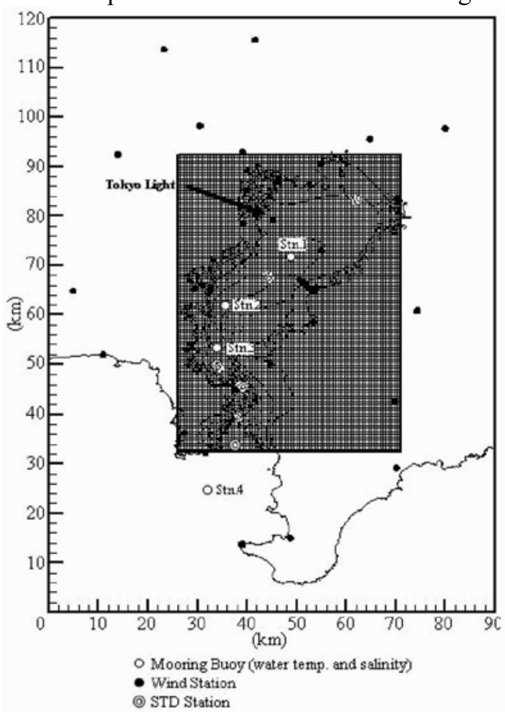


Figure 4. *Model domain and the numerical grid, and locations of mooring buoys, STD (salinity, temperature, depth measurements) and wind stations.*

The domain was divided horizontally into a grid of 600 m and vertically into 21 σ levels. The surface stress by the wind is one of the most important external forces in the bay. The calculations were performed for two different wind fields. One pattern consisted of a uniform wind over the entire computation region, based on the wind measured at the Tokyo light beacon off Tokyo Port (Japan Coast Guard; JCG), and the other was a wind field obtained by interpolating the wind velocities measured at 33 wind stations in and around Tokyo Bay operated by the Ministry of Land Infrastructure and Transport (MLIT), Japan Meteorological Agency (JMA), JCG, and Japan Highway Public Corporation (Figure 4). The wind velocities at 33 points were interpolated between the model grid points using a hyperbolic weight function (Yanagi and Igawa, 1992). Finally, the following formula was used to convert the resulting wind field into a wind stress field at the sea surface:

$$\tau = \rho C_D U^2 \tag{1}$$

where CD is a bulk transport coefficient which is given by a function of the wind speed as follows (Honda and Mitsuyasu, 1980):

$$C_D = \begin{cases} (1 - 1.89 \times 10^{-2} U) \times 1.28 \times 10^{-3} & (U \leq 8.0 \text{ m/s}) \\ (1 + 1.078 \times 10^{-1} U) \times 5.81 \times 10^{-4} & (U \geq 8.0 \text{ m/s}) \end{cases} \tag{2}$$

The vertical distributions of temperature and salinity at the open boundary were provided based on STD measurements by Chiba Prefecture (Figure 4). The sea level at the open boundary was also provided from the four main tidal constituents (K1, O1, S2 and M2) based on the harmonic constants at Yokosuka (JCG). This model also took into account the heat flux from the atmosphere and the river discharge from major rivers flowing into the head of the bay. The heat flux calculations were performed using weather data measured around Tokyo Bay by JMA. Also, the river discharge was based on data measured by MLIT.

3.1.2. Pelagic short-necked clam larvae model

In these computations, the pelagic short-necked clam larvae were assumed to behave as a passive tracer of currents. Although the observed spatial distribution of these larvae (Figure 2) was presumably affected by active behavior of the larvae, at the present little is known about the active behavior of pelagic short-necked clam larvae in real seas. There is a great need for research into the active behavior patterns of pelagic short-necked clam larvae, and it is hoped that data on this topic will become available in the future. This study also took into consideration the natural decrease of pelagic larvae resulting from predation by Aurelia (a jellyfish) and Noctilca (a zooplankton). For the sake of simplicity, the rate of predation by Aurelia and Noctilca was assumed to be proportional to the observed numbers of these predators, and the proportional coefficient was adjusted so that the model reproduced the measured distribution of the larvae. In the model, the tracers that approached within 3 km of the open boundary were assumed to be flushed out to the Pacific Ocean and therefore eliminated from the calculations.

4. RESULTS

4.1. Oceanographic model verification

Figure 5 shows the model- and HF radar-derived surface current fields. (For a description of the performance of HF radar, see Yanagi et al., 2000 and Hinata et al., 2005.) These are daily averaged velocities. Although not shown in this figure, when the wind speed observed at the Tokyo light beacon was applied to the entire computation region, it was not possible to reproduce the seaward flow observed off Sodegaura on August 1 or the clockwise circulation that appeared on August 9. This result demonstrates that it is important to consider the spatial distribution of wind velocities, even when performing computations on a scale as small as Tokyo Bay (20 km × 50 km). Although the temperature and salinity distributions are not shown here in detail, but the results obtained with the numerical model generally exhibited good reproducibility. According to a comparison with observation data obtained by mooring buoys situated at three locations in the bay (Figure 4), the root-mean-square errors of the temperature and salinity were about 0.43-0.82 (°C) and 0.28-0.43, respectively. However, the model predicted a slightly thinner surface mixed layer than was measured.

4.2. Networks between sea areas mediating the transport of pelagic larvae

After verifying the oceanographic model, we first attempted to numerically reproduce the distribution of umbo-stage larvae on August 6, using as a starting point the observed distribution of D-shaped larvae on August 2. As a result, we achieved a fairly good match between the calculated and observed results. We then numerically simulated the advective process of pelagic larvae up to August 9, by the date it is thought that most of the pelagic larvae spawned on or around July 30 would have converted to bottom-clinging behavior, and we investigated the existence of the ecological networks between various regions in the bay.

First, we studied the routes travelled between August 2 and August 9 by pelagic short-necked clam larvae that were assumed to have been spawned in the region around Tokyo Port in the north west part of the bay, in the Banzu tidal flats situated on the east cost in the central part of the bay (a typical habitat for short-necked clams), and in the sea around Cape Futtsu on the east coast near the bay mouth (Figure 6, Animation 1). These regions are where the most D-shaped larvae were found on August 2 (see Figure 2).

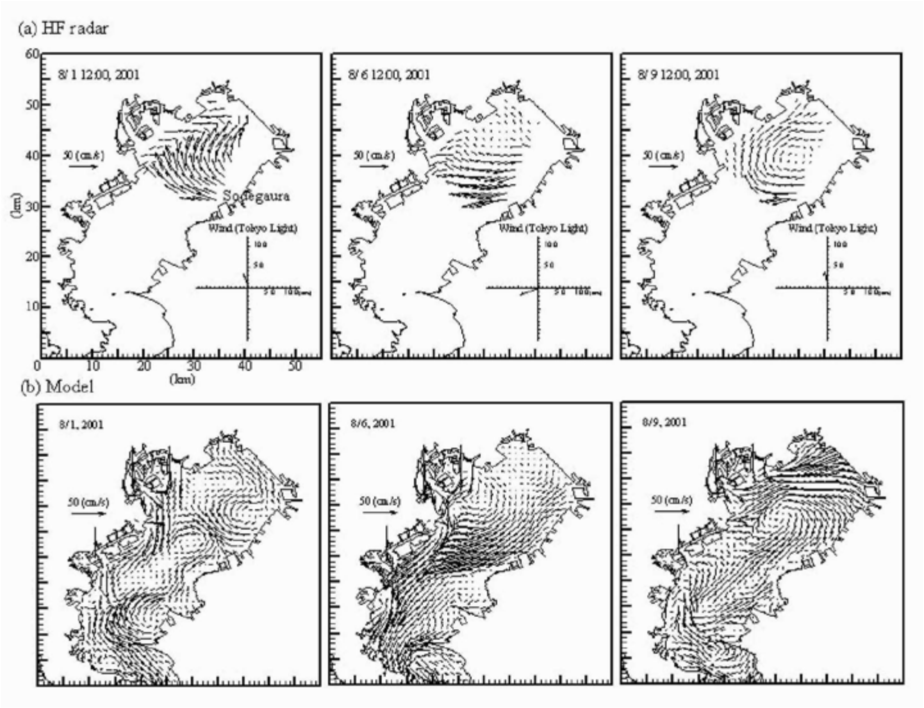


Figure 5. *Comparison of HF radar and model derived 25 hour running average surface currents at 1200 LST on 1 August 2001 (left panels), 6 August 2001 (middle panels), and 9 August 2001 (right panels), together with the 25 hour running average wind velocity vectors at Tokyo light (see location in Figure 4).*

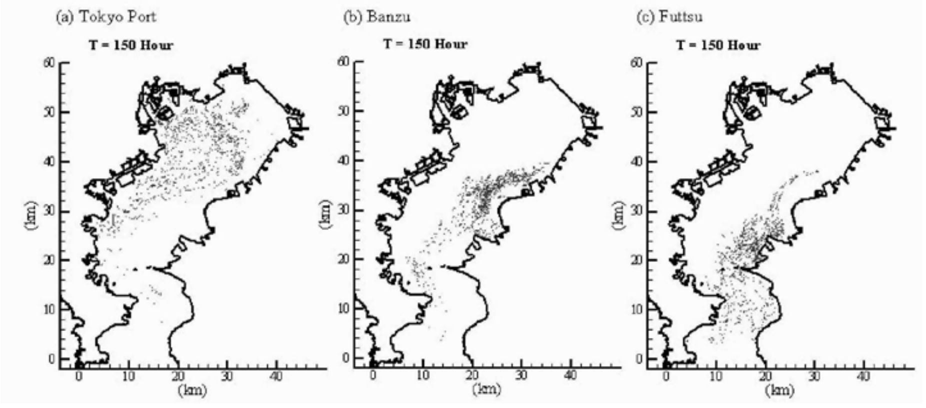
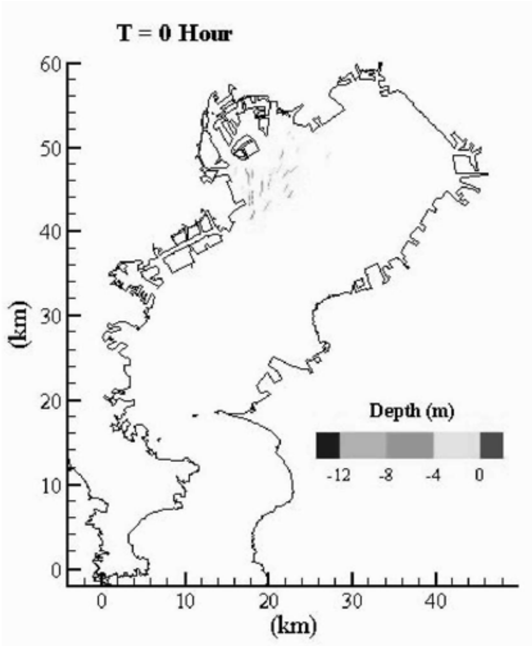


Figure 6. *Tracks of passive tracers released from (a) Tokyo Port , (b) off Banzu tidal flat, and (c) off Cape Futtsu for 150 hours.*



Aniamtion 1. *Tracks of passive tracers released from Tokyo Port (color of tracer changed by its depth).*

In the model, most of the pelagic larvae spawned near Tokyo Port were trapped by the clockwise circulation existing in the bay head (Fujiwara et al., 1997), and were thus transported east for a while before returning to the west coast of the bay during the 3rd and 4th of August. After that, some of the pelagic larvae were carried toward the bay mouth by estuarine circulation developing along the west coast. This circulation was driven by the major rivers flowing into the northeastern region of the bay.

At the same time, most of the pelagic larvae spawned around the Banzu tidal flats gradually moved towards the head of the bay, but the distance they moved was smaller than that on the west coast. Also, most of the pelagic larvae spawned around Cape Futtsu on the east cost near the bay mouth were carried to the region north of Cape Futtsu, and thus did not move as far as those on the west coast. However, it is likely that most of the pelagic larvae that were present in the surface layer around Cape Futtsu on August 2 flowed out from the bay. Thus, due to the weaker flows in the areas around Banzu and Cape Futtsu compared with the west coast of the bay (see Figure 5), most of the pelagic larvae spawned in the former areas tended to remain in the same area.

Figure 7 shows the relative quantities of pelagic larvae transported between each of the sea areas in the bay. As this figure shows, the greatest quantities correspond to larvae spawned near the Banzu tidal flats and Cape Futtsu (typical short-necked clam habitats) that returned to their spawning grounds. As mentioned above, this is due to the weaker flows in these areas, and to the fact that the spawning rates are higher because there are numerous adult short-necked clams living around the Banzu tidal flats and Cape Futtsu (e.g. Kasuya et al., 2003). The quantity of larvae transported between these two typical short-necked clam habitats was the second largest, but a more or less equal number of larvae moved away from and returned to the area around Tokyo Port. These numbers were also same as that of larvae transported from the region around Tokyo Port to the regions off Kawasaki and Yokohama to the south. Thus, on the west coast of the bay, the pelagic larvae spawned in the region around Tokyo Port are transported relatively longer distances towards the bay mouth by the estuarine circulation as mentioned above. On the other hand, in the regions around Banzu and Futtsu, a greater proportion either remains in their spawning areas. It is inferred that there was little in the way of transport between the west and east coasts of the bay at this time. We estimated that the

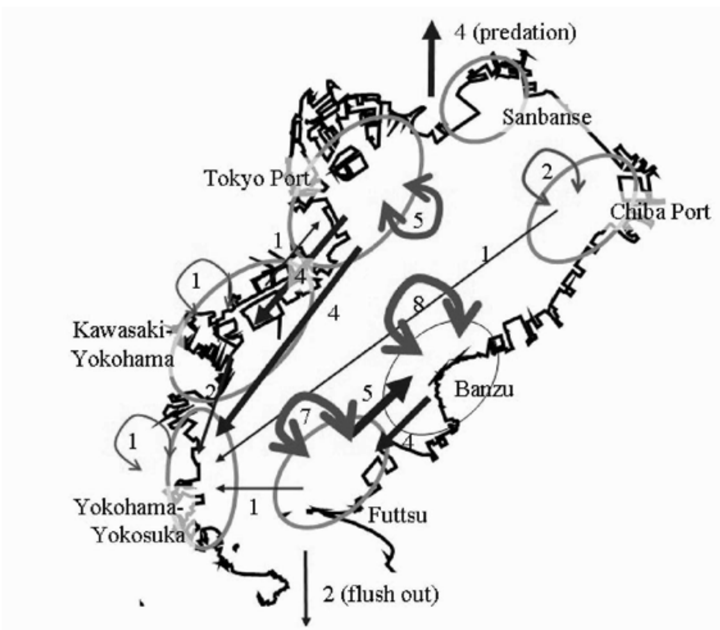


Figure 7. *Schematic diagram of the larvae network in Tokyo Bay. Numerals (non dimensional) indicate comparative quantity of larvae transport between various regions.*

quantities predated by Aurelia and Noctilca were the same as the quantities transported between Banzu and Futtsu, and that were almost twice as much as the quantity flowed out from the bay.

These results thus suggest that in Tokyo Bay pelagic short-necked clam larvae are transported throughout the entire bay from their spawning grounds in the tidal flats and shallows, and that the transportation patterns on the east and west coasts of the bay are different each other. This means that even if the adult short-necked clams in one habitat become extinct due to problems of water qualities such as upwelling of hypoxic water mass from the bay bottom, the recruitment of pelagic larvae from other areas should allow the short-necked clam population to recover in the devastated habitat. However, due to the characteristics of the flow field, it appears that some areas are more likely to act as sources of larvae than others. Also, it should be possible to preserve the short-necked clam resources of the whole bay by creating artificial tidal flat habitats for them in suitable areas in order to strengthen the networks of pelagic short-necked clam larvae over the whole of Tokyo Bay.

5. CONCLUSIONS

This study demonstrates the existence of an ecological network of short-necked clam larvae in Tokyo Bay. It also shows that the whole Tokyo Bay ecosystem may be improved if suitable spawning grounds, even if they are small, are conserved and restored. Thus, our strategy for zoning the primary implementation area in the Tokyo Bay Restoration Plan is justified.

It should be noted that this paper has only introduced the fate of pelagic larvae in one spawning case. Further observations and numerical studies will be needed to clarify how the advection of pelagic larvae contributes to the reproduction of short-necked clams. It will also be necessary to study how pelagic short-necked clam larvae respond to factors such as water temperature and salinity, the temporal variation of their specific gravity, and to add these results in the pelagic short-necked clam larvae model. Beside this uncertainty, inherent in any ecosystem model, this study demonstrates the importance of an ecological network in Tokyo Bay.

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CHAPTER 4

CIRCULATION PROCESSES IN TOKYO BAY

KEISUKE NAKAYAMA

1. INTRODUCTION

1.1. Modulation of the estuarine circulation

The exchange of sweater between Tokyo Bay and the Pacific Ocean significantly controls the water quality in the Bay (Figure 1). Unoki (1998) demonstrated that the estuarine circulation plays a dominant role in the sea-water exchange. The strength of the estuarine circulation varies according to the strength of stratification due to solar radiation and the discharge from rivers. For example, the estuarine circulation becomes weaker from summer to winter, which in turn suppresses the sea-water exchange, because solar radiation is weaker and the river discharge is smaller during the winter. The sea-water exchange is strongest during the summer.

Takao (2004) demonstrated that the residence time varied in 2002 from 40 days in winter to 20 days in summer. In contrast to this seasonal variation of estuarine circulation, Hibino et al. (1999), Nakayama et al. (2000), Shimizu et al. (2001), and Yanagi et al. (2003) reported that vertical mixing around the bay mouth, which is modulated fortnightly by the spring-neap tide cycle, may at times suppress the sea-water exchange in Tokyo Bay. Since the residence-time variation due to this fortnightly modulation can exceed 10 days - that is, more than half the residence time during summer - this mechanism needs to be clarified from the viewpoint of maintaining Tokyo Bay's water quality and ecological system.

1.2. Previous studies

The fortnightly modulation of the estuarine circulation has been investigated at several locations. For example, the vertical mixing and the internal waves due to the presence of a sill in a fjord have been observed to cause a fortnightly variation in exchange rates (Geyer and Cannon, 1982). The fortnightly change in exchange rates has been found to be a function of the turbulence due to tides; that is, the turbulence was a function of tidal amplitude. Around the mouth of the Hudson River, the structure of a coastal front was found to vary with the spring-neap cycle due to changes in the vertical structure of mixing as indicated by the Richardson number

E. Wolanski (ed.), The Environment in Asia Pacific Harbours, 47–65.

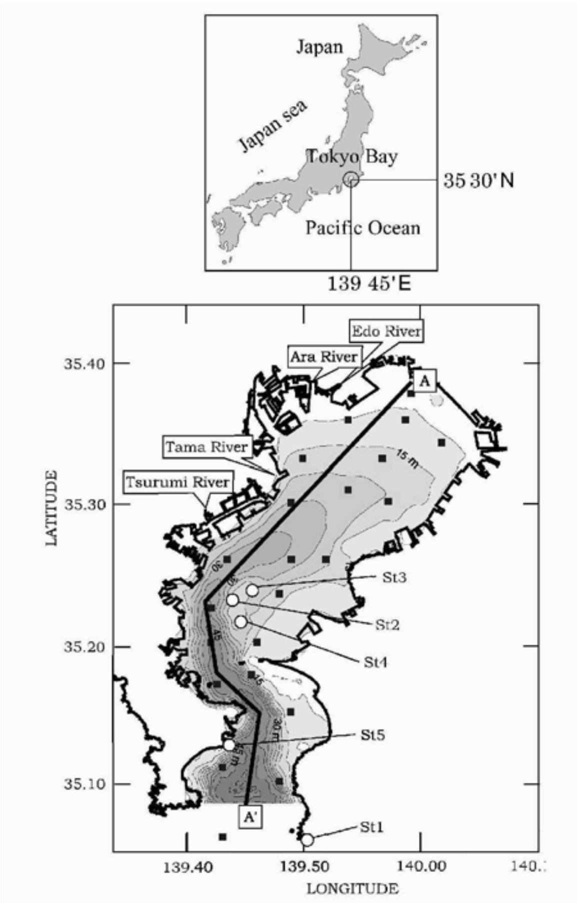


Figure 1. *Bathymetry of Tokyo Bay. Squares indicate the location of spatial observations and circles denote the location of stationary observations (St. 1-5). The main rivers flowing into the bay are the Edo, Ara, Tama, and Tsurumi rivers.*

and eddy viscosity (Peters, 1997). Hibiya and LeBlond (1993) also found that fortnightly modulations in salinity resulted from turbulence generated by tidal flow over a sill.

In addition to the studies concerning the relationship between turbulence and fortnightly modulations, Stigebrandt (1976) demonstrated the relationship between internal waves and fortnightly modulations, whereby the variations resulted from the breaking of internal waves. In particular, deep-water did flow over a sill when the internal Froude number was greater than 1. In contrast, when the internal Froude number was less than one, only upper water was observed over the sill. Griffin and LeBlond (1990) applied an internal Froude number analysis to the Juan de Fuca Strait and the Strait of Georgia. They demonstrated that an internal hydraulic jump related to the spring-neap cycle controlled the renewal of deep water. The study suggested that when the internal Froude number was greater than one, mixing was enhanced and deep-water renewal was suppressed.

In many studies, computational analysis has been used to investigate the mechanism responsible for the deep-water renewal produced by the spring-neap cycle. Nunes and Simpson (1994) used various turbulent closure model schemes to demonstrate that mixing was enhanced during the spring tide, and that estuarine circulation due to baroclinic energy was greater during neap tides. To clarify the motion of coastal fronts, Noh and Fernando (1993) used a two-dimensional non-hydrostatic numerical model to investigate the effect of turbulence on the density-driven currents responsible for the low-frequency oscillation in salinity.

As there are no sills in Tokyo Bay, the above results may not apply to the fortnightly modulation of its estuarine circulation. Studies related to the spring-neap cycle modulation in estuarine circulation in Tokyo Bay were undertaken in December, 1998 (Hibino et al., 1999; Nakayama et al., 2000; Shimizu et al., 2001; Yanagi et al., 2003). Hibino et al. (1999) provided evidence from spatial density measurements that the estuarine circulation was maximum during neap tides. Upper-water withdrawal and deep-water intrusion were also found to occur during the neap tide; this was based on Acoustic Doppler current meter (ADP; Alec Electronics) data obtained around the mouth of Tokyo Bay where the strength of estuarine circulation could be evaluated from the net velocity after the tidal components were eliminated. Nakayama et al. (2000) provided additional evidence from a salinity time series that suggested a fortnightly modulation in estuarine circulation; they found that the surface salinity decreased and the near-bottom salinity increased from the spring tide to the neap tide, which confirmed the result obtained by Hibino et al. (1999). Nakayama et al. (2000) concluded from ADCP data that the fortnightly modulation in estuarine circulation resulted from a change in the kinetic energy difference between the upper and lower layers due to the tide, but they offered no theoretical explanation for this.

Shimizu et al. (2001) and Yanagi et al. (2003) discussed the fortnightly modulation in residual flow from observations obtained by Hibino et al. (1999) and HF radar data obtained from the Port and Harbour Research Institute. Yanagi et al. (2003) concluded from numerical results obtained using the Princeton Ocean Model that the enhanced residual flow at neap tide was due to the decrease of the vertical eddy viscosity and to the estuarine front expansion. They also did not offer a theoretical explanation of the relationship between the vertical eddy viscosity and the strength of estuarine circulation.

1.3. Fortnightly modulations of estuarine circulation and the relationship between baroclinic and kinetic energy

Figure 1 shows the bathymetry of Tokyo Bay, which is about 15 km wide and 50 km long. During spring tides, the maximum velocity around the mouth of the bay reaches 1.2 m s^{-1}. The depth of the bay changes sharply from 300 m at the mouth of the bay to 15 m at the head of the bay. The outlets of the main rivers (the Edo, Ara, Tama, and Tsurumi rivers) are shown in Figure 1; the average discharges of these rivers is, respectively, 100 m^{-3} s^{-1}, 90 m^{-3} s^{-1}, 40 m^{-3} s^{-1}, and 7 m^{-3} s^{-1}. An Acoustic

Doppler current meter (ADP; Alec Electronics) was deployed at St4 to a depth of 20 m, and concurrent samples were collected over 1 min every 10 min (Hibino et al., 1999; Nakayama et al., 2000). The ADP measured east-west, north-south, and vertical velocity components at 1 m intervals.

The maximum measured current velocity was approximately 0.6 m s^{-1} in the NE-SW direction near the surface (Figure 2a). Because of the bottom friction, the velocity in the upper layer was greater than that in the lower layer, indicating that the tide provided more kinetic energy to the upper layer. The low-frequency fluctuations were found to be significant in the velocity data from which the tidal component was eliminated (Figure 2b). The predominant current direction was SW at depths of 3 m, 9 m, and 15 m during the spring tide due to the bay's topography. This suddenly changed during the neap tide, with the current direction becoming NE at a depth of 15 m, while the SW current at depths of 3 m and 9 m became stronger. This was typical of the observed fortnightly fluctuations (Figure 2b). These findings are evidence of upper-water outflow and the intrusion of oceanic water at depth during the neap tide. This process seemed to be inhibited during the spring tide.

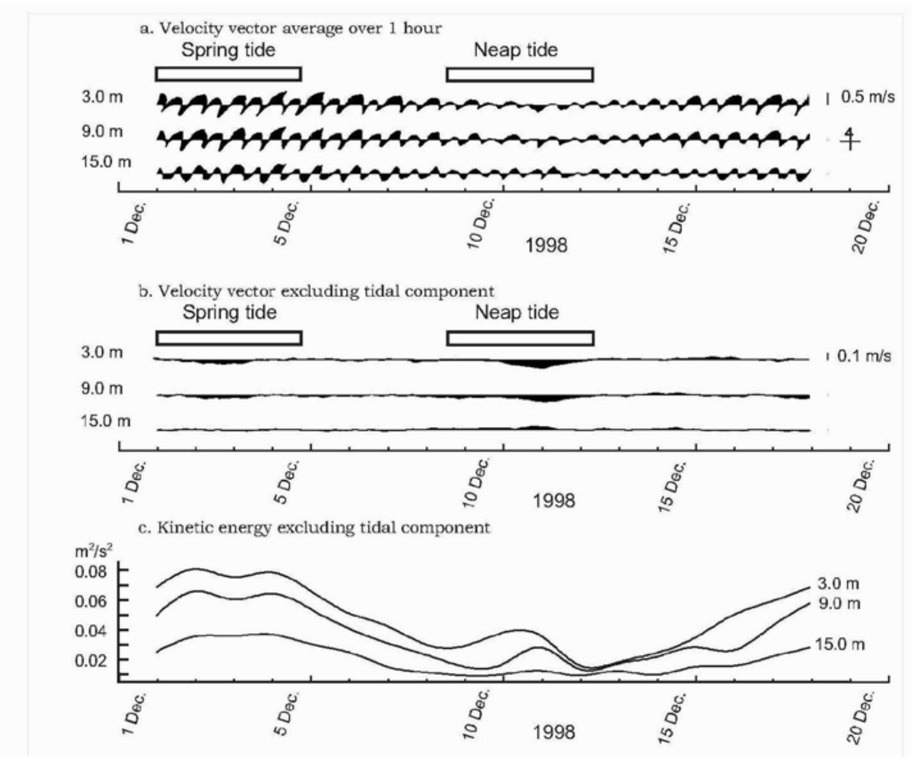


Figure 2. *Time-series plot of the horizontal velocity vector at 3, 9 and 15 m depth at site St 4, from December 2 to 19, 1998. (a) Velocity vector averaged over 1 hour. (b) Velocity vector from which the tidal component was eliminated. (c) Kinetic energy from which the tidal component was eliminated.*

The difference in the low-pass filtered kinetic energy between 3 m and 15 m was larger during the spring tide than during the neap tide, except when deep-water intrusion occurred (Figure 2c). There were neither heavy rainfalls nor large river discharges at that time; hence the influence of rainfall and river discharge on the estuarine circulation was probably negligible. Therefore, the difference in the kinetic energy between the upper and lower layers may affect the observed change in the estuarine circulation strength in Tokyo Bay.

1.4. Theoretical analysis

The effect that a difference in the kinetic energy between two layers has on estuarine circulation was recently investigated using a two-layer model (Figure 3; Nakayama et al., 2005). To resolve the effects of the tides on the estuarine circulation, the two-layer system was further divided into two parts: tidal (barotropic) and baroclinic components (Figures 3b and 3c).

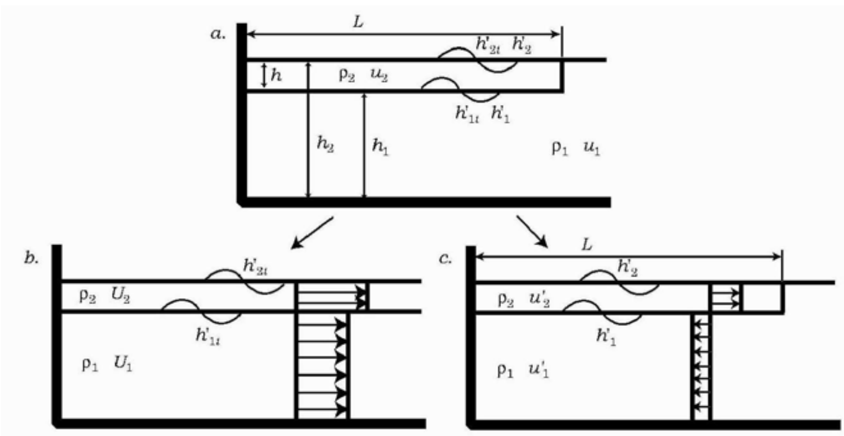


Figure 3. *(a) Schematic diagram of the two-layer system. The left and right sides respectively correspond to the head and the mouth of the bay. (b) Schematic diagram of velocity fields due to the barotropic component. (c) Schematic diagram of velocity fields due to the baroclinic component.*

The lack of baroclinic energy yields a velocity field shown in Figure 3b. In this figure, U_1 and U_2 are the velocity from the barotropic components in the lower and upper layers, respectively; $U_1 < U_2$ due to bottom friction. Because the main purpose of this study was to evaluate the effect that a difference in the kinetic energy between the lower and upper layers has on the baroclinic component, average velocities in the upper and lower layers were used to define the difference in kinetic energy between the two layers. Where the baroclinic energy is only given, it is assumed that velocity fields with u'_1 and u'_2 appear, as shown in Figure 3c, which would cause the estuarine circulation. In a two-layer system, the momentum equations for each layer are:

$$\frac{\partial u_2'}{\partial t} + \frac{1}{2}\frac{\partial}{\partial x}(U_2+u_2')^2 = -g\frac{\partial h_2'}{\partial x} + r_I\frac{1}{h}(U_2-U_1)|U_2-U_1|$$
$$-r_I\frac{1}{h}(U_2+u_2'-U_1-u_1')|U_2+u_2'-U_1-u_1'|, \tag{1}$$

$$\frac{\partial u_1'}{\partial t} + \frac{1}{2}\frac{\partial}{\partial x}(U_1+u_1')^2 = -g\frac{\partial h_2'}{\partial x} - \varepsilon g\frac{\partial h_1'}{\partial x} + r_B\frac{1}{h_1}U_1|U_1|$$
$$+r_I\frac{1}{h_1}(U_1-U_2)|U_1-U_2| - r_B\frac{1}{h_1}(U_1+u_1')|U_1+u_1'|$$
$$-r_I\frac{1}{h_1}(U_1+u_1'-U_2-u_2')|U_1+u_1'-U_2-u_2'|, \tag{2}$$

and the equations for the conservation of volume are:

$$\frac{\partial h_2'}{\partial t} - \frac{\partial h_1'}{\partial t} + \frac{\partial}{\partial x}\{U_2(h_2'-h_1')\}$$
$$+\frac{\partial}{\partial x}\{u_2'(h+h_{2t}'+h_2'-h_{1t}'-h_1')\} = 0, \tag{3}$$

$$\frac{\partial h_1'}{\partial x} + \frac{\partial}{\partial x}(U_1 h_1') + \frac{\partial}{\partial x}\{u_1'(h_1+h_{1t}'+h_1')\} = 0, \tag{4}$$

where U_1 and U_2 are the barotropic velocities, t is the time, and x is the horizontal coordinate along the bay. Here, h is the depth from the surface to the interface, h_1 is the level of the interface from the bottom, and h_2 is the water level from the bottom. g is the gravitational acceleration, ρ_2 and ρ_1 are the upper and lower layer densities, respectively, and $\varepsilon = (\rho_1 - \rho_2)/\rho$, u_2' and u_1' are the velocities from the baroclinic components in the upper and lower layers, h_{1t}' and h_1' are the displacements from h_1 due to the tide and the baroclinic components, respectively, r_B and r_I are the friction coefficients on the bottom and interface, and h_{2t}' and h_2' are the displacement from h_2 due to the tide and baroclinic components, respectively.

When the velocities generated by the tide are greater than those generated by the baroclinic energy, $U_1 >> u_1'$ and $U_2 >> u_2'$ and the displacement on both the surface and interface is smaller than the thickness of the layers, Equations (1) to (4) can be simplified and integrated from x=0 to x=L to yield:

$$\frac{h+h_1}{h_1}\int_{x=0}^{L}\frac{\partial u_2'}{\partial t}dx =$$
$$+\epsilon gh - \left\{\frac{1}{2}(U_2+u_2')^2|_{x=L} - \frac{1}{2}(U_1+u_1')^2|_{x=L}\right\}$$
$$+\int_{x=0}^{L} f\frac{h_1+h}{h_1 h}(u_1'-u_2')\{|U_1+u_1'-U_2-u_2'|+|U_1-U_2|\}dx$$
$$+\int_{x=0}^{L} f_B\frac{1}{h_1}u_1'\{|U_1+u_1'|+|U_1|\}dx, \tag{5}$$

$$\frac{h+h_1}{h_1}\int_{x=0}^{L}\frac{\partial u_2'}{\partial t}dx = E_0 - E_1 + E_3 + E_3, \tag{6}$$

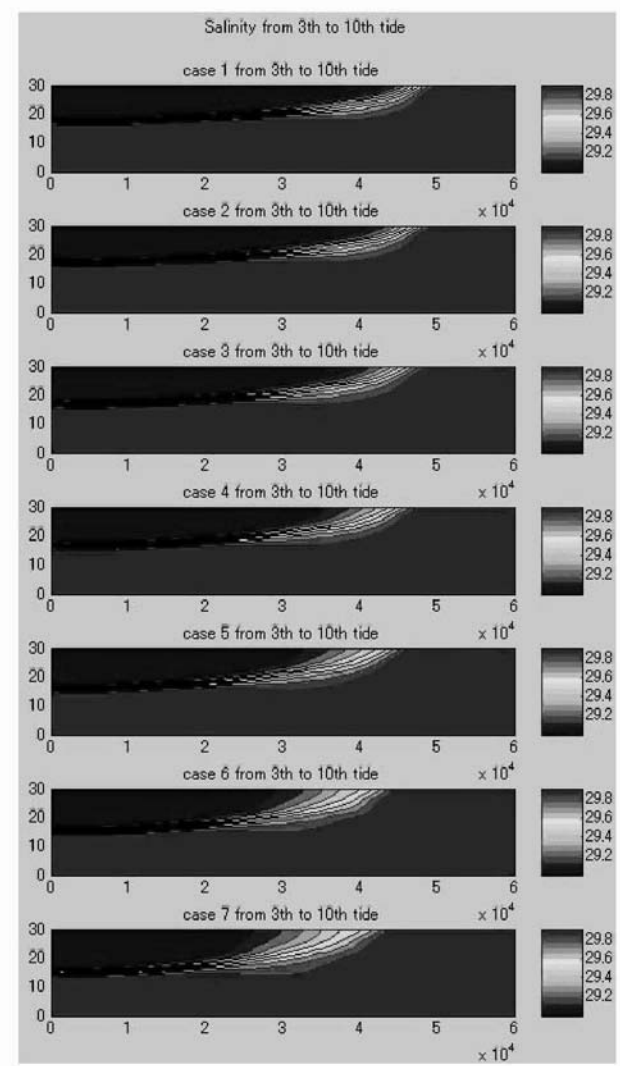
where E_0, E_1, E_2 and E_3 correspond to the first, second, third and fourth term in Equation (5). The change in the withdrawal distance of the front can be derived as $u'_2\Delta t$ by assuming that u'_2 is constant in the upper layer.

1.5. Validation of theory using the MEL3D non-hydrostatic three-dimensional model

A three-dimensional non-hydrostatic tidal model, MEL3D, was used to verify the theoretical model (Nakayama et al., 2001; Nakayama and Satoh, 1999; Nakayama and Imberger, 2005). A turbulent closure scheme (Deardorff, 1975 and 1980; Klemp and Wilhelmson, 1978; Schumann, 1991; Lilly et al., 1967; Schmidt and Schumann, 1989) and the cubic-interpolated pseudoparticle (CIP) method (Yabe et al., 1990; Yabe et al., 1991; Yabe et al., 2005) were used in the MEL3D model. The change in the turbulent Prandtl number caused by the Brunt-Vaisala frequency (Schmann, 1991) was used for calculating the vertical eddy diffusion under conditions of a steady stratification. The movement of the surface elevation was calculated using the arbitrary Lagrangian-Eulerian computing method (ALE method) (Hirt et al., 1972). The residual cutting method was used to solve Poisson's equation (Tamura et al. 1997).

The same topography as used in the theoretical analysis was used in the numerical computation (Figure 3a). The numbers of grid points were 60, 10 (horizontal), and 30 (vertical), and the grid sizes were 1.0 km, 1.0 km (horizontal), and 1.0 m (vertical). Following the field experiments of Hibino et al. (1999), the upper and lower layer thicknesses were 10 m and 20 m, respectively, the tidal period was 43200 s, L was 40000 m, ε was 0.00076, and Δt was 72 s. A non-slip boundary condition was imposed at walls. All velocity components were set to zero at the boundary, while the pressure and the values for wave height were extrapolated by applying a non-normal-gradient condition in the horizontal direction. A radiative boundary condition was imposed on the seaward open boundary.

The shape of the front was given by a step-like line as an initial condition. An extreme case where the difference in kinetic energy between the upper and lower layers was much greater than the baroclinic energy was included in the numerical computations to investigate the range of applicability of the theory. Therefore, tidal amplitudes at the open boundary were given that were larger than the actual amplitudes: 1.0 m, 1.25 m, 1.5 m, 1.75 m, 2.0 m, 2.25 m, and 2.5 m for cases 1-7. The model was run for ten tidal cycles with the horizontal density difference not included until a periodic flow was obtained at the end of the third tidal cycle (Animation 1).



Animation 1. *Predicted, salinity distribution for cases 1-7 from the 3rd tidal cycle to 10th tidal cycle.*

The interface was defined as the line along which the density was $(\rho_2+\rho_1)/2$ and the average interface was obtained for cases 1-7 by averaging the current interface for one tidal period (Figure 4, Animation 2). Clearly, the front in case7 moved landward and seaward in case 1 moved toward the right from the seventh to tenth tide. To quantitatively evaluate the net withdrawal distance of the upper water, the location of the upper-water front was computed by averaging the horizontal distance of the interface from the initial condition in the upper layer when the tidal component was excluded. The net withdrawal distances from the seventh to tenth tide, as obtained from a three-dimensional numerical computation, were 1.624 km, 1.226 km, 0.628 km, 0.142 km, -0.274 km, -0.644 km, and -1.514 km for cases 1-7. These values agreed well with those obtained from the theory.

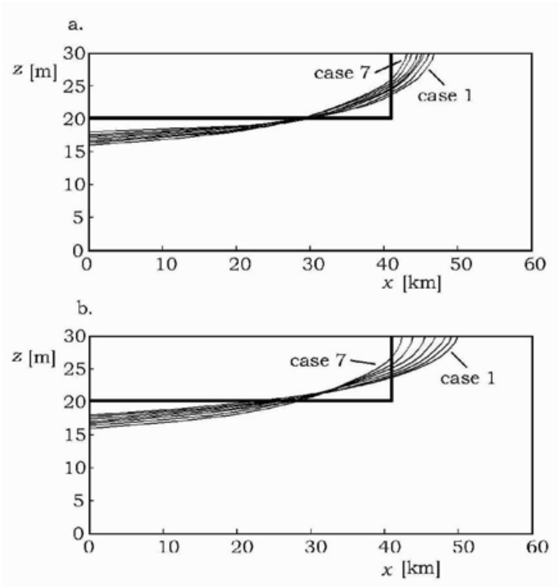
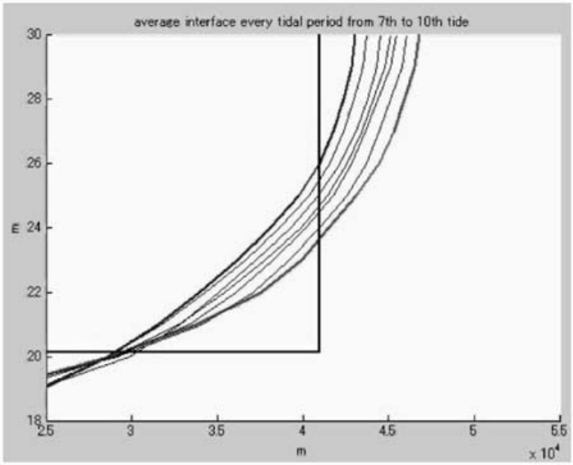


Figure 4. *Predicted, tidally-averaged location of the interface for cases 1-7 at (a) the 7th tidal cycle and (b) 10th tidal cycle.*

1.6. Vorticity effect and N_{id}

As the withdrawal distances from the simulation and theory were in good agreement, a theoretical understanding of the mechanism suppressing the withdrawal distance may have been achieved. By comparing of the order of each term in Equation (6), E_1/E_0 was found to be the most suitable non-dimensional parameter, N_{id}, for modeling the withdrawal distance:



Animation 2. *Predicted, tidally-averaged location of the interface for cases 1-7 from the 7th tidal cycle to 10th tidal cycle.*

$$N_{id} = \frac{(U_2 + u_2')^2/2 - (U_1 + u_1')^2/2}{\epsilon g h}. \tag{7}$$

Equation (7) demonstrates that a negative acceleration might be applied to the front when Nid > 1 (Nakayama et al., 2005). Nid can also be described in terms of vorticity by modifying Equations (1) to (4) and integrating from x = 0 to L.

$$L\frac{\partial \omega}{\partial t} = \frac{\epsilon g h}{h_2} - \frac{1}{2}\{(U_2 + u_2' + U_1 + u_1')W\}\,|_{x=L}$$
$$- \int_{x=0}^{L} f\frac{h_2}{h_1 h}\{|U_1 + u_1' - U_2 - u_2'| + |U_1 - U_2|\}\omega dx$$
$$+ \int_{x=0}^{L} f_B \frac{1}{h_1 h_2} u_1' \{|U_1 + u_1'| + |U_1|\} dx. \tag{8}$$

where $\omega=(u'_2-u'_1)/h_2$ and $W=\{(U_2+u'_2)-(U_1+u'_1)\}/h_2|_{x=L}$. ω is the vorticity indicating the strength of estuarine circulation. W is the vorticity due to tide; the larger the tidal amplitude, the larger W becomes. The first term on the right side is the baroclinic component, which enhances estuarine circulation. The ratio of the second term to the first on the right side of Equation (8) corresponds to $-N_{id}$. $W > 0$ when $(U_2 + u'_2 + U_1 + u'_1) > 0$ and $W < 0$ when $(U_2 + u'_2 + U_1 + u'_1) < 0$. Thus, the second term on the right side is always negative and inhibits the estuarine circulation (Figure 5). Therefore, N_{id} is also a useful factor for evaluating the effect of the tides on the strength of estuarine circulation due to the baroclinic component in terms of the vorticity equation.

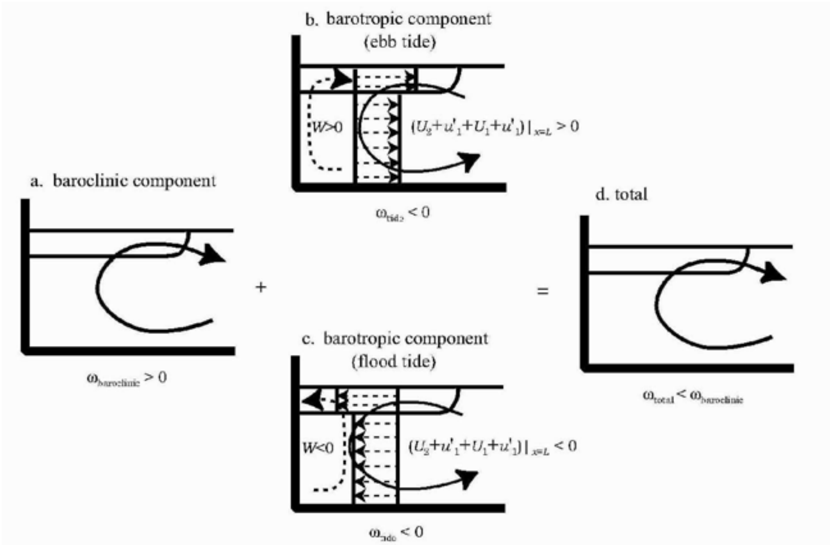


Figure 5. *Schematic diagram showing the effect of vorticity caused by the baroclinic component and tide. $\omega_{baroclinic}$ denotes the strength of the estuarine circulation due to the baroclinic component, ω_{tide} denotes the strength of the estuarine circulation due to the tide, and ω_{total} denotes the total strength of the estuarine circulation. (a) Strength of estuarine circulation due to the baroclinic component. (b) Solid arrow: estuarine circulation due to the tide during ebb tide. Dotted arrow: vorticity due to the tide during ebb tide. (c) Solid arrow: estuarine circulation due to the tide during flood tide. Dotted arrow: Vorticity due to the tide during flood tide. (d) Total strength of the estuarine circulation.*

1.7. Application of the theory to Tokyo Bay

An attempt was made in December 1998 to measure in the field of spring-neap tide cycle modulation of the upper-water. Equation (5) was used to tidally-induced vorticity impact on the estuarine circulation in Tokyo Bay. Field experiment results obtained by Hibino et al. (1999) were used to estimate the values for U_1, U_2, and ε. In terms of withdrawal speed, the calculated withdrawal distance from the spring to neap tide was 5.9 km. This value was consistent with the value (6 km) from field experiments (Hibino et al., 1999). Because the averaged value of N_{id} was 1.06 during the spring tide and 0.24 during the neap tide, it appears likely that at spring tide the greater value of N_{id} suppressed upper-water withdrawal. N_{id} may therefore be useful for estimating the occurrence of fortnightly modulations in estuarine circulation.

1.8. Conclusion

To determine the mechanism responsible for fortnightly modulations in estuarine circulation during spring-neap tides, a two-layer model consisting of tidal (barotropic) and baroclinic components was developed. Through a vorticity analysis, the vorticity due to tide, W, was found to be the main component of the total vorticity, ω, highlighting the strength of estuarine circulation. The larger was the vorticity due to tide, the lower was the vorticity of estuarine circulation.

N_{id} was thus found to be a useful parameter for estimating the effect that the vorticity due to tides has on estuarine circulation; e.g., when the maximum N_{id} is close to 1, the withdrawal of the front will be inhibited if the two-layer system is valid. Therefore, it is clear that fortnightly modulations in estuarine circulation occur when the maximum value of N_{id} is close to 1 or greater than 1 during the spring tide and close to zero during the neap tide. Tokyo Bay changes from $N_{id} > 1$ to $N_{id} < 1$ from spring to neap tides in December 1998.

2. HORIZONTAL CIRCULATION AND CONVERGENCE DUE TO WIND DURING SUMMER

2.1. Introduction

Nutrients or toxic substances flowing from rivers into Tokyo Bay are transported by physical processes and taken in by aquatic plants or animals. This may affect the water quality and the ecological system of Tokyo Bay, which is a typical enclosed bay in Japan. Therefore, understanding the transportation process is necessary to manage Tokyo Bay's ecological system and biodiversity. The transportation process, which can greatly affect water quality in an enclosed bay, may be controlled by long-term processes, such as seasonal changes in the residual current (see for example, Guo and Yanagi, 1996 and 1998). In general, the flow field in an enclosed bay depends on stratification and external forces such as tides and wind. The effects of stratification, tides, and wind are defined as those of the baroclinic, barotropic, and boundary external force components. Unoki (1998) demonstrated that the baroclinic component is the dominant driver of the seawater exchange in Tokyo Bay. In contrast to the baroclinic component, the wind may mainly influence the residual current around the head of Tokyo Bay because the water depth in that part of the bay is less than 20 m and the barotropic component does not play a great role in the flow field.

Previous studies have shown the significance of the wind on a stratified flow field. In particular, Kasuya et al. (2002) demonstrated that because of the wind the level of the interface was lowered around the center of the bay head, taking on a bowl-like shape, during summer. A decrease in the interface level was also found to cause convergence of the upper water around the centre of the bay head.

Furthermore, HF radar experiments done at the same time as the density field experiments revealed a clockwise horizontal circulation around the bay head, and the center of this circulation was located at the same point as the center of the interfacial trough and the convergence. At the time of the field experiments, the river discharge was small, so the influence of river inflow on the clockwise horizontal circulation was probably negligible. It was suggested that wind might have played a role in the decrease of the interfacial level and the convergence of the upper water because of the spatial variation of the wind field; however, no theoretical explanations of the interfacial trough and convergence were offered by Kasuya et al. (2002).

2.2. Previous studies

There have been previous studies related to horizontal circulation around the head of Tokyo Bay. A long-term numerical simulation was done using a hydrostatic three-dimensional model, in which tide and wind effects were ignored, to confirm that the clockwise horizontal circulation occurred because of the estuarine circulation (Tanaka, 2001). Fujiwara et al. (1997) demonstrated the same phenomena in Osaka Bay. They found that the clockwise horizontal circulation was caused by the Coriolis effect on the upwelling around the bay head due to estuarine circulation. They also found that the clockwise horizontal circulation was enhanced during a northerly wind and suppressed during a southerly wind. Because the upwelling resulted in clockwise horizontal circulation, the level of an interface rose around the bay head, taking on a bowl-like shape, which was different from the phenomenon shown by Kasuya et al. (2002).

Internal waves have also been considered influencing the horizontal circulation around the bay head (Suginohara, 1974; Suzuki and Matsuyama, 2000). A northerly wind was found to cause upwelling in the eastern part of Tokyo Bay and downwelling in the western part. The interfacial displacement caused by the upwelling proceeded along the eastern coast as an internal Kelvin wave, and then proceeded along the coastal region and collapsed into downwelling in the western part of the bay. The internal Kelvin wave thus drove the clockwise horizontal circulation around the bay head. Nakayama et al. (2004) demonstrated that the clockwise horizontal circulation and the convergence observed during the winter by Kasuya et al. (2002) occurred during a southerly wind in summer rather than a northerly wind. Therefore, this theory cannot also be applied to explain the phenomenon described by Kasuya et al. (2002).

2.3. Field experiment in August 2001

A field experiment was carried out to measure the salinity and water temperature at 65 stations with a spatial interval of about 3.5 km and a vertical interval of 0.1 m in Tokyo Bay on August 2nd, 6th, and 10th in 2001. The bay was divided into eight areas, and one ship was used in each area so that the field experiments could be completed within five hours. The field experiments were done during daylight. Salinity and water temperature, from which the density was calculated, were measured with AST-500 sensors (Alec Electronic Corporation).

The horizontal distributions of density at depths of 8 m and the vertical distribution of density along line A-A' are shown in Figures 6a and b. At a depth of 8 m a lower density area appeared with a diameter of about five km around the centre of the bay head (Figure 6a). A pycnocline was formed from a depth of 5 m to 10 m and the level of the interface was found to drop in the area around st.16 and st.23 (Figure 6b). Interestingly, the location of the centre of the decrease in the level of the interface matched that of the lower density area at a depth of 8 m. Therefore, we concluded from the field experiments that the decrease in the level of the interface appeared with the convergence of the upper-layer water around the head of Tokyo Bay.

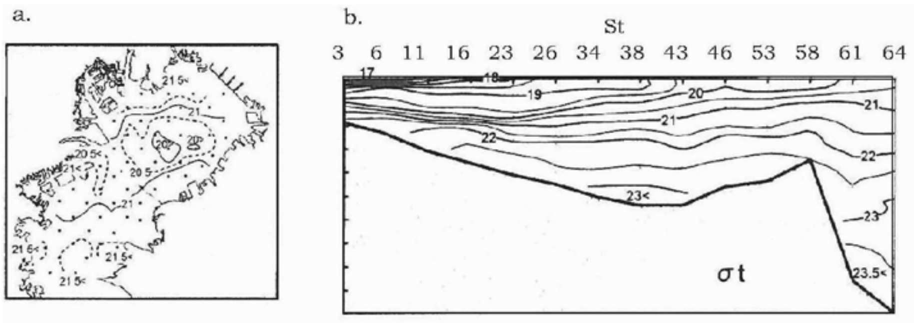


Figure 6. *(a) Horizontal distribution of density, σ_t, at a depth of 8 m. (b) Vertical distribution of density, σ_t, along the longitudinal line, A-A' shown in Figure 1.*

2.4. Wind field around Tokyo Bay

The effect of wind was expected to be the dominant influence on the flow field, which caused the decrease in the level of an interface and convergence, around the head of Tokyo Bay because the tide effect was negligible and the river discharge was small enough to be ignored on August 10th, 2001. In addition, the wind effect might be greater during the summer than the winter because a clearer interface is formed in summer and the upper-layer depth, which is influenced by wind, becomes smaller. 22 meteorological monitoring stations were used to estimate the wind field around Tokyo Bay. An exponential weighted function was used to interpolate the wind field from the data of these 22 stations. High-frequency components whose period was less than 24 hrs were eliminated from this data because the focus of this study was the low-frequency processes. The horizontal distribution of wind vorticity was computed from the estimated wind field so that the influence of the wind field on the horizontal circulation could be determined (Figures 7a and b). Southerly or southwesterly winds were dominant on the 9th and 10th of August 2001 when the lower density appeared at the center of the bay head at a depth of 8 m. The wind around the western part of the bay head was stronger than that around the eastern part. This variation in the wind field caused the negative vorticity around the bay head.

2.5. Wind vector and vorticity in August 2001

Time series of wind vector and vorticity data for August 2001 were obtained by averaging the data from five representative points at the head of Tokyo Bay (Figures 8a and b). Southerly winds caused negative vorticity, while northerly winds caused positive vorticity. This might have been due to a topographical effect: the hills on the eastern side of Tokyo Bay are higher than those on the western side. The higher hills on the eastern side seemed to reduce the speed of wind adjacent to the ground surface because of the roughness of the terrain and the blocking effect. Furthermore, it was found that as the wind speed increased, the absolute value of the vorticity also increased.

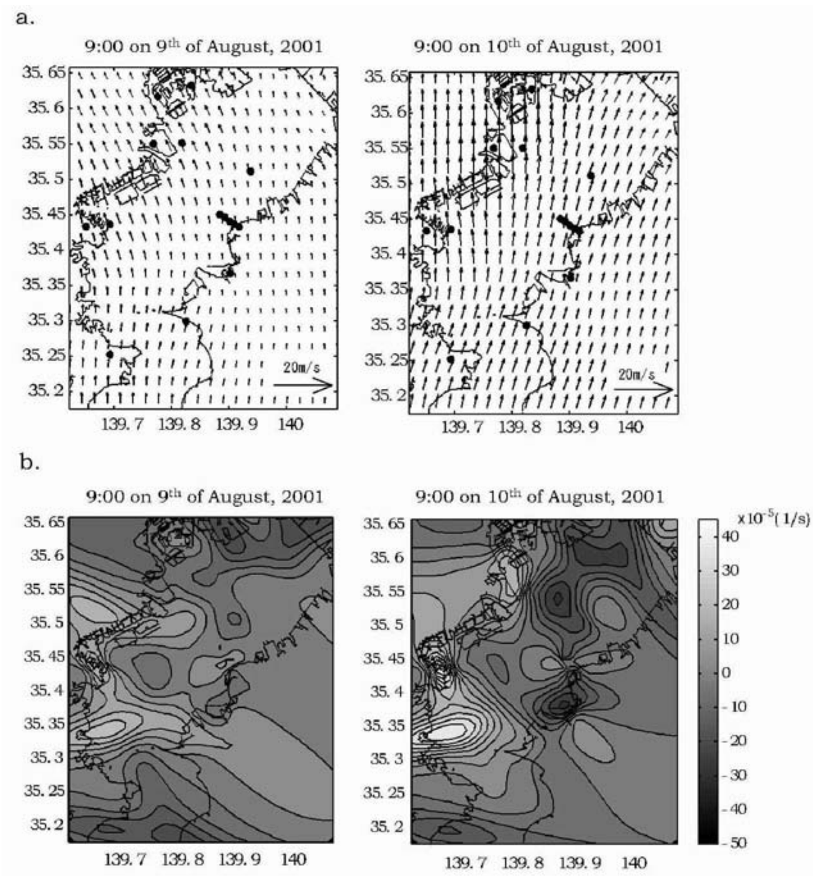


Figure 7. *(a) Estimated wind field on August 9 and 10, 2001. (b) Vorticity of wind on August 9 and 10 in 2001.*

2.6. Application of non-hydrostatic three-dimensional model, MEL3D

Simplified initial conditions were used to reproduce the decrease in the level of the interface with the MEL3D non-hydrostatic three-dimensional model. A two-layer system was given as the initial conditions, with a difference in specific density of 0.0015 and an interface depth of 10 m, based on field experiment results from the 10th of August 2001. The wind field shown in Figure 7 was used in the numerical computation. The computation result confirmed that the clockwise horizontal circulation around the bay head was created by a wind field in which negative vorticity existed around the bay head (Figure 8a, Animation 3). Although simplified initial conditions were used in the numerical computation, the horizontal distribution of density was similar to that obtained through the field experiments (Figures 6a and 8b). The convergence was also found to result from the Coriolis effect on the clockwise horizontal circulation in the upper layer. This may be evidence that the clockwise horizontal circulation and convergence around the bay head were induced by the wind effect when the negative vorticity of wind appeared around the bay head during southerly winds in summer.

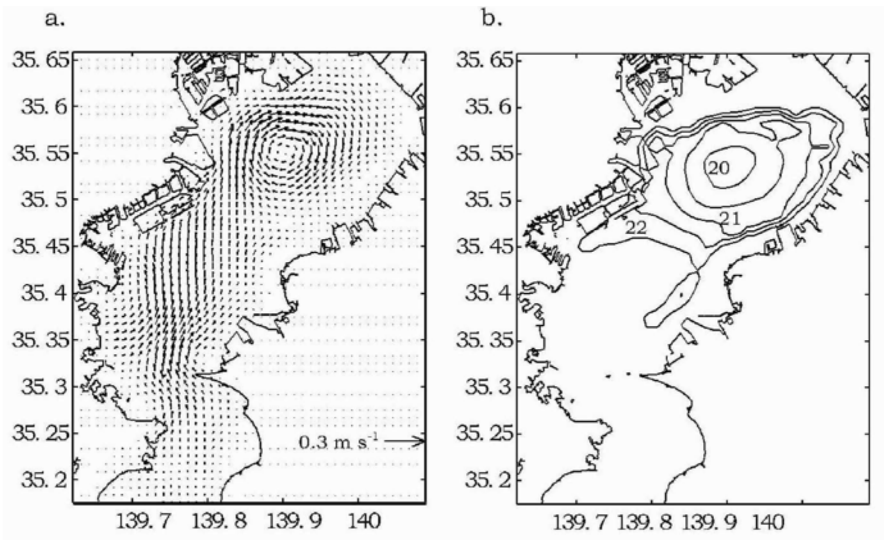
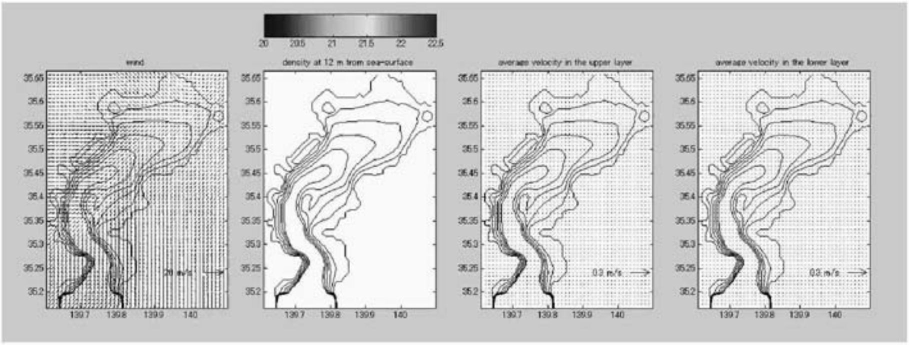


***Figure** 8. (a) Average velocity vector in the upper layer from the numerical computation. (b) Density distribution at the initial interfacial level from the numerical computation.*



Animation 3. *From left to right; Estimated wind field, Vorticity of wind, Average velocity vector in the upper layer, and Average velocity vector in the lower layer on August 9 and 10, 2001.*

3. CONCLUSION

It is found that southerly winds created negative vorticity around the head of Tokyo Bay because of the topographical features surrounding Tokyo Bay. The higher the wind speed, the larger the absolute value of vorticity became during our experiments in the summer of 2001. Through computational analysis using the MEL3D non-hydrostatic three-dimensional model, it was found that negative vorticity was caused by clockwise horizontal circulation around the bay head. Convergence in the upper layer was also found to be due to the Coriolis effect on the clockwise horizontal circulation. Because the southerly wind is usually dominant during summer, the clockwise horizontal circulation and convergence are likely to often appear around the head of Tokyo Bay in summer.

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CHAPTER 5

EFFECTS OF OCEANIC WATER INTRUSION ON THE TOKYO BAY ENVIRONMENT

HIROFUMI HINATA

1. INTRODUCTION

In Tokyo Bay (Figure 1), the river discharge into the head of the bay gives rise to horizontal density gradients that lead to an estuarine circulation (Nakayama, 2005) as shown in chapter of "Circulation Phenomena in Tokyo Bay". If the oceanic water, i.e. the Kuroshio warm water, approaching to the bay and it is denser than the bay bottom water, then a classical estuarine circulation prevails whereby there is an outflow of bay waters at the surface and an inflow of oceanic water in the lower layer. However, fluctuations of the circulations in the bay can occur on time scales ranging from a few days to a few months due to the effects of climate and/or Kuroshio current fluctuations in the time scales. As a result, the depth at which oceanic water intrudes is also though to vary with oceanic and climate conditions.

Yanagi et al. (1989) measured the water quality at the mouth of Tokyo Bay. They suggested that a three-layer circulation may exists whereby the Kuroshio water flows into the middle layer of the bay where the water density is the same, while highly turbid water flows out from the bay in the surface layer and lower layer. If this hypothesis is correct, then the inflow of oceanic water into the middle layer should have a significant effect on the behavior of the hypoxic water mass that develops in the lower layer of the bay during the summer.

In September 1998, the intrusion of oceanic water into the middle layer of Tokyo Bay occurred intermittently. At this time, Hinata et al. (2001) used ADCP (Acoustic Doppler Current Profiler) observations and water quality observations to take direct measurements of the flow structures inside the bay and the heat/material flux between the bay and ocean. This chapter revisits the observations of Hinata et al. (2001) to describe the three-dimensional flow structures inside the bay and the heat and material flux between the bay and ocean that occur when oceanic water intrudes into the middle layer inside the bay during the summer.

E. Wolanski (ed.), The Environment in Asia Pacific Harbours, 67–78.

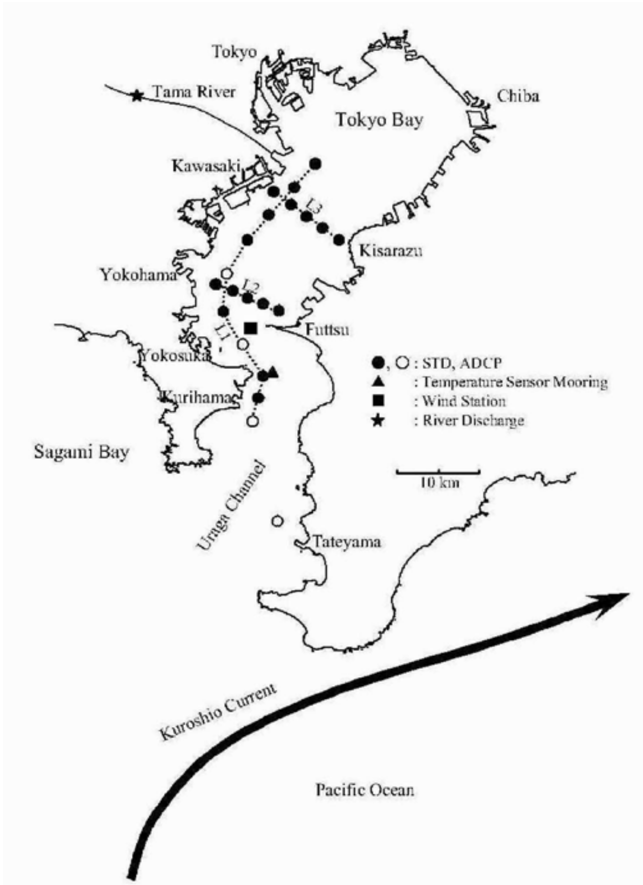

Figure 1. *Map of study area. Although the water temperature mooring was made in a location where the water was 50 m deep, the actual measurements were made extended from 0 to 30 m depth. Open circle plots are referred to in section 3.*

2. OCEAN STRUCTURES AND FLOW STRUCTURES INSIDE THE BAY DURING MIDDLE LAYER INTRUSION OF OCEANIC WATER

Figure 2 shows the water temperature (0-30 m) and wind velocity vectors at the mouth of Tokyo Bay during the period August to September 1998, together with the river discharge of the Tama River during the same period. Although the water temperature measurements were made in a location where the water was 50 m deep, the actual measurements were made extended from 0 to 30 m depth. The water temperature and wind velocity are 25-hour moving averages. In August, the wind blew alternately from the north and south with a period of a few days, while in September the wind blew predominantly from the north except during the passage of typhoons #5 and #7. Due to the effects of typhoons and autumn rains, the river discharge was the largest between the end of August and early September, and again in late September.

In August temperature stratification developed and it fluctuated at time scales of a few days. In September, the middle layer temperature (at depths of 10-30 m) intermittently rose sharply (as indicated by the arrows in Figure 2). The change in water temperature was much smaller in the surface layer (at depths of 0-6 m) than in the middle layer. These sharp increases in water temperature all occurred during periods when the wind was blowing predominantly from the north. Even in August there were periods when the water temperature rose when the wind was predominantly from the north, but on these occasions the increase in water temperature was significantly lower than those that occurred in September. Accordingly, it seems unlikely that these sharp increases in water temperature are primarily attributable to the presence of northerly winds. The variations of tidal range may not be responsible either for these increases in water temperature because the first increase occurred just after a neap tide, whereas the second and third increases occurred between a spring tide and a neap tide.

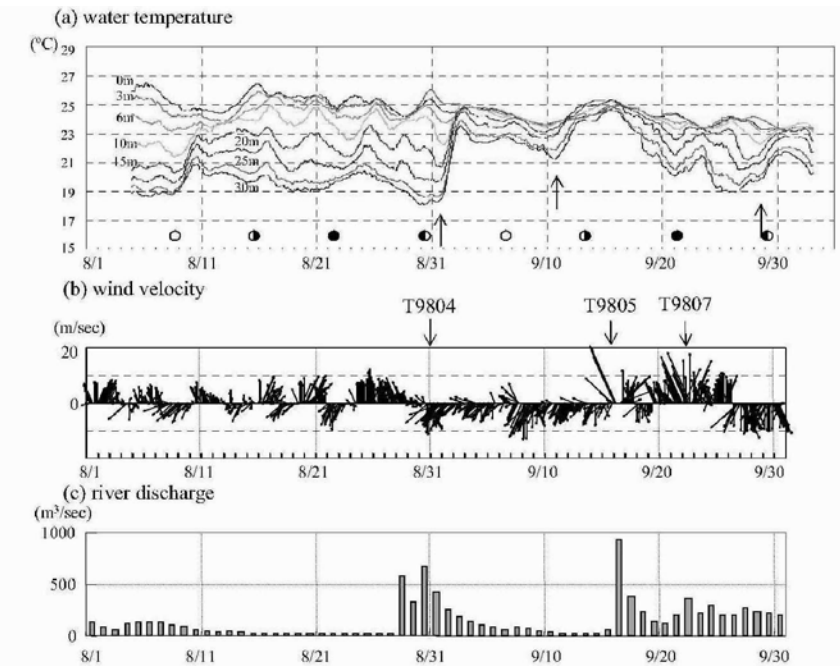

Figure 2. *Time series plot for August to September 1998 of (a) 25-hr running average water temperature at the bay mouth, of (b) wind velocity at the bay mouth, and of (c) river discharge of Tama River. T9804, T9805, and T9807 indicate typhoons that passed through the study area. Arrows in (a) indicate the abrupt increase of water temperature at the bay mouth.*

Further, it appears unlikely that the surface heat transfer from the atmosphere could have caused a sharp change in the water temperature of the middle layer but not the surface layer. It appears thus that the middle layer temperature increases were caused by the horizontal advection of warm water masses. Therefore, to study the physical phenomenon behind these sharp increases in water temperature, it is necessary to consider the oceanic and flow structures inside the bay and around the bay mouth, and this can be done using the ADCP and the water quality observations.

Hinata et al. (2001) used ADCP and water quality observations to measure the oceanic and flow structure inside the bay and around the bay mouth during the second episode where sharp increases in water temperature occurred. Figure 3 shows their data of the water temperature, salinity, turbidity, density and baroclinic flow distribution (baroclinic velocity component) along the transect L1 on September 15, 1998.

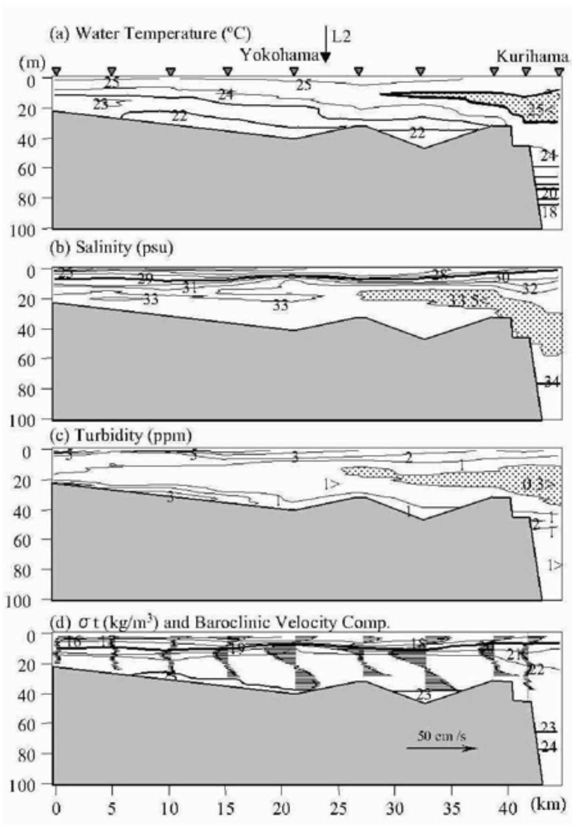

Figure 3. *Vertical distributions along the transect L1. (a) water temperature, (b) salinity, (c) turbidity, and (d) σ_t and the baroclinic component of NS velocity on 15 September 1998 along L1. Triangle indicates the location of ADCP and STD points. Hatched region represents a warm (≥ 25 °C) water mass with low turbidity (≤ 0.3 ppm) which can clearly be seen flowing into the bay in the middle layer.*

The hatched region indicates a warm (≥ 25 °C) water with low turbidity (≤ 0.3 ppm) which can be seen intruding in the middle layer from the bay mouth to the central part of the bay. The salinity of this warm water mass is 33.5-34.0, which is about 1 less saline than the Kuroshio water.

This intrusion of warm water into the middle layer developed mainly along the σ_t = 22 kg m^{-3} isopycnal as far as the sea off Yokohama. The maximum velocity of the intrusion was about 0.3 m s^{-1} in the vicinity of the bay mouth. Conversely, from the surface layer and lower layer between the bay mouth and the central part of the bay, there were outflows of warm, low salinity, highly turbid water from the surface layer of the bay, and cold, highly turbid water from the lower layer of the bay, respectively. This resulted in the formation of a distinct three-layer flow structure in the region between the bay mouth and the central part of the bay. Figure 3 demonstrates that the abrupt water temperature increase occurring on September 11, 1998 was caused by the intrusion of a warm oceanic water mass into the middle layer of the bay. This three-layer flow structure, which was confirmed by ADCP observations, supports the flow structure inferred by Yanagi et al. (1989) from the water quality measurements at the bay mouth.

However, the results of observations made along the transect L2 at the bay mouth (Figure 4) clearly show that the flow and oceanic structure at this time were more complex because they were 3-D in nature. Although a three-layer flow structure developed on the deeper western side of the section, the bay water flowed seaward at all depths on the eastern side. The warm water mass intruded into the bay at speeds of up to 0.3 m s^{-1} and was around a core on the western side. Conversely, the highly turbid bay water flowed seaward in the surface layer and bottom layer at the western side of the section, and in all layers at the eastern side. Although not shown in this figure, the mass of warm water intruded towards the head of the bay at the eastern side of the transect L3 (see Hinata et al., 2001, Figure 11). This eastward shift of the intruding warm water as it flowed toward the head of the bay was presumably caused by the earth's rotation. However, in the cross section at the bay mouth, the maximum velocity of the intruding warm water was much smaller than at the bay mouth, and the temperature and salinity of the water were also lower presumably due to the mixing with the bay water.

3. CHANGES IN THE DENSITY BALANCE BETWEEN THE OCEANIC WATER AND THE WATER INSIDE THE BAY DURING AUGUST AND SEPTEMBER

Although not shown here, similar ADCP and water quality measurements were also made on 18 and 19 August 1998. From these measurements, it was confirmed that the flow structure in the bay during mid-August (when there were no sharp temperature increases) was that of a classical two-layer (upper/lower) estuarine circulation. Hence, warm, highly turbid bay water with low salinity flowed out from the surface layer, while cold oceanic water with high salinity flowed in from the lower layer.

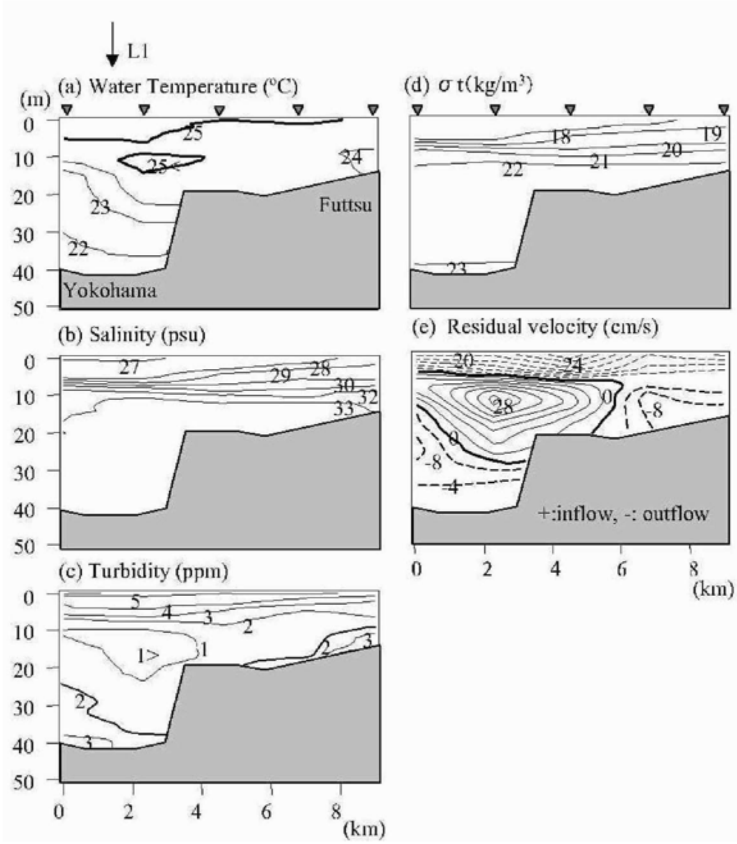

Figure 4. *Same as Figure 2 but measured along the transect L2. Residual velocity was calculated based on 12-hr shuttle ADCP observation along L2.*

Figure 5 shows the vertical density distributions measured at STD stations from the central part of the bay (off Yokohama) to Uraga Channel (off Tateyama) during the periods August 3-12 and September 7-10. The later period was just before the second abrupt water temperature increase. A comparison of densities at equal depths reveals that in early August the density basically became larger seaward, and that the density was larger seaward in the upper layer as shown in early August whereas the density was larger on the bay side in the lower layer during the period September 7-10. In early September, the density of the surface layer at the central part of the bay was smaller than in early August mainly due to the increased fresh water discharge into the bay. As a result, the horizontal density gradient between the Uraga Channel and inside the bay became larger than it had been in early August. The density in the lower layer (25-50m) became smaller seaward due to the warm water mass (T = 25-26 °C, S = 33.5-34, σ_t= 22 kg m^{-3}) in the Uraga Channel extending to 40 m depth.

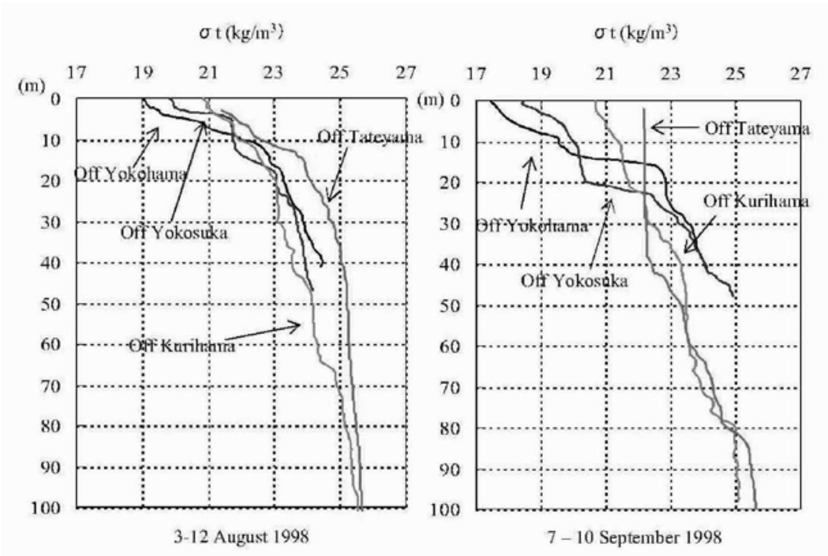

Figure 5. *Vertical distributions of σ_t at STD stations from the bay head to the mouth of the bay indicated by ○ in Figure 1: (a) August 3-12, 1998; (b) September 7-10, 1998.*

Thus, the horizontal density balance between the bay and the ocean differed significantly between these two periods. In August 5-12, 1998, a classical two-layer estuarine circulation pattern developed with an outward-flowing surface layer and an inward-flowing lower layer. However, in the horizontal density gradient of September 7-10, 1998, a three-layer flow structure developed in which the warm water mass in the Uraga Channel surface layer intruded into the middle layer of the bay where the density of the bay water was same as that of the warm water, while the bay water flowed out from the surface layer and lower layer (Fujiwara and Yamada, 2002). From late August until early September, this three-layer structure was presumably strengthened by northerly winds and by the larger horizontal density gradient due to the increase of fresh water inflow into the bay (Hinata, 2000).

During the period when intrusion of oceanic water into the middle layer occurred, the Kuroshio current approached the Uraga channel (Hinata et al., 2001). However, judging from the salinity of the oceanic water that intruded into the middle layer (33.5-34.0), it is apparent that the water mass was not the Kuroshio water (salinity 34.7) but coastal water that had been affected by fresh water. In addition, this water appeared to be affected by surface mixing by strong northerly winds in September. However, at present it is still not clear the generation mechanism for this water mass, and for the intermittent middle layer intrusion. These observations point to a complex oceanic influence on the circulation in Tokyo Bay, sketched in Figure 6. These findings show that the flow structure that occurs in the bay during a middle layer intrusion of oceanic water is very different

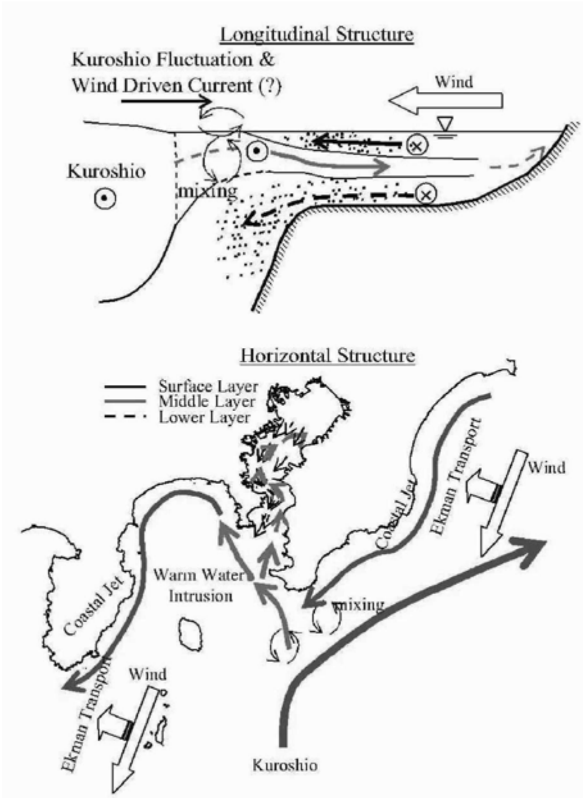

Figure 6. *Schematic diagram of the flow structure during oceanic warm water intrudes into the middle layer of the bay.*

from the classical estuarine circulation. The flux of water mass and heat, and the behavior of the hypoxic water mass, are thus very dependent on the details of the prevailing oceanographic processes.

4. COMPARISON OF THE HEAT/MASS FLUX DURING A CLASSICAL ESTUARY CIRCULATION AND DURING A MIDDLE LAYER INTRUSION OF OCEANIC WATER

In this section, the heat and material flux into/out of Tokyo Bay are compared for the situation on August 18-19, 1998, when a classical estuarine circulation prevailed, and September 15, 1998, when there was a middle layer intrusion of oceanic water. Figure 7 shows the flux distributions of heat, salinity and turbidity passing through the cross the transect L1at the bay mouth.

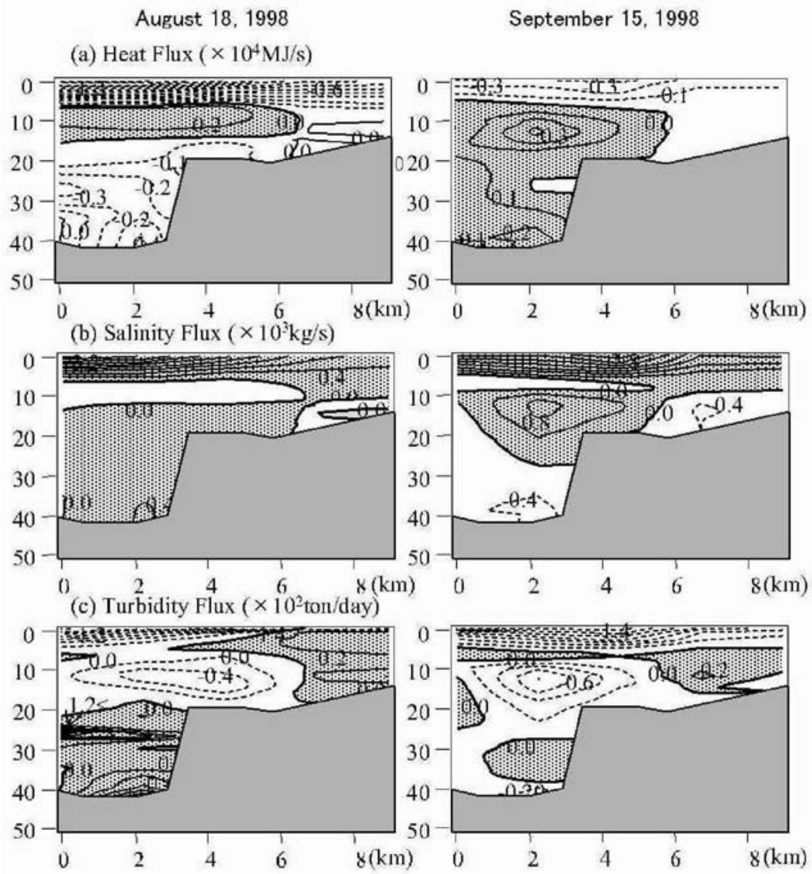

Figure 7. *Heat flux, (b) salinity flux and (c) turbidity flux across transect L2 on August 18, 1998 (left panels) and on September 15, 1998 (right panels). Hatched region indicates a landward flux.*

4.1. Heat flux

During the middle layer intrusion (September 15, 1998), heat flowed out into the ocean from the surface layer at the western side of transect and from all layers at the eastern side. However, since warm coastal water intruded into the middle layer on the western side and cold water flowed out into the ocean from the lower layer on the western side, the net heat flux was from the ocean into the bay. On the other hand, during the estuarine circulation prevailed (August 18-19, 1998), the warm surface layer in the bay flowed out into the ocean while cold oceanic water flowed

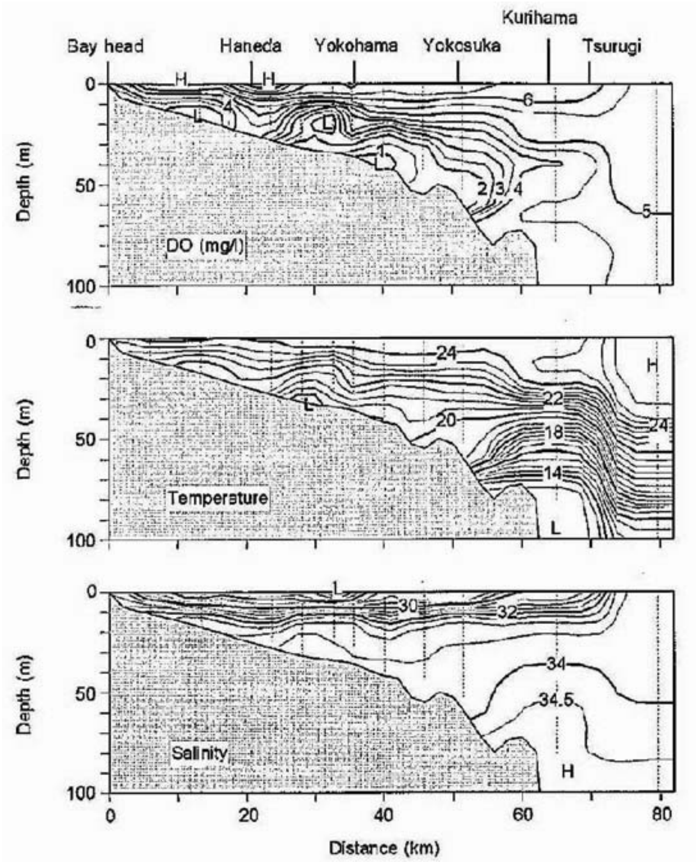

Figure 8. *Longitudinal distributions of (a) dissolved oxygen (DO) in mg l^{-1}, (b) water temperature (°C), and salinity on September 7, 1998 (after Fujiwara and Yamada, 2002).*

into the bay from the lower layer, resulting in a net heat flux from the bay into the ocean.

4.2. Salinity flux

During the middle layer intrusion (September 15, 1998), there was a seaward salinity flux in the lower layer. However, the oceanic water intruded into the bay in the middle layer and the less saline bay surface water flowed out into the ocean, resulting in a net salt flux into the bay. During the estuarine circulation event (August 18-19,1998), salinity was transported from the ocean into the bay over almost the entire cross section. This is because a less saline bay surface layer flowed out into the ocean while saline ocean water flowed into the bay at the lower layer.

4.3. Turbidity flux

During the middle layer intrusion event (September 15, 1998), oceanic water with low turbidity intruded into the middle layer in the bay, while highly turbid bay water flowed out into the ocean from the surface layer and lower layer, resulting in a seaward net suspended matter flux. On the other hand, during the estuarine circulation event (August 18-19, 1998), turbid bay surface water flowed out into the ocean, while turbid water also flowed into the bay in the lower layer. As a result, there was hardly any net export of suspended sediment from the bay into the ocean. The three-layer flow structure effectively transports suspended matter existing in the bay to the ocean by pushing out the turbid bay bottom and surface water toward the ocean. Ii is found that during the two- and three-layer events, the volume transport across the section was the same (Hinata et al., 2001), but the heat and suspended matter fluxes were quite different between these two events.

Recently, based on water quality measurements made in early September 1998, Fujiwara and Yamada (2002) showed that the hypoxic water mass generated in the bottom layer at the head of the bay was flush out seaward by the three-layer flow structure (Figure 8). Their result also demonstrates that the intermittent oceanic water intrusion in the middle layer has a significant impact on the bay environment.

5. CONCLUSIONS

This study demonstrates the importance of the oceanic circulation offshore from Tokyo Bay in controlling the fluxed of heat, salinity and suspended matter between the bay and the ocean. It is thus important to take into account these oceanographic processes when assessing the environmental status of Tokyo Bay.

Since it is not a predictable, periodic phenomenon, there are many unknowns regarding the frequency and duration of middle layer intrusions of oceanic water. Based on high-frequency oceanic radar observation, Hinata et al. (2005) demonstrate that the Kuroshio warm water intrudes into Sagami Bay due to the wind field fluctuations with a synoptic time scale of 8 – 11 days when the Kuroshio approaches Sagami Bay. In addition, it is well known that the passage of the Kuroshio frontal eddies induces the warm water intrusion into the southern coast of Japan with the synoptic time scale (Akiyama and Saitoh, 1990; Kimura and Sugimoto, 1993). These previous studies suggest that the Kuroshio fluctuations play a crucial role in the occurrence of the intermittent/periodic middle layer intrusion into Tokyo Bay. This intrusion appears to have a large impact on the water quality of Tokyo Bay in summer. In the future, detailed studies are needed to quantify, through field and modeling studies, the dynamics and the impact on water quality, on oceanic water inflows in Tokyo Bay.

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CHAPTER 6

INFLUENCE OF THE DEEP WATERWAY PROJECT ON THE CHANGJIANG ESTUARY

JIANRONG ZHU, PINGXING DING, LIQUAN ZHANG, HUI WU, AND HUIJIANG CAO

1. INTRODUCTION

In the estuarine environment, physical transports and processes play a key role in the transport of nutrients and pollutants, while biological and chemical processes determine the ecosystem community response to the stresses. As a result, human activities, such as large-scale estuarine and coastal engineering which changed local hydro-kinematics and damaged the habitats for the organisms, have a profound impact on the estuarine ecosystems. This is the case of the Changjiang Estuary, in China (Figure 1). The estuary is impacted by the deep waterway project. The objective of this project in the Changjiang Estuary was to improve the navigation capacity for Shanghai harbour, because there exited sand bars at the mouth and the minimum water depth was 7m. When the first and second phases of the project were completed, the water depth was increased to 8 and 10m, respectively. The third phase is in progress now and the water depth will be increased to 12.5m. The dredged channel is protected by dykes. The north and south dykes will be more than 40 km long (Figure 1), at an elevation of 0.33m above the mean sea level, and the width of the main deep waterway is 300m. This large-scale project will inevitably produce great impacts on the dynamic process in the Changjiang Estuary, starting by changes in salinity and estuarine dynamics.

The Changjiang Estuary has three bifurcations and four outlets to the sea (Figure 1). The patterns of saltwater intrusion in each outlet are different. The saltwater intrusion in the Changjiang Estuary was studied extensively from observation data (Han, 1983; Mao et al, 1995; Xu and Yuan, 1994; Shen et al., 1980, 2003; Mao, 1995; Yang and Zhu, 1993; Zhu, et al., 2003). The saltwater intrusion during winter is most severe in the North Branch due to its funnel shape and lower runoff, and from there the intruding saltwater could spill over into the South Branch during the spring

E. Wolanski (ed.), The Environment in Asia Pacific Harbours, 79–92.

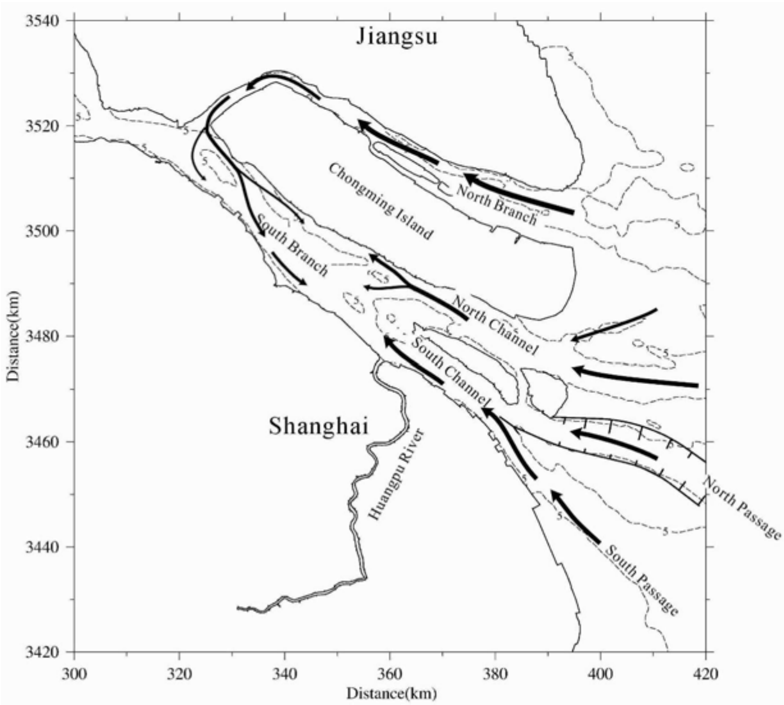


Figure 1. A conceptual diagram of saltwater intrusion (arrows) in the Changjiang Estuary following the second phase of the deep waterway project. The Changjiang Estuary consists of the North Branch, South Branch, North Channel, South Channel, North Passage and South Passage. Shanghai City is located on the two sides of the Huangpu River and near the Changjiang Estuary.

tide. From there this saline water mass moves downstream in the South Branch influencing the water quality in the South Branch during the following middle and neap tides

These studies also showed that the saltwater intrusion in the whole Changjiang Estuary was determined mainly by the Changjiang River discharge and tide range, and was also influenced by the wind stress, sea level fluctuations, the currents over the adjoining continental shelf, mixing, and human activities (such as the Three Gorges Dam, the South-North Water Diversion project, and the Deep Waterway Project). To predict these effects remain very difficult because the system is complex with three river bifurcations, many tidal flats, tidal flood and ebb channel even in the same bifurcation. Some efforts were made to model the estuary by applying a vertical averaged 2-D numerical model (Xiao and Zhu, 2000). However there remained two problems to solve. One problem was that when a vertical averaged 2-D numerical model was applied, the saltwater wedge could not been reconstructed. The second problem was that some important dynamic factors were not included in the models, such as the wind stress and the continental shelf current off the mouth.

In this chapter, an improved 3-D ECOM model has been applied to study the impact of the deep waterway project on the saltwater intrusion, and the degradation of biodiversity in the recent decades and the impact of the project on biodiversity has been quantified and analysed.

2. INFLUENCE OF THE PROJECT ON SALTWATER INTRUSION

2.1. Numerical model

The proper fitting of the coastlines is a critical issue for a successful simulation and prediction of currents in an estuary and coastal waters. For this reason, Chen et al. (2001) has developed a non-orthogonal coordinate transformation model based on the 3-D coastal ocean circulation model developed by Blumberg and Mellor (1987). This model incorporates the Mellor and Yamada level 2.5 turbulent closure scheme to provide a time and space dependent parameterization of vertical turbulent mixing (Mellor and Yamada, 1974, 1982; Galphin et al., 1988).

2.1.1. The primitive equations

Intruding the horizontal non-orthogonal curvilinear and vertical stretched sigma coordinate system, the governing equations of ocean circulation and water masses (consisting of momentum, continuity, salinity, and density equations) as follows:

$$\frac{\partial DJu_1}{\partial t}+\frac{\partial DJ\hat{U}u_1}{\partial \xi}+\frac{\partial DJ\hat{V}u_1}{\partial \eta}+\frac{\partial J\omega u_1}{\partial \sigma}-Dh_2\hat{V}[v_1\frac{\partial}{\partial \xi}(\frac{J}{h_1})-u_1\frac{\partial}{\partial \eta}(\frac{J}{h_2})+Jf]-Dh_2u_1v_1\frac{\partial}{\partial \xi}(\frac{h_3}{h_1h_2})$$
$$=-h_2gD\frac{\partial \zeta}{\partial \xi}+\frac{gh_2D}{\rho_o}\frac{\partial D}{\partial \xi}\int_\sigma^0\sigma\frac{\partial \rho}{\partial \sigma}d\sigma-\frac{gh_2D^2}{\rho_o}\frac{\partial}{\partial \xi}\int_\sigma^0\rho d\sigma+\frac{1}{D}\frac{\partial}{\partial \sigma}(K_m\frac{\partial Ju_1}{\partial \sigma})+DJF_x \tag{1}$$

$$\frac{\partial DJv_1}{\partial t}+\frac{\partial DJ\hat{U}v_1}{\partial \xi}+\frac{\partial DJ\hat{V}v_1}{\partial \eta}+\frac{\partial J\omega v_1}{\partial \sigma}+Dh_1\hat{U}[v_1\frac{\partial}{\partial \xi}(\frac{J}{h_1})-u_1\frac{\partial}{\partial \eta}(\frac{J}{h_2})+Jf]-Dh_1u_1v_1\frac{\partial}{\partial \eta}(\frac{h_3}{h_1h_2})$$
$$=-h_1gD\frac{\partial \zeta}{\partial \eta}+\frac{gh_1D}{\rho_o}\frac{\partial D}{\partial \eta}\int_\sigma^0\sigma\frac{\partial \rho}{\partial \sigma}d\sigma-\frac{gh_1D^2}{\rho_o}\frac{\partial}{\partial \eta}\int_\sigma^0\rho d\sigma+\frac{1}{D}\frac{\partial}{\partial \sigma}(K_m\frac{\partial Jv_1}{\partial \sigma})+DJF_y \tag{2}$$

$$\frac{\partial \zeta}{\partial t}+\frac{1}{J}[\frac{\partial}{\partial \xi}(DJ\hat{U})+\frac{\partial}{\partial \eta}(DJ\hat{V})]+\frac{\partial \omega}{\partial \sigma}=0 \tag{3}$$

$$\frac{\partial JDs}{\partial t}+\frac{\partial JD\hat{U}s}{\partial \xi}+\frac{\partial JD\hat{V}s}{\partial \eta}+\frac{\partial J\omega s}{\partial \sigma}=\frac{1}{D}\frac{\partial}{\partial \sigma}(K_h\frac{\partial Js}{\partial \sigma})+DJF_s \tag{4}$$

$$\rho_{total}=\rho_{total}(\theta,s) \tag{5}$$

where

$$\omega=w-\sigma(\hat{U}\frac{\partial D}{\partial \xi}+\hat{V}\frac{\partial D}{\partial \eta})-[(1+\sigma)\frac{\partial \zeta}{\partial t}+\hat{U}\frac{\partial \zeta}{\partial \xi}+\hat{V}\frac{\partial \zeta}{\partial \eta}] \tag{6}$$

In the above equations, the new coordinate system (ξ, η, σ) is defined as: $\xi = \xi(x, y), \eta = \eta(x, y)$, $\sigma = \frac{z-\zeta}{H+\zeta}$. The corresponding velocities are $u_1 = \frac{h_2}{J}(x_\xi u + y_\xi v), v_1 = \frac{h_1}{J}(x_\eta u + y_\eta v)$

The vertical coordinate σ varies from −1 at $z = -H$ to 0 at $z = \zeta$. In these equations, *x, y*, and *z* are the east, north, and vertical axes of the Cartesian coordinate; ζ is the water surface elevation; *H* is the total water depth;

$$\xi_x = \frac{y_\eta}{J}, \quad \xi_y = -\frac{x_\eta}{J}, \quad \eta_x = -\frac{y_\xi}{J}, \quad \eta_y = \frac{x_\xi}{J}$$

where, J is the Jacobin function in the form of

$$J = x_\xi y_\eta - x_\eta y_\xi$$

the subscript symbols (ξ and η) indicate derivatives. The metric factors h_1 and h_2 of the coordinate transformation are defined as

$$h_1 = \sqrt{x_\xi^2 + y_\xi^2}, h_2 = \sqrt{x_\eta^2 + y_\eta^2} \quad \hat{U} = \frac{1}{J}(h_2 u_1 - \frac{h_3}{h_1} v_1), \hat{V} = \frac{1}{J}(h_1 v_1 - \frac{h_3}{h_2} u_1)$$

in which, $h_3 = y_\xi y_\eta + x_\xi x_\eta$; θ the potential temperature; *s* the salinity; *f* the Coriolis parameter; g the gravitational acceleration; K_m the vertical eddy viscosity coefficient; and K_h the thermal vertical eddy friction coefficient. F_u, F_v and F_s represent the horizontal momentum and salt diffusion terms. ρ and ρ_o are the perturbation and reference density, which satisfy $\rho_{\text{total}} = \rho + \rho_o$. F_u, F_v and F_s are calculated by Smagorinsky's formula (1963) in which the horizontal diffusion is directly proportional to the product of horizontal grid sizes. K_m and K_h are calculated using the modified Mellor and Yamada (1974, 1982 and 1988) level 2.5 turbulent closure scheme.

2.1.2 Boundary and initial conditions

The model domain (Figure 2) includes the whole Changjiang Estuary and the adjoining inner shelf of the East China Sea. The non-orthogonal curvilinear grid was designed to fit the coastline and improve the local resolution (Figure 2). The highest horizontal resolution in the upper North Branch near the bifurcation and in the deep waterway is 50 m, and vertical grid was divided into 10 layers. The deep waterway in second phase is considered, and the minimum water depth in the main waterway is 10 m. These grids are small enough to resolve the small spatial variation of the topography. A wetting-drying method was added into the model to reproduce the moving boundary over the tidal flats.

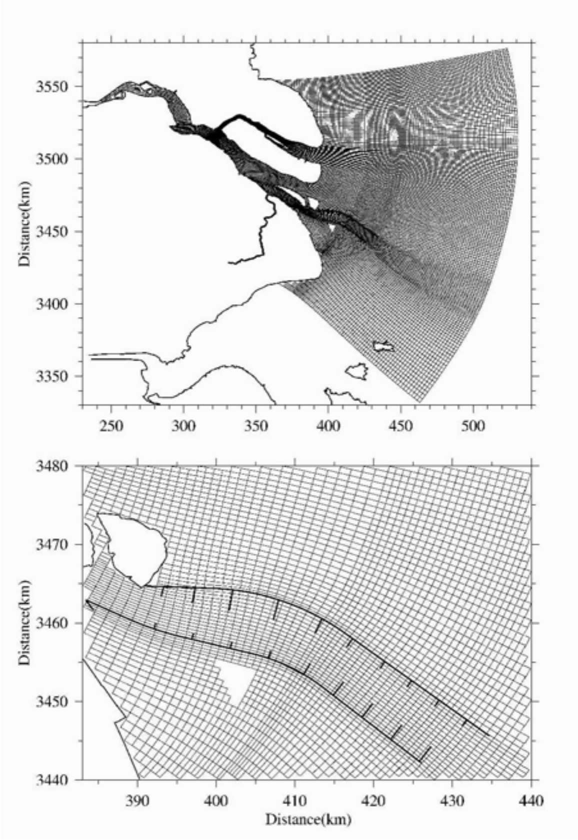


Figure 2. *Model domain and grids (upper), the amplified grids at the area of the deep waterway (lower).*

The surface and bottom boundary conditions for the momentum and heat equations are given by

$$\frac{\rho_0 K_m}{D}\left(\frac{\partial u_1}{\partial \sigma}, \frac{\partial v_1}{\partial \sigma}\right) = (\tau_{0\xi}, \tau_{0\eta}); \omega = 0, \qquad \text{at } \sigma=0$$

$$\frac{\rho_0 K_m}{D}\left(\frac{\partial u_1}{\partial \theta}, \frac{\partial v_1}{\partial \theta}\right) = (\tau_{b\xi}, \tau_{b\eta}); \omega = 0, \qquad \text{at } \sigma=-1$$

Where ($\tau_{o\xi}$, $\tau_{o\eta}$) and $(\tau_{b\xi}, \tau_{b\eta}) = C_d\sqrt{U^2+V^2}(U^2+V^2)$ are the ξ and η components of surface wind and bottom stresses. The surface wind stress was calculated based on the neutral steady state drag coefficient developed by Large and Pond (1981). The bottom drag coefficient C_d was determined by matching a logarithmic bottom layer to the model at a height z_{ab} above the bottom,

$$C_d = \max[k^2 / (\ln \frac{z_{ab}}{z_0})^2, 0.0025]$$

where k=0.4 is the Karman's constant and z_o is the bottom roughness parameter, which was taken as 0.01m in this study. The lateral boundary condition is specified as $v_n = 0$ where v_n is the normal velocity component at the boundary.

The initial condition of the velocity is $u_1 = v_1 = 0$ and $\zeta=0$.

The upstream open boundary was specified by the Changjiang River discharge (9100 m^3 s^{-1}) as observed in January 1999 at Datong station which is ~640 km upstream from the river mouth. The eight main tidal constituents, M_2, S_2, K_1, O_1, N_2, K_2, P_1, and Q_1 were considered. A residual water level is added to reproduce the continental currents at the open boundaries off the Changjiang mouth. The continental shelf currents off the Changjiang mouth are driven mainly the Taiwan Warm Current and Subei Coast Current. These currents can bring salt seawater just into the area off the Changjiang mouth, and become the source of the high salinity water that can intrude into the Changjiang Estuary (Shen, et al., 2003). The harmonic constants of the eight main tidal constituents and residual water level at the open boundaries were provided by the results of the large domain model of the East China Sea and Yellow Sea (Zhu, 2003). This model includes the effect of the monsoon in Eastern Asia and the strong northerly wind in winter that induces the landward Ekman transport. This transport facilitates saltwater intrusion in the estuary. Data fro the wind stress in January and February were obtained from the Editorial Board for Marine Atlas (Climatology, 1992). The initial salinity was taken from several observation data inner the mouth and from the Marine Atlas (Hydrology, 1992) off the mouth. The temperature variation was not considered and the temperature set to be 4°C. The model was run with and without the deep waterway, and the outputs were compared to estimate the impact of the project on the saltwater intrusion.

2.2. Results and discussion

Figure 3 shows the predicted horizontal distribution of surface salinity during a middle tide following a spring tide, for both before and after the deep waterway project. Chongming Island was completely surrounded by water with salinity greater than 0.5, which was undrinkable. The saltwater intrusion in the South Channel was influenced by two sources of salt water, namely oceanic water downstream and saline water spilling in from the North Branch upstream. This saltwater spill from the North Branch was considerable in January and February 1999, and move downstream while oscillating upstream and downstream with the flood and ebb tide current. The model reproduced the dynamic process of the saltwater intrusion successfully (Animation 1). The North Branch had water with salinity greater than 15 due to its funnel shape and the low volume of river discharge. The saltwater intrusion was stronger on the north side of the channel than on the south side, an effect that is due to the Coriolis force. The saltwater intrusion near the bottom was stronger than near the surface, and was stronger during spring tides than during neap tides (not shown).

Compared with the situation before the project, the saltwater intrusion after the deep waterway project had been alleviated distinctly in the North Channel. This is bbecause the dykes blocked the southward drift of freshwater by the northerly monsoon and Coriolis force, and made the salinity decrease near the mouth of the North Channel. In the North Passage, the project had increased the water depth. On the one hand, it could increase the Changjiang discharge flowing into the passage for the bifurcation estuary due to the increase of the volume, which could reduce the saltwater intrusion in the North Passage. On the other hand, it could also increase the baroclinic circulation and enhance the saltwater intrusion. The impact of the project on the saltwater intrusion depended on their combined effects. The saltwater intrusion at the upper section of the project was clearly increased, while at the lower section the salinity intrusion was measurably decreased.

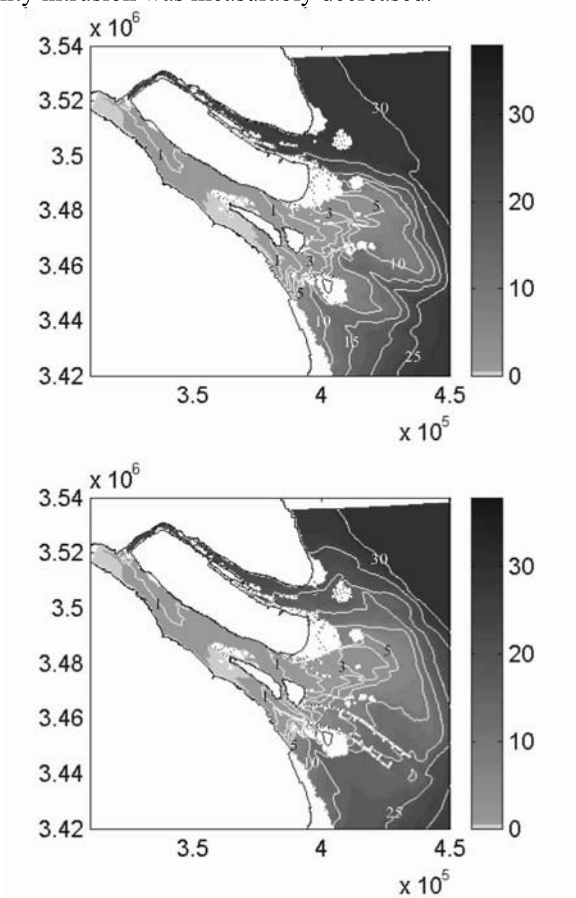


Figure 3. *Distribution of surface salinity in middle tide after spring tide (a) before and (b) after the deep waterway project.*

Figure 4 shows the location of the three main transects S1, S2, and S3, and a number of sites discussed below. The saltwater intruded as a salt wedge; the salinity was much higher at the bottom than at the surface (Figure 5). Comparing the salinity distributions before and after the deep waterway project, the saltwater intrusion decreased in the North Channel, especially along the eastern side (Figure 5a1 and a2).

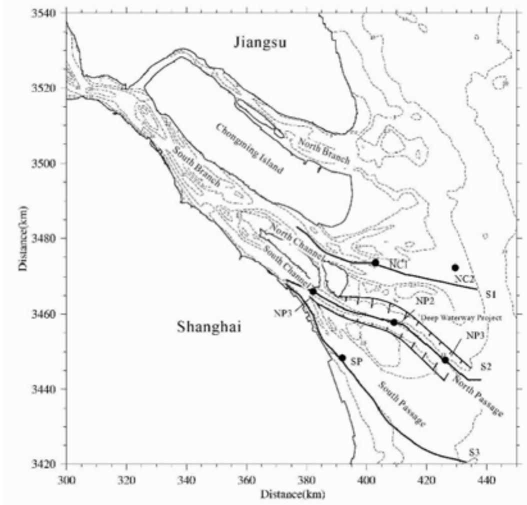
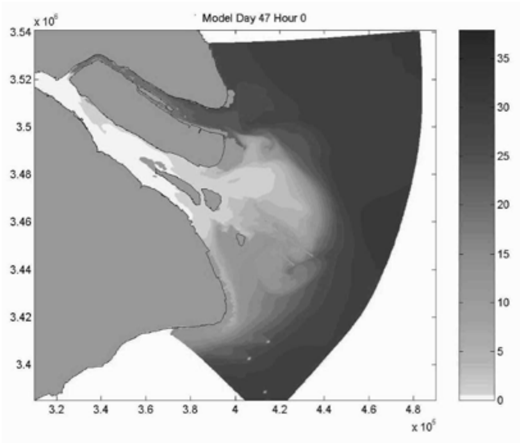


Figure 4. *The model results output sites and the three longitudinal sections S1, S2 and S3.*



Animation 1. *Dynamic process of the saltwater intrusion in the Changjiang Estuary.*

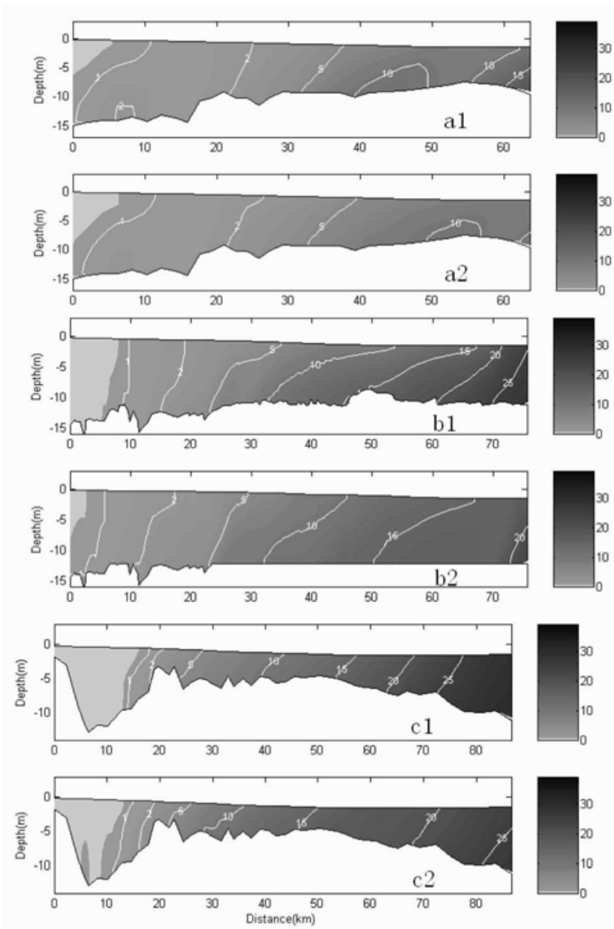


Figure 5. *Distribution of salinity along the longitudinal section S1 (a), S2(b) and S3(c) (shown in Figure 4) during a middle tide after a spring tide before (labelled 1) and after (labelled 2) the deep waterway project.*

In the North Passage, the saltwater intrusion increased at the upper section of the project and decreased at the lower section (Figure 5b1 and b2). In the South Passage, the saltwater intrusion increased at the western side and decreased at the eastern side (Figure 5 c1 and c2). From the longitudinal salinity distribution, it is apparent that there existed vertical salinity variations and a saltwater wedge. Clearly, the three dimension numerical model was necessary to simulate the saltwater intrusion in this estuary.

There are also large temporal salinity variations (Figure 6). This figure also shows that the salinity at the sites NC1 and NC2, located in the sand bar area in the North Channel, decreased after the project. The salinity at site NP1 and the upper section of the project site in the North Passage increased after the project. At the middle section of the project site, the salinity was generally reduced. At the lower section, the salinity was significantly reduced. The salinity at site SP in the South Passage increased after the project.

3. INFLUENCE OF THE PROJECT ON ECOSYSTEMS

3.1 Decrease in species number

According to the ecological survey from October, 1997 to May, 2002 by the Environmental Monitoring Center of East China Sea (Wang, et al., 2004), 138 genera and 402 species of phytoplankton, 19 types and 480 species of zooplankton, and 97 species of benthos were recorded at the Changjiang Estuary (Table 1). However, comparing this inventory with that twenty years earlier, it is clear that the number of species number of plankton and benthos has decreased dramatically, that the community composition of plankton and benthos is now much simpler, and the biodiversity is degraded. The biomass of dominant phytoplankton has increased and tended to form a bloom in some cases, while the biomass of benthos has dramatically decreased.

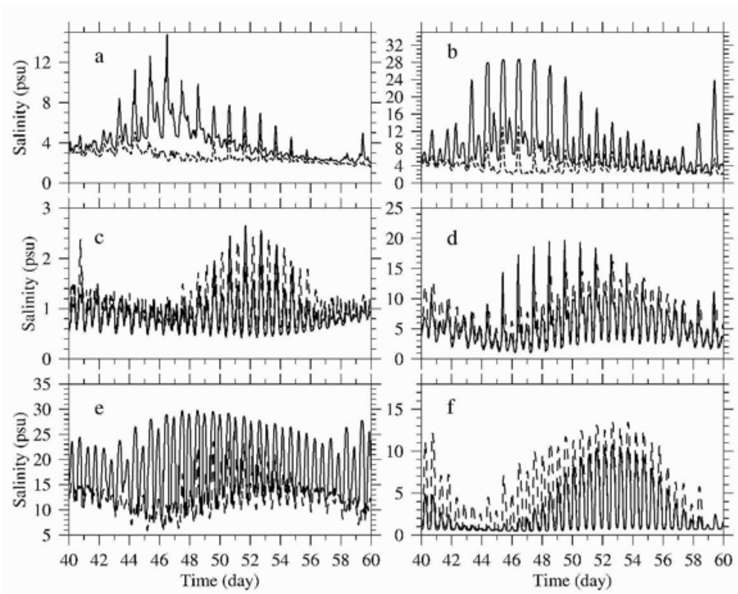


Figure 6. *Temporal surface salinity variation at sites NC1(a), NC2(b) (shown in Figure 4) in the North Channel, at sites NP1(c), NP2(d), NP3(e) in the North Passage, and at site SP(f) in the South Passage. Solid line: before the project; dashed line: after the project.*

The species numbers of phytoplankton, zooplankton and benthos in the Changjiang Estuary and the brackish waters influenced by the runoff of the Changjiang River decreased sharply as compared with 20 years ago. There were 97 species of phytoplankton in the Changjiang Estuary in the early 1990s and 63 species in the late 1990s, while the species number decreased to 15 - 50 in the early 21st century. The decrease in species number occurred mainly among the species of *Diatomophyceae*, accordingly the species proportion of *Dinophyceae* increased slightly. The change in the number of phytoplankton species in the brackish waters in the upper estuary showed the same trend as in the lower estuary, while the size of the decrease was smaller.

The number of zooplankton species in the Changjiang Estuary was 105 in the early 1980s, 76 in the early 1990s, and 25 in the early 21st century. The species number of zooplankton in the brackish waters was 81 in the early 1980s, 33 in the late 1990s, and 35 in the early 21st century, which was about 41 percent of that in the early 1980s.

The benthos in the subtidal zone was fairly rich. According to the Pollution Survey of East Sea during 1978-1979 (Ye et al., 2004), the species number of benthos in the Changjiang Estuary was 52 in 1978, and this dropped sharply to below 20 in the early 21st century. The species number of benthos in the brackish waters was 81 in 1978 and declined remarkably to below 15 in the early 21st century. Both showed a distinct decrease after the late1990s.

Table 1. *The biodiversity of plankton and benthos at the Changjiang Estuary (year 1997-2002) (redrawn from Wang, et al., 2004).*

Type	*Number of genus and species*		*Density or biomass*
Phzytoplankton	Diatomophyceae 69 genera 259 species		
138 genera	Dinophyceae 17 genera 53 species		
402 species	Chlorophyceae 15 genera 36 species		2.6 x 10^6 No./m3
	Cyanophyceae14 genera 22 species		(low water)
	Chrysophyceae 5 genera 6 species		
	Euglenophyta 3 genera 5 species		2.96 x 10^7 No./m3 (high
	Haptophyceae 8 genera 11 species		water)
	Prasinophyceae 2 genera 3 species		
	Raphidophyceae 2 genera 2 species		
	Xanthophyta 1 genera 2 species		
	Cryptophyta l genera 2 species		
Zooplankton	Copepoda 177 species	Planktonic Larva 63 species	
19 types	Amphipoda 40 species	Chaetognatha 28 species	39 mg m^{-3}
480 species	Pelagic polychaeta 24 species	Pelagi gastropod 22 species	(low water)
	Ostracoda 20 species	Hydromedusae 18 species	292.7 mg m^{-3}
	Euphausiacea 18 species	Mysidacea 12 species	(high water)
	Ciliate 10 species	Cladocera 9 species	
	Tunicata 9 species	Other type 7 species	
	Siphonophores 6 species	Ctenophora 3 species	
	Cumacea 3 species	Decpoda 10 species	
	Isopoda 1 species		
Benthos	Crustacea 17 species		15.5 g/ m^{-2}
97 species	Mollusca 23 species		(low water)
	polychaeta 48 species		8.93 g m^{-2}
	Echinodermata 7 species		(high water)

3.2 Remarkable changes in species composition and biomass

In the last twenty years, remarkable changes have occurred in the species composition of the organisms in the Changjiang Estuary. Species sensitive to environmental pollution have decreased, while the pollution tolerant species have increased, and these are becoming the dominant species. The density of phytoplankton and biomass of zooplankton in the middle 1990s and the early 21st century showed a remarkable increase as compared with the 1980s, while the biomass of benthos decreased continuously since the 1980s (not shown).

3.3 Simplification of the community structure and degradation in biodiversity

The community structure of phytoplankton in the Changjiang Estuary is becoming unstable, the species composition simplified, the dominance of few species more distinct, the distribution of individual quantity remarkably uneven, and few dominant species such as *Skeletonme costatum* tended to form a bloom when the environmental condition became suitable. All of these observations indicate that the environment and habitats of the Changjiang Estuary are becoming degraded.

The community structure of phytoplankton showed an obvious seasonal succession in the 1980s, with the dominant species of *Diatomophyceae*, such as *Rhizosolenia robusta*, *Norm melosira* and *Skeletonme costatum*. From the 1990s to 21st century, the dominant species was mainly *Skeletonme costatum*, which accounted for more than 80% of the sum of phytoplankton. The community structure of zooplankton was relatively stable and became simplified, with the dominant species of mainly *Polychaeta*. The community structure of benthos has since changed dramatically; indeed almost no benthos could be found in many areas of the estuary.

The phytoplankton in the brackish waters showed a more predominant community structure than that in the Changjiang Estuary, with *Nitzschia hassall*, *Coscinodiscus ehrenberg* and *Skeletonme costatum* et al. as the dominant species. However, the frequency and scale of blooms has increased since the 1990s.

3.4 The impacts of the deep waterway project on biodiversity

The factors influenced the biodiversity in the Changjiang Estuary include natural factors such as the fluctuation in the quantity of runoff, storm tides, upwelling and blooms, and the impacts induced by human activities such as over-fishing, land reclamation, oil spills, sewage discharge, large-scale hydraulic and waterway regulation projects, including the deep waterway project. Among these, the impacts of human activities have had a significant impact on the biodiversity in the estuarine ecosystems.

The human activities have changed the pattern of saltwater intrusion as mentioned in the above sections. The deep waterway project has greatly changed the local hydro-kinematics conditions and the characteristics of silt deposition, and caused severe habitat fragmentation. The variation in salinity has had a great influence on the species composition of benthos. The disturbances from dredging have seriously damaged the habitats for the organisms especially for the benthos, and this is reflected in the dramatic decrease in the species number and biomass, the simplification in community structure, and the degradation in biodiversity since the late of 1990s. Therefore, the long-term monitoring of the impacts of the deep waterway project on the estuarine ecosystems and biodiversity is necessary and ecological restoration for these damaged estuarine ecosystems is essential.

4. SUMMARY

Using the improved 3-D ECOM model with a high resolution grid and incorporating various dynamic factors, the impact of the deep waterway project on the saltwater intrusion in the Changjiang Estuary was analysed. Salinity in the South Branch is shown to be influenced by saltwater intrusion downstream and from spillage of brackish water from the North Branch upstream. In the North Channel, the saltwater intrusion had been alleviated distinctly after the deep waterway project, because the dykes of the project blocked off the southward drift of the brackish water plumes under forcing by the northerly monsoon and the Coriolis force. The saltwater intrusion in the project area was intensified at the upper section and alleviated at the lower section. In the South passage, the saltwater intrusion was intensified as the background salinity increased and the river discharge decreased. The deep waterway project had an obvious impact on saltwater intrusion in the Changjiang Estuary.

The impacts of the deep waterway project on planktons and benthos have been quantified and analyzed. The species number and biomass of planktons and benthos have decreased dramatically as compared with 20 years ago. The degradation of biodiversity at the Changjiang Estuary appears due to the cumulative impacts of eutrophication caused by human activities such as agriculture and sewage discharge, as well as land reclamation and estuarine engineering projects.

5. ACKNOWLEDGEMENTS

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CHAPTER 7

IMPACT OF HUMAN ACTIVITIES ON THE HEALTH OF ECOSYSTEMS IN THE CHANGJIANG DELTA REGION

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AND YING WU

1. INTRODUCTION

Most of the world mega-cities are located within the distance of 100-200 km from the coast, and the delta region is the place where the modern civilization developed in human history. In the mainland of China, the eleven provinces in the coastal region represents ca. 15 % of the national land surface area, but sustain 40 % of national population with 60 % of total GDP.

Coastal environment represents one of the most dynamic habitats on the earth, i.e. land – ocean interface, and supports some of the most diverse and productive ecosystems (LOICZ: www.loicz.org). Vulnerable coast environments have been disturbed by anthropogenic activities, particularly owing to the rapid industrialization and urbanization in the 20th Century (Kim et al., 1999; Bouloubassi et al., 2001). Land-based activities over the drainage basin (i.e. land clearing, damming, harbor construction and land reclamation) have a profound impact on the coastal environment by changing the fluvial fluxes of water and sediments, perturbation by navigation and engineering construction (e.g. tunnel and bridge), and loss of delta habitat, for example, the Changjiang (Zhang et al., 1999). Moreover, the projected future climate changes will affect the physical, sedimentary and ecological processes in coastal environment, which can be considered an added stress on already overstressed ecosystems; this will further reduce the ability of coastal systems to provide goods and services. For instance, the loss of habitat by coastal erosion, changes in biogeochemical processes by altering nutrient fluxes, and shifts in adjacent marine biological community structure by pollution as well as overfishing, have all been identified as major, on-going impacts. Their occurrence, particularly in areas where rapid economic innovation and population growth take place like China, can be severe (World Resources Institute: www.wri.org).

E. Wolanski (ed.), The Environment in Asia Pacific Harbours, 93–111.

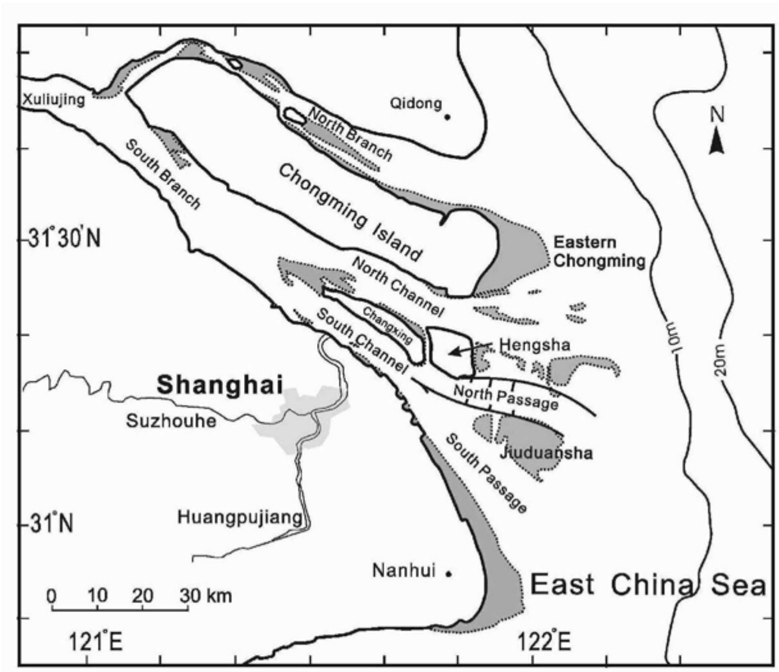

Figure 1. *Study area of the Changjiang Delta Region, which shows the two tributaries, i.e. Huangpujiang and Suzhouhe, draining into the Changjiang Estuary after flowing through Shanghai. The shadowed areas in the figure illustrate the inter-tidal zones of delta region. The Shanghai Port together with ship yards are along the Huangpujiang from its confluence with Changjiang upstream over a distance of ca. 30 km, which will be moved to Changxing and Hengsha in the main stream before the 2010 World Exhibition at Shanghai.*

Prolonged and intensive use of inorganic fertilizer in agriculture, changes in land use patterns, deforestation, and discharge of industrial and municipal wastes have all contributed to the eutrophication of coastal waters on a global scale. The estuarine and coastal regions showing such degradation include the North Sea, Baltic Sea, Adriatic Sea, and North America (De Jonge et al., 1994; Richardson and Heilmann, 1995). Over last 50 years, flux of natural and synthetic materials from the terrestrial sources to the coastal environment has increased by 1.5-2 folds (Meybeck & Ragu, 1995).

Salt marshes are an important sink for nutrients and anthropogenic substances, such as heavy metals in the continuum from land to the ocean (Gambrell, 1994; Callaway et al., 1998; Olivie-Lauquet et al., 2001). The impact of heavy metals and synthetic organic pollutants discharged into salt marshes can be substantial and have potential threat to the health of aquatic food-web (Turner, 1990; Wright & Mason, 1999). Pollutants (e.g. heavy metals) accumulated in the coastal sediments can be remobilized and removed to the deep-water area, impacting on the health of ecosystems (Cundy et al., 1997). Requirement for the protection of coastal environment has promoted the monitoring of pollutants in anthropogenically disturbed ecosystems, particularly harbours and delta regions (Wises et al., 1995; Wright & Mason, 1999; Olivie-Lauquet et al., 2001).

In this study, the status of the Changjiang (Yangtze River) Delta Region is examined (Figure 1). The emphasis is given to the anthropogenic impact on the health of the ecosystem in the delta and adjacent coastal waters, along with the progress of urbanization of Shanghai.

2. CIVILIZATION AND SHANGHAI HARBOR

The early development of civilization in Shanghai can be traced back to the 751 AD in the Tang Dynasty. In the early 1950s, the population of Shanghai was 5.72 million (Figure 2). Shanghai's population increased rapidly to reach over 10 million in 1960, by expansion into urban areas. In 2003, the population of Shanghai reached up to 13.4 million. Concurrently, the proportion of agricultural population in 1950s was 5-10 %, indicating the industrial and commercial character of the city; the population of farming inhabitants was increased to ca. 40 % in 1970-1980 owing to the incorporation of rural regions into the administration of the Shanghai (Figure 2). The agricultural population drops after 1980s following the innovation of economics nation wide and reaches 20 % in the new millennium.

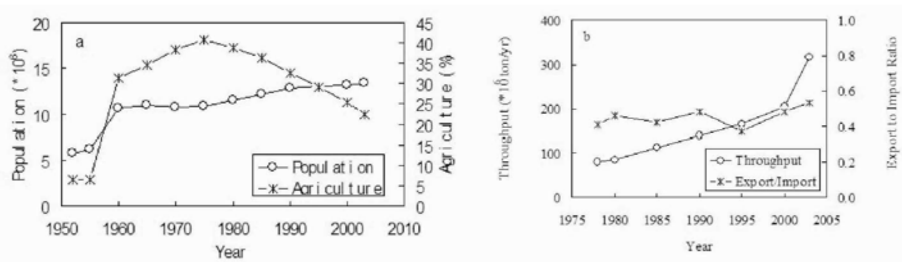

Figure 2. Evolution of Shanghai in the second of 20th century, with (a) change in population for the city and proportion of habitants in agriculture, and (b) increase in the throughput of the harbor after 1975 with the ratio of export to import values (Data Source: www.shtong.gov.cn).

Cargo throughput data for Shanghai harbour were first recorded in the Ming Dynasty, 500-600 years ago, when the annual throughput was ca. 200×10^3 tons, of which about half was for agricultural products and food-stuff transportation (Data Source: www.shtong.gov.cn). In the late 1970s, this throughput amounted to 80×10^6 tons, with the export to import ratio of 0.41 and 30×10^3 of TEU (i.e. standard container). In 2003, the cargo throughput was ca. 320×10^6 tons, with 11.3×10^6 for TEU and an export to import ratio of 0.53 (Figure 2).

3. CHANGES IN RIVERINE FLUXES

The Changjiang carries 1×10^{12} m^3 yr^{-1} of fresh water and 0.4×10^9 tons yr^{-1} (i.e. average from 1951-2004) of terrestrial sediments to the East China Sea. In Figure 3 is shown the DIN (i.e. dissolved inorganic nitrogen) concentration ($DIN=NO_3^-+NO_2^-+NH_4^+$) measured in the main channel at 180 km inland from the river mouth.

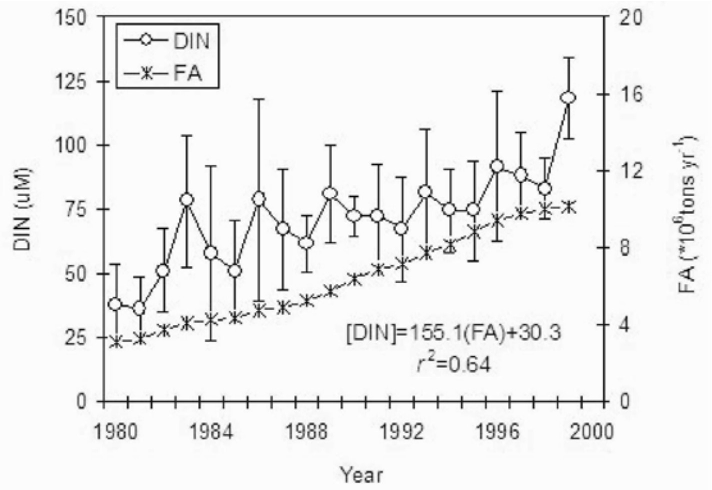

Figure 3. *Change in river composition of nutrients in the Changjiang Estuary, with DIN (i.e.* $DIN=NO_3^-+NO_2^-+NH_4^+$*) measured at Nantong (180 km inland from the river mouth) and the application of chemical fertilizers (FA) in the Changjiang Drainage Basin after 1980 (Data Source: Zhang, 2002).*

The concentrations of DIN increased from 40-50 μM in early 1980s to ca. 120 μM by 2000, that is, at a rate of increase by 2-3 μM yr^{-1} over last two decades (Figure 3). In the mean time, the application of chemical fertilizers in agriculture over the Changjiang Watersheds increased by 3-4 folds (Figure 3); 60 % of the fertilizers are composed of nitrogen compounds, with 20 % for phosphorus-containing chemicals (Zhang, 2002). In the lower reaches of Changjiang, the DIN to DIP (i.e. dissolved inorganic phosphorus, PO_4^{3-}) ratio in the main stream was ca. 50 in 1980s and 100-150 in 2000; this induces the photosynthesis in the estuarine and coastal waters turning to be phosphorus limited (Zhang, 2002). The concentration of nutrients in tributaries of the Changjiang is even much larger when flowing through the urban areas due to sewage discharges. For example, in the Huangpujiang and Suzhouhe, which flow through the Shanghai, the concentrations of NH_4^+ and PO_4^{3-} can be up to 450 μM and 25-30 μM, respectively (Zhu et al., 2004).

In addition, the Changjiang River carries also other pollutants to the East China Sea. For instance, the COD (i.e. chemical oxygen demand) load of the river is 2-8×10^6 tons yr^{-1} in 2002-2004; also oil pollutants amount to 20-70×10^3 tons yr^{-1} (Table 1). Concentrations of oil pollutants in the Changjiang Estuary can be up to 30-60 μg l^{-1} (State Oceanic Administration, 2004).

The ∑-PCBs in the surface sediments from the Changjiang Delta range from 0.2 ng g^{-1} to 20 ng g^{-1} (Liu et al., 2003). Low concentrations (e.g. <1.0 ng g^{-1}) of ∑-PCBs can be found in the rural area along the coast, while elevated levels (i.e. 5-20 ng g^{-1}) occur in the main channel and near the Shanghai. The concentrations of DDT in surface sediments are 0.5-3.5 ng g^{-1}, showing also higher values in the region affected by urban waste discharges (Chen et al., 2002). Sediment core samples collected from the adjacent coastal region show a considerable variation of deposition rate, i.e. 1-4 cm yr^{-1}, depending on the location of stations. Sediment cores in the delta front and the pro-delta region reveals a minimum concentration of HCH and DDT before 1950s; the concentrations of HCH and DDT increased to reach a maximum in the 1970s; they decreased thereafter (Chen et al., 2002). This implies that the quality of coastal environments has been recovered in recent decades after the national ban issued in 1980s on the use of some of organic pesticides in agriculture (e.g. HCHs and DDTs), although trace amount of these kind of organic pollutants are still reported from sediments in coastal environments (Wu et al., 1999).

Table 1. *Change in discharge of pollutants from the Changjiang to the East China Sea. (Adapted from State Oceanic Administration, 2002-2004).*

Year	COD ($\times 10^6$ tons yr^{-1})	Oil pollution ($\times 10^3$ tons yr^{-1})
2002	2.48	49.6
2003	2.72	69.9
2004	7.80	24.9

4. CHANGE IN HABITATS OF THE DELTA REGION

In the Changjiang Delta, salt marshes typical of a temporal climate zone are well developed; however, these have been increasingly reclaimed in past five decades. The total area of wetlands in the Changjiang Delta is estimated at about 1550 km^2, with an elevation of -6 to 5 m. More than one-third of the inter-tidal wetland is colonized by marsh vegetation, including, for example, *Phragmites* and *Scripus*, providing crucial habitats for a wide variety of widlife, e.g. migratory birds and water fowls (Shi et al., 2001). Previous studies in this region revealed the rapid adjustment of the coastal area after the reclamation, following sedimentation and biomass allocation (Yang, 1998; Sun et al., 2002). A rapid interaction between the estuary and salt marsh in terms of exchange of sediment, leading to erosion at one site and accumulation in another site, has also been noted (Hutchinson and Yu, 1998; Xu et al., 2001).

With the rapid increase in the population in coastal areas of China, the reclamation of tidal wetlands has become a common solution to provide land for the human settlement. Nearly 1000 km^2 of wetland has been reclaimed in the Changjiang Delta Region since the 1950s, which is 65 % of the whole wetland area of delta region and about twice the current total area of inter-tidal zone. For instance, more than half of Chongming Island is derived from reclamation in the period 1940-2003, and newly formed wetlands in the delta region, e.g. Hengsha and Jiuduansha show also rapid changed in surface areas, accelerated by the human activities (Figure 4).

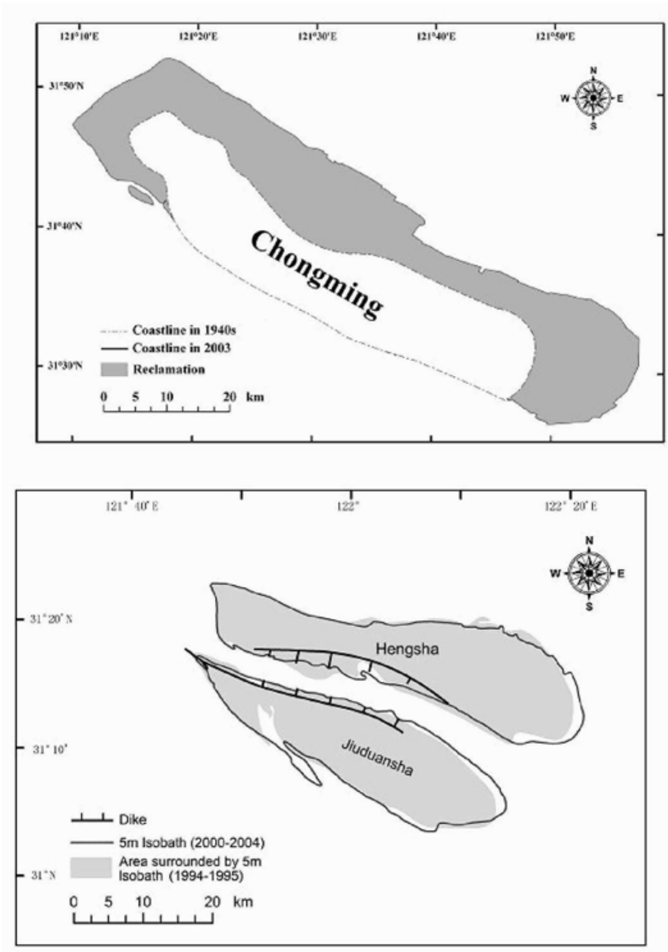

Figure 4. *Anthropogenic perturbations of the wetlands of the Changjiang estuary resulted in the expansion of Chongming Island due to the reclamation of tidal areas since the 1940s (upper panel), and effect of dikes constructed in 1999 on the 5 m isobath in the Hengsha and Jiuduansha shoals (lower panel).*

In the growing seasons, *Scirpus*, the pioneer plant, and *Phragmites* colonize the upper part (i.e. high marsh) of the inter-tidal zone. In the period of 1950-1990, several seawalls were constructed in salt marshes. Although most of the original vegetation of marsh areas (e.g. reeds) was destroyed, the *Scirpus* marsh and the bare mud flats were preserved. In recent years, a seawall was constructed at 0 to -2 m elevation. For example, in the reclamation of Nanhui, the southeast part of the Changjiang Delta Region, a seawall was built at the mean low tide line in 1999 (unpublished data).

The annual sediment load of the Changjiang decreased from 0.5×10^9 tons yr^{-1} in 1950s to $0.3\text{-}0.4\times10^9$ tons yr^{-1} in the 1990s. As a result, the delta responded by decreasing its rate of progradation by 60-65 %, while the vertical accretion rate decreased from 50-60 mm yr^{-1} in 1960-1980 to 10-15 mm yr^{-1} in the 1990s (Yang et al., 2002). In Figure 5 are shown two typical profiles of delta accretion. These data point to rapid accretion of the delta front in the period 1958-1978, stability in 1978-1998, and some recent erosion in the lower reaches of this region. This implies that the sub-aqueous delta front is sensitive to changes in fluvial sediment flux; however the delta accretion is not simply proportional to the fluvial sediment flux (Yang et al., 2003).

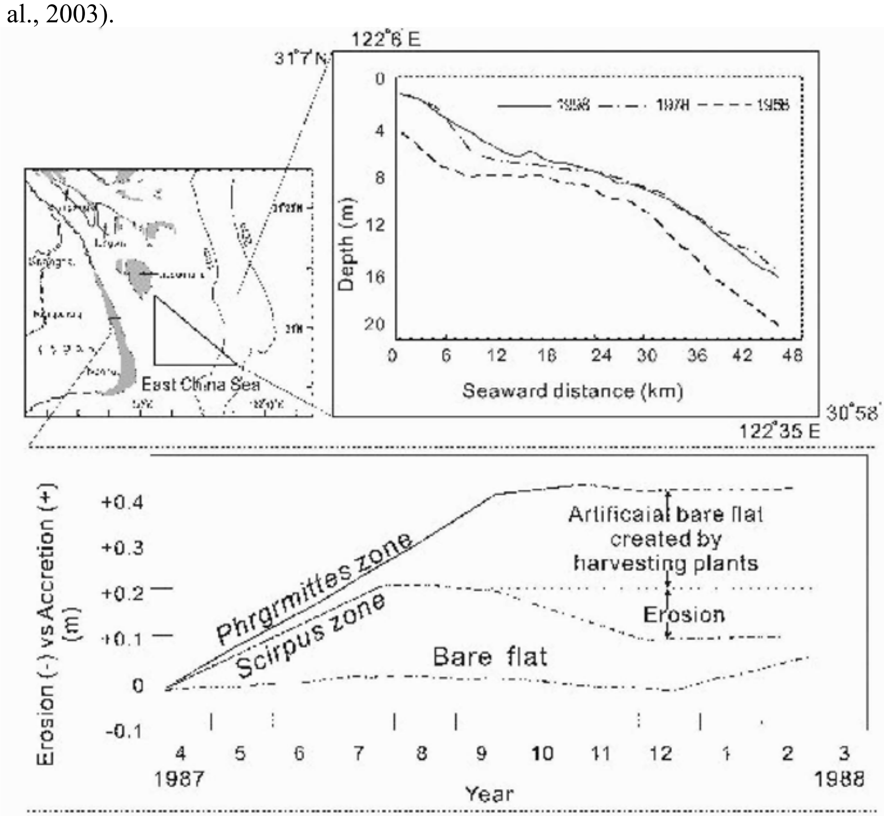

Figure 5. *Map of the Changjiang Delta Region with the major inter-tidal zone as shaded areas (upper left), change in evolution of delta front profiles after 1958 (upper right, and effect of vegetation on the change in salt marsh elevation (bottom).*

The human impact on the wetlands can be deduced from a study of erosion and accretion processes in salt marshes and adjacent bare mudflats. During the growth seasons of spring and summer, the marsh surface is rapidly covered by vegetation (e.g. *Phragmites* and *Scirpus*), which induces a rapid vertical accretion of the marsh area by up to 30-40 cm, while the adjacent mudflat remains relatively stabile. *Phragmites* vegetation is more efficient at trapping sediment than is *Scirpus* vegetation; hence the marsh surface colonized by *Phragmites* accretes almost twice as fast as the area with *Scirpus* cover (Figure 5). When the plant vegetation of the salt marsh is harvested, the surface area becomes a "bare mudflat" and is readily eroded. Hence the accretion of delta region is then stopped and replaced by erosion, and marsh retreats. In winter, the natural bare mudflats accrete considerably by storm-induced sediment deposition; the harvested marsh areas remain largely unchanged, because they are seldom submerged by the tides with elevation being higher than the high tide level (Figure 5).

The Changjiang Estuary has a large block-bar and/or shoal at river mouth that impedes navigation. To improve navigation, a deep channel was dredged and two jetties were constructed in the North Passage in 1998, and later two jetties were constructed to narrow the navigation channel and increase the water depth by dredging. After the construction of these jetties, a significant deposition of sediments occurred in the areas sheltered by these dikes. For instance, the transverse creeks deeper than 5 m disappeared due to sediment depositions at Hengsha and Jiuduansha (Figure 4). For instance, at the Jiuduansha the progradation rate of the area at 0-5 m depth greatly increased since the construction of the dikes (Table 2).

Table 2. *Progradation rate (km^2 yr^{-1}) of area at <5 m isobaths in the Jiuduansha area of the Changjiang Delta.*

Period	Progradation rate (km^2 yr^{-1})
1989-1995	5.5
1995-1999	2.4
1999-2000	10.1
2000-2004	1.0

5. CHEMICAL COMPOSITION OF DELTA SEDIMENTS

Across the tidal flat of Chongming Island, surface sediment fine landward, for instance, the medium size is ca. 20 μm in the high marshes and 60 μm in the bare mudflats in both the wet and dry seasons (Kang et al., 2003). The 4-63 μm size fraction dominates the sediment distribution, being 80 % in the high marshes and ca. 40 % in the bare mudflats.

The concentration of heavy metals (e.g. Co, Cr, Cu, Fe, Mn, Ni, Pb, V and Zn) decreases by a factor 2-4 from the high marsh to the bare mudflat, heavy metals, even after the normalization of absolute values to Al (Figure 6). The total organic carbon (TOC) in the high marsh (ca. 0.8 %) is four times higher than that in the bare mudflat (ca. 0.1-0.2 %), and shows a positive correlations with metal composition

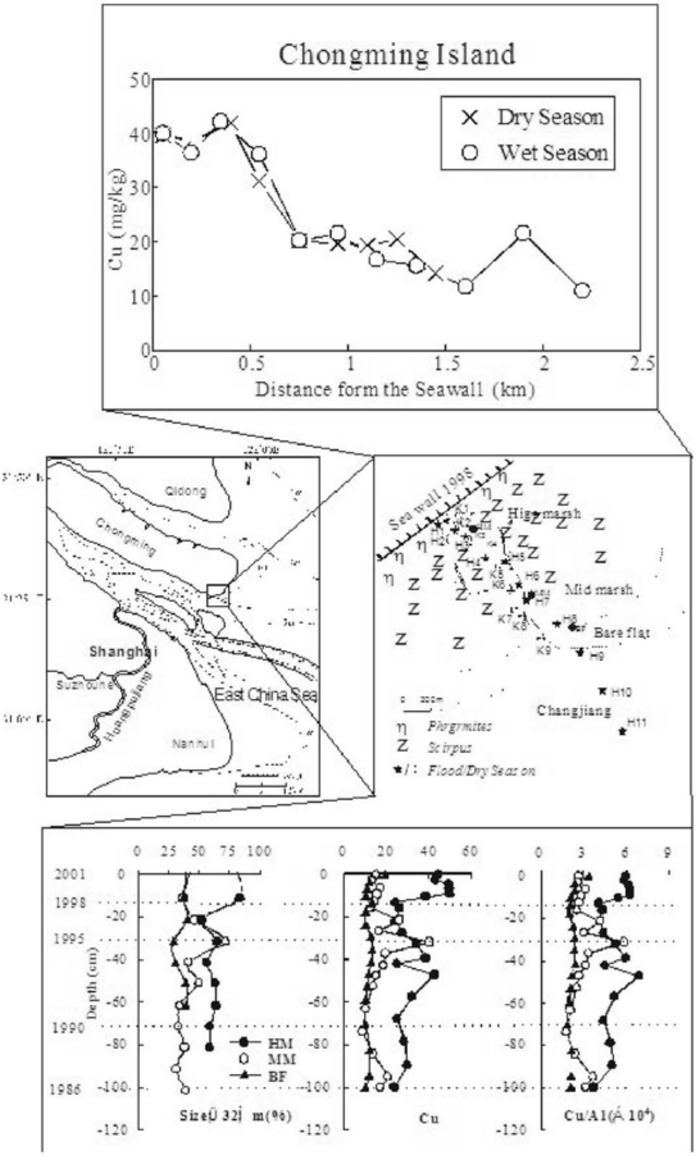

Figure 6. *Map of the Changjiang Delta Region showing the field measurements at the salt-marsh area of the Chongming Island. It shows the profiles of fine sediment (<32 μm), trace metals (e.g. Cu) and Cu/Al ratio from core samples from high marsh (HM), mid-marsh (MM) and bare-flat (BF) in the lower panel, and metal concentrations in surface sediments in a cross-section from high marsh to bare mudflats in wet and dry seasons in the upper panel, respectively.*

(Kang et al., 2003). The concentration of total organic carbon in core samples of salt marsh increases with higher content of clay component of sediments.

Sediment cores collected from high and low marshes and bare mudflat from Chongming Island show that the concentrations of trace metals increase with higher fine sediment fractions, e.g. size fraction of ≤32 μm, indicating the dominance of active sedimentary dynamics (e.g. fine sediment exchange between wetland and water channel) on the metal distribution (Kang et al., 2003). The metal concentrations in the bare mudflats are comparable to that in the suspended materials, showing dominance of terrestrial source in the delta region. There has been no significant difference in the chemical compositions for sediment samples from the Changjiang Estuary in last two decades (Zhang, 1999).

The sediment cores collected from the salt marsh in the Changjiang Delta Region show a wide range of deposition rates, up to 0.1-0.2 m yr^{-1}, depending upon the position of samples. The ^{210}Pb profiles in core sediments from salt marsh indicate rapid adjustment of sediment accumulations following anthropogenic perturbations. For instance, the seawall constructed in 1998 on Chongming Island has induced change in sediment accumulation rate in the high marsh from 8-10 cm yr^{-1} to 4-5 cm yr^{-1} soon after construction. Also as a result of the seawall construction, there was a higher component of fine sediment; as a result the metal concentrations increased, while in the deeper part of the sediment cores the metal concentrations were more stable and within the range of chemical compositions for the water column of Changjiang Estuary (Figure 6). Again, the reclamation of wetland in the delta region of Changjiang dramatically reduces the sediment delivery from the water channel to the marsh area, the limited supply of sediment particles in the high marsh can be trapped by the vegetation, such as *Phragmites* and *Scirpus*, particularly in spring tides of summer and fall (Yang et al., 2003), which can induce the higher metal concentrations as compared to the bare flat where the sediment size composition usually have more abundant coarse mineral particles.

The concentrations of heavy metals in delta sediments are higher in the area off the sewage drainage, but relatively low in other samples and comparable to the national background value for soils. Moreover, the metal concentrations from salt marsh sediments in the Changjiang Delta Region are lower than those from Europe, for example, the Thames Estuary and salt marsh along the Atlantic coast of Spain, either in absolute levels and metal to Al ratio (Attrill and Thomes, 1995; Carimen et al., 1997).

6. FISHERIES IN THE ESTUARY

The spawning, hatching, and recruitment of fish species have been greatly impacted by the engineering activities in the Changjiang Estuary, including dredging of navigation water-way port and seawall construction. The salinity at the river mouth has decreased by up to 10-15 after the construction of jetties of the deepwater navigation channel (see also Zhu et al., 2005). Also, the changes in dynamic conditions generate more frequent resuspension of bottom sediments and increase the sediment flux through the channel area, which affects the composition of benthic fauna. The biodiversity of benthic fauna has decreased for the last 25 years. Within the benthic fauna, polychaeta, mollusca, and crustacea show a dramatic reduction in species composition and biomass.

The dredging of bottom sediments for navigation causes the loss of habitat for benthic fauna, for example, *Eroicheir sinensis*. Moreover, the resuspension of bottom sediments in dredging activity results in the alteration of life history of crab (e.g. zoea and megalopa); high turbidity affects the feeding and food-size selectivity of larvae and induces the retard of the molt cycle of stage zoea I (Wang et al., 1999a). Also, the remobilization of heavy metals by dredging in the Changjiang Estuary causes a negative physiological behavior (e.g. spawn and hatch) of benthic fauna; the release of metals (e.g. Cd^{2+}, Cu^{2+}, Hg^{2+}, Pb^{2+} and Zn^{2+}) from resuspended sediments can induce high mortality of crab larva (i.e. stages of zoea I-IV), with 24 hr – LC_{50} at metal level of 100 μM (Wang et al., 1999b).

The benthic fauna were relatively abundant in the 1970s, with species number of 150-160 (Figure 7). The species abundance for benthic fauna fell to 30-40 in 1990 and then to 10-20 in 2002 (Ye et al., 2004a). At the same time, the major components of the benthic fauna, i.e. polychaeta, mollusk and crustacean, all show a concurrent reduction of species abundance by 80-90 % from 1970s to 2002. After the 1990, the species abundance of benthic fauna remains low but rather stable, with polychaeta and mollusk being more abundant than crustacean (Figure 7).

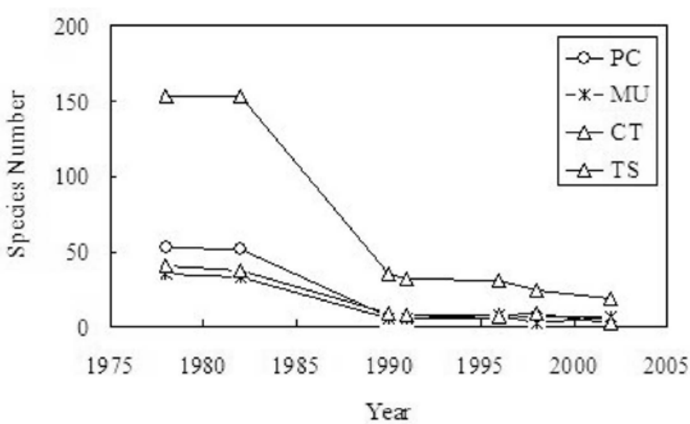

Figure 7. *Changes in benthic fauna in the Changjiang Estuary, which indicates that the total species (TS) number of community decreased by 6-8 times in the period 1978-2002, together with reduction in species compositions for major benthic structure, including polychaeta (PC), mollusk (MU) and crustacean (CT), respectively (Adapted from Ye et al., 2004b).*

The comparison to the data of 1990s reveals that traditional key species of benthic fauna, such as *Palaemon gravieri*, *Exopalaemon annandalei*, *Collichthys lucidus*, *Harpodon nehereus*, *Eriochier leptognathus*, *Ericthonius sp.*, and *Capitellethus sp.* were not found anymore in 2002 (Ye et al., 2004a-b). Moreover, the biomass abundance of benthic fauna in sediments in areas affected by the engineering activity is on average only 21.6 individuals m^{-2} in 2003-2004, which is 85.9 % that in the 1980s, that is, before the substantial dredging for navigation (Ye et al., 2004b).

Dredging induces the systematic excavation of bottom sediments; constructions for harbor and bridge destroy habitats necessary for the spawning and recruitment of fish species, while reclamation of salt marsh for agriculture and human settlements result in further loss of habitat in the Changjiang Delta Region. Consequently, wildlife is dramatically impacted and become less abundant than in the past. For instance, the biomass of *Stolephorus*, a traditional commercial fish species in the region, has been considerably reduced after 1990 and shows almost no landing records in last 5-10 years (Data Source: www.shtong.gov.cn). In the early 1960s, the catch of *Stolephorus* in the Changjiang Estuary, as reported from the Year Book of Shanghai, was 560 tons yr^{-1}, the landing of this species dropped to 5-10 tons yr^{-1} in 1990 (Figure 8). Although the extinction of *Stolephorus* is believed to have started from the destruction of spawning ground by pollution before the construction of the jetties, the jetty construction interferes, however, with the recruitment of adult species by blocking the migration routes, making reproduction unsustainable. Similarly, the landing records for crabs in this region show shut-down of commercial values after 1980s owing to the over-fishing and destroy of habitats by human beings in coastal environment (Figure 8), and the catch of crabs was banned in 1990 owing to a population crash.

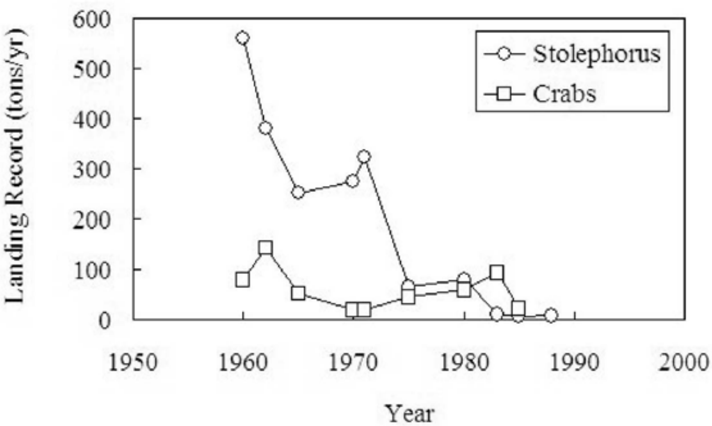

Figure 8. *Change in landing records for Stolephorus and crabs in the Changjiang Estuary based on the data from municipality of Shanghai, which shows the dramatic reduction in fish catch in the period of 1980-1990 (Data Source: www.shtong.gov.cn).*

From the local government statistics, the fishery industry has collapsed since the initiation of channel dredging and construction of jetties in the Changjiang Estuary in 1990s. *Coilia mystus* (Linnaeus) and *Coilia nasus* Temminek et Schlegel are two of few remaining native fish species and once served as most important fishery resources with important commercial value (Ni, 1999). One of the plausible explanations for their decreased biomass is the combination of two factors, namely the pollution-induced change in plankton community and the function of jetty modifying the bottom morphology, the hydrographic conditions (e.g. circulation) and the salinity as well as blocking the migratory routes of fish (Figure 9). As the jetties narrow the water channel and increase the water depth by dredging, the natural structure of bottom sediments in adjacent areas is under continuous modification, e.g. by erosion and dredging, and the dredged sediments are removed to the wetlands nearby, which causes the irreversible damage to benthic fauna. As shown in Figure 9, the construction of jetty can eventually block the migratory routes and damage the spawning ground for certain important fishery species in this region, inducing collapse of recruitment.

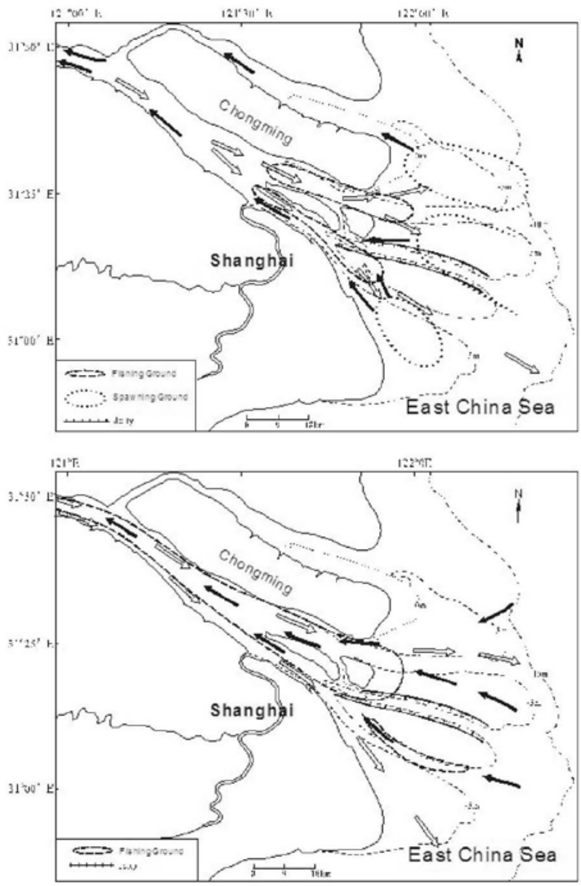

Figure 9. *Map of the Changjiang Estuary, which shows the migration routes of Eriochier sinensis (upper panel) and Coilia mystus (lower panel) with fishing and spawning/hatching fields; the two jetties in the main channel are also shown. The full and open arrows show the up-river and down-river migration routes of the adults and larvae, respectively. (Adapted from Yu, 1998, and Ni et al., 1999).*

Acipenser scnensis Gray migrates up-stream the estuary to spawn; it has now become so rare that it has been put in the national list of threatened and endangered species. The reason for the population crash is that from May to September the larvae of *Acipenser scnensis* migrate down the estuary to feed and grow before going offshore as adults migrating to the Yellow Sea and the East China Sea. Based on field observations, dense populations for *Acipenser scnensis* larvae were found to the southeast of Chongming Island. As fish habitats in that region were destroyed by coastal engineering (i.e. reclamation) in the Changjiang Estuary after 1990, the recruitment of this species decreased. This effect is compounded by overfishing and the construction of the jetties, putting the standing stocks of this species into a critical status.

7. HARMFUL ALGAL BLOOMS IN THE ADJACENT COASTAL WATERS

The records of harmful algal blooms (HABs) in China can be traced in literature back to 1950s (Zhang, 1994). In the periods of 1950s-1990s, 118 HABs were reported in the coastal environment of East China Sea, accounting for 35-40 % of total national records (Zhou et al., 2001). It is indicated that after the 1990s, the appearance and economic loss caused by HABs in China have dramatically increased, e.g. by 5-10 folds. For instance, in 2004 four major HABs are reported in the region off the coast from Shanghai (i.e. off the Changjiang Estuary), i.e. about 10 % of total events recorded for the East China Sea (Table 3), and 60 % of HABs recorded in 1990-2001 in the region off the Changjiang Estuary occur in spring of the year (Zhou et al., 2003), which is further increased to 60-80 % in the new millennium. Moreover, the HABs off the coasts from Changjiang Delta Region differ from those in other areas of China Seas by longer duration, larger area and diverse species compositions. The most extensive HABs in the East China Sea are usually found 50-100 km off the river mouth from Changjiang at water depth of 30-40 m, with surface area up to 1000-1500 km^2 along the coastal front region further offshore from high turbidity Changjiang effluent plume that can be easily seen from satellite images. Concentrations of chlorophyll-a can be as high as 300 mg m^{-3} induced by HABs in the region of salinity for 25-30. In this region, the HABs in spring are usually caused by dinoflagellates, notably the *Prorocentrum dentatum*, while in summer and autumn the dominated species become the diatoms, for example, *Skeletonema coastum*, when Changjiang floods. Data from mesocosm experiments show that diatoms (e.g. *Skeletonema coastum*) become more abundant

Table 3. *Records of HABs in the coastal waters off Shanghai with comparison to the East China Sea (Data are from State Oceanic Administration, 2001-2004).*

Year	Shanghai	East China Sea	All China Seas
2001	2	34	77
2002	5	51	79
2003	8	86	119
2004	4	53	96

than dinoflagellates (e.g. *Prorocentrum dentatum*) towards the addition (e.g. riverine input) of plant nutrients (e.g. PO_4^{3-}) and hence the overwhelming species through competition, whereas at low nutrient case dinoflagellates remain dominant (Li et al., 2003).

Frequent HABs in the region offshore from the Changjiang Estuary is believed to be linked with increase in land-source input of plant nutrients and coastal eutrophication (Zhou et al., 2003), however, new evidence from field observation suggests that in the early spring the blooms of harmful algal species be affected by the coastal circulation that brings the nutrients from offshore waters (e.g. Kuroshio) and hence maintains high biomass via vertical mixing. Indeed, the blooming of harmful algal species in spring is facilitated by the change in weather from northerly to southerly monsoons, which induces coastal upwelling. One of the questions with the mechanism of spring blooming of dianoflagellates (e.g. *Prorocentrum dentatum*) rather than diatoms is the life history is poorly understood, although it is hypothesized that dinoflagellates (i.e. *Prorocentrum dentatum*) may have over-winter strategy as resting cysts of in the southern part of the East China Sea, this needs to be tested by the carefully designed field measurements in the near future.

8. DISCUSSION AND CONCLUDING REMARKS

Traditionally, the Changjiang Delta Region delivers extensive services to human beings, and this plays an important role in the civilization, economy and social development of China. The historical records of early settlement can be traced back to ca. 1500 years ago. It was one of the trade centers in Ming and Qing dynasties and became the most densely populated area in China in the 20th century. The Changjiang Delta Region is among the most economic advanced regions in China. The GDP value of Shanghai, for example, reached about 650×10^9 Yuan of RMB in Year 2003, with a rate of economic increase by >10 % on average over the last decade (Data Source: www.shtong.gov.cn). Also, the delta region serves as critical connection between oceanic trade and inland transportation upstream the Changjiang over a distance of 4000 km. The port of Shanghai had in 2004 a throughput of about 0.35×10^9 tons yr^{-1}, among the top three of the world.

The delta region sustains a richness of biological resources. For instance, in the Changjiang Estuary, about 160 species for phytoplankton and 174 species for zooplankton have been reported. Moreover, there are 14 orders and about 120 species of fish, with 92 % from *Teleostean* and 8 % from *Elasmobranch*, respectively. The fish ecosystem in the delta region is composed of fresh-water, brackish-water, marine and migratory species (Table 4).

To describe the human impact on the health of ecosystems in the Changjiang Delta Region in full dimension (i.e. time and space), like in many other world systems, is difficult, if not impossible, because of the multitude of direct and indirect effects. Land-based activities in the river basin result, for the delta region, in an increase in nutrient influx and other pollutants. While the resulting high level of nutrients may not necessarily result in a serious eutrophication and in blooms of harmful phytoplankton species in the inner part of delta region, owing to high

Table 4. *List of the major components for the fish in the Changjiang Estuary. (Adapted from Zhang and Jiang, 1983, and Chen et al., 1999).*

Category of fish	Contribution (%)	List of Major Species
Riverine Species	17.4	*Coilia*, *Protosalanx hyalocranius*, *Mylopharyngodontis piceus*, *Hypophthalmichthys molitrix*, *Aristichthus nogilis*, *Carassius auratus*, *Parapelecus argenteus*, *Leiocassis longirostris*
Brackish Species	21.6	*Fugu xanthopterus*, *Leiocassis longirostris, Lateofabrax japonicus*, *Mugil cephalus*, *Mugil soiuy*, *Hemisalanx prognathus*, *Chaeturivhthys* , *Periophthalmus cantonensis*
Marine Species	57.2	*Muraenesox cinereus*, *Clupanodon punctatus*, *Llisha clongata*, *Selipinna taty*, *Harpodonle nehereus*, *Zebrias zebra*, *Eleutheronema tetradactylum*, *Pneumatophorus japonicus*, *Scomberomorus niphodius*, *Trichiurus haumela*, *Stnomateoides argenteus*, *Collichthys lucidus*
Migratory Species	3.8	*Anguilia japonica*, *Psephurus gladius*, *Acipenserda bryanus*, *Trachidermus faxciatus* , *Tenualoosa reevesi*, *Coilia nasus*, *Coilia mystus*, *Acipenser sinensis*

turbidity, other pollutants (e.g. sewage from urban source) have been identified to induce detrimental effect on the water quality and aquatic ecosystems. For instance, the landing records of *Stolephorus* by the municipality of Shanghai show a dramatic decrease of recruitment in the 1980s, that is, due to pollution from urban waste discharge in their spawning ground in the upper part of the estuary; the problem was exacerbated by overfishing and it became critical by construction of the jetties in the Changjiang Estuary. Hence the resource has never recovered after 1990 (Figure 8). Similarly, the over-fishing of *Eriocheirs sinensis* in the delta region caused the collapse of the fisheries, which later further deteriorated by the loss of hatching ground by coastal engineering works; the situation is critical for this species for which a fishing ban has been issued in the late 1980s (Figure 9). Other important species in this region include *Coilia ectenes*, *Coilia mystus*, and *Exopalaemon annandalei*; their growing and/or hatching grounds in the estuary are affected by the construction of the jetties (Figure 9).

The estuary is a multi-functional system, providing foods (e.g. fishery) and variety of services to the society, including biodiversity, transportation, recreation, agriculture and urbanization, which is under the regulation of human activity and react with positive and/or negative feed-back to the human society, being either sustainable (i.e. negative) or unsustainable (i.e. positive).

While human settlement in the Changjiang Delta Region is expected to continue in the near future, the conflicts of shortage of land for human settlement and is becoming critical. While reclamation of wetlands in the coastal area is required to maintain the progress of urbanization in the Shanghai, the environmental consequences include the loss of habitats for wildlife, the destroy of spawning and hatching grounds for marine living resources, and this change their community structure and put those economically valuable species in danger of extinction.

Pollution by trace metals and synthetic organic materials can cause the problems at molecular level and induce the genetic diseases, which in turn alter the strategy at organism level and change the whole food-chain. An increased waste discharge from Changjiang into the East China Sea has caused frequent harmful algal blooms in coastal waters further offshore from delta region, and a serious hypoxia problem has recently reported in literature, which induces the concern of public society nation wide. Dredging in the Changjiang Estuary for navigation and construction of jetties in the main channel have been reported to further destroy the fishing grounds and block the migratory route of traditional economic species in the region. The Changjiang Estuary is losing its traditional values at the ecosystem level; its historical role of providing multiple services to society is changing to a simplified service system, e.g. land area for settlement and waterways for transportation and trade.

The on-going engineering constructions in the Changjiang Delta Region are expected to further modify the habitats and affect community structure of wildlife, which can affect the whole ecosystem via food-web interactions, unless proper management is made to protect the habitats and to improve the water environment at the ecosystem level.

9. ACKNOWLEDGEMENTS

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CHAPTER 8

GEOGRAPHICAL AND ECONOMICAL SETTING OF THE PEARL RIVER ESTUARY

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XIAOHONG WANG, AND QIUHAI LIU

1. INTRODUCTION

A shown in Figures 1 and 2, the Pearl River discharges into the South China Sea (SCS) through eight distributaries, locally called “the eight Gates” (or distributaries). The four western Gates (distributaries), the Modaomen Gate, Jitimen Gate, Hutiaomen Gate and Yamen Gate, discharge directly into the SCS. The four eastern gates, Humen Gate, Jiaomen Gate, Hongqimen Gate, and Hengmen Gate, discharge their waters into the “Lingdingyan”, which will be called the “Pearl River Estuary” (PRE) in subsequent discussions.

The PRE has an area of over 2,000 km^2, varying in width between 15 km at the northern end and about 35 km at the southern end, with a length of about 70 km. There are two deep channels, which are used for shipping. The western channel connects the SCS via the Lantau Channel (Figure 2) through the southeast side of the estuary mouth. The eastern channel leads to the SCS through Hong Kong waters. The water depth increases from north to south and, in the southern part of the estuary, the water depth decreases from east to west. Except for the deep channels and the areas around the two outer islands, Wansham Islands and Dangan Islands (Figure 2), where the water depth ranges from 20 to 30 m, most of the PRE is quite shallow with a water depth between 2 and 10 m.

The annual average discharge of the Pearl River is around 10,000 m^3 s^{-1}, 53% of which flows through the four eastern gates. Eighty percent of the total discharge occurs in wet season between April and September, the ratio of maximum to minimum discharges in a year varying between 3 and 6 times (Chen and Heinke, 2002).

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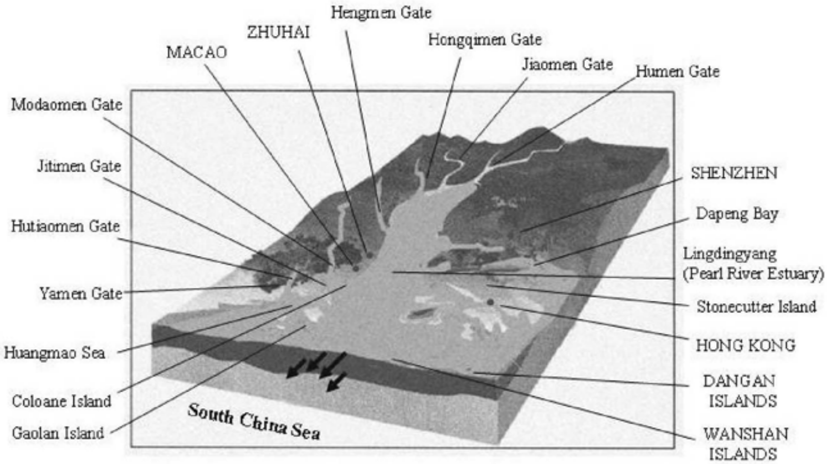

Figure 1. Schematic diagram of the Pearl River estuary (adapted from Chen and Heinke, 2002).

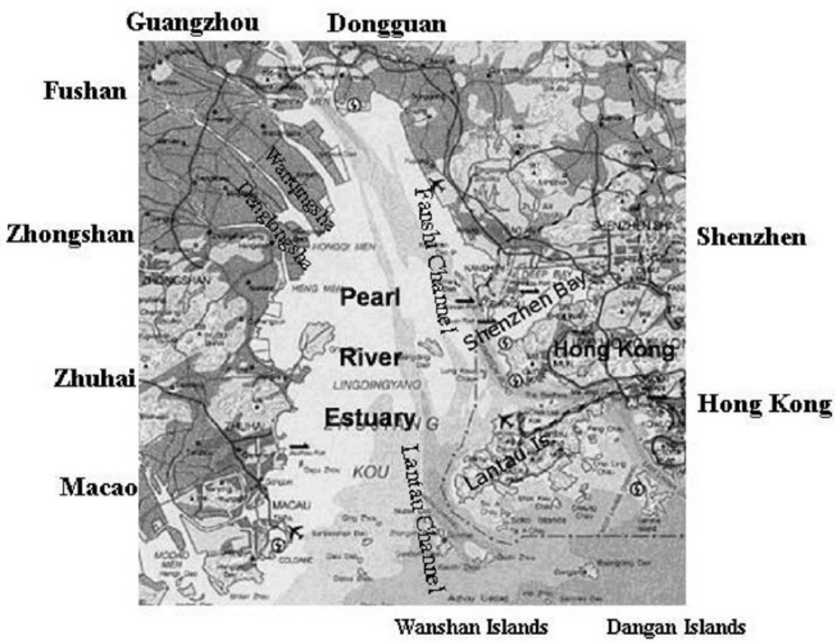

Figure 2. The Pearl River estuary (PRE).

2. RIVER DISCHARGE AND SEDIMENT LOAD

The Pearl River has three major branches (Xijiang, Beijiang and Dongjiang) and a network of small rivers in the river delta (Figure 1). Table 1 summarizes the discharge and the suspended sediment load of the rivers. The Xijiang River has by far the largest amount of discharge and suspended sediment transport, accounting for 77% of the water discharge and 86% of the suspended sediment discharge.

Table 1. *Water and sediment transportation from the Pearl River (adapted from Luo and Zhen, 2000; PRWERC 1993).*

	Water Discharge		Suspended Sediment Load	
	Annual discharge (10^9 m^3)	% of the total	Annual transport (10^6 ton)	% of the total
Xijiang River (Gaoyao)	225.0	67.5	76.60	86.5
Beijiang River (Shijiao)	49.0	14.7	8.17	9.2
Dongjiang River (Boluo)	28.0	8.4	2.94	3.3
Rivers in the Pearl River Delta	31.3	9.4	1.01	1.0
Total	333.3	100.0	88.72	100.0

Table 2 lists the average annual discharge of water through the eight Gates (distributaries). The total discharge through the eight Gates is slightly less than that of the Pearl River (Table 1), because some of the discharge from the Pearl River enters the sea through small rivers in the delta. Table 3 lists the average yearly volume deposition inside the PRE and near the four western Gates (distributaries). High deposition rates (> 2 cm y^{-1}) occur outside the delta slope and over the shoals west of the PRE. The deposition rates reach peak at 5-8 cm y^{-1} in some local areas in Modaomen Gate, PRE, and Denglongsha and Wanqingsha shoals (Figures 1 and 2). Very high rates of 10-15 cm y^{-1} have been recorded in parts of the channels.

3. HUMAN POPULATION

The total population in the Pearl River basin rose rapidly from 43.1 million in 1956 to 89.42 million in 1993 (Figure 3). The yearly population growth rates were 2.14%, 1.72% and 1.85% for the periods 1956-1980, 1980-1985, and 1985-1993, respectively. The population density in the region reached 211 people km^{-2} in 2000, much higher than that of the whole country (PRWRC, 2004).

Table 2. *Tidal range at the water discharge through the eight Gates of the Pearl River (adapted from Luo and Zhen, 2000; PRWERC 1993). 1=Humen Gate; 2=Jiaomen Gate; 3=Hongqimen Gate; 4=Hengmen Gate; 5=Modaomen Gate; 6=Jitimen Gate; 7=Hutiaomen Gate; 8=Yamen Gate.*

		1	2	3	4	5	6	7	8	Total
Tidal range (m)	Average	1.63	1.36	1.21	1.11	0.86	1.01	1.20	1.24	
	Maximum	3.39	2.81	2.79	2.48	2.29	2.71	2.66	2.95	
Annual river discharge ($10^8 m^3$)		603	565	209	365	923	179	202	196	3260

Table 3. *Sedimentation in the Pearl River estuary and neighboring waters (adapted from Luo and Zhen, 2000; Dong 1986).*

		Area of deposition (km^2)	Yearly deposition (10^4 m^3)	Average deposition rate (cm y^{-1})
River channels		700	1450	2.1
Eastern 4 Gates	Lower PRE	1000	2000	2.0
	Upper PRE	1000	800	0.8
Western 4 Gates	Modaomen Gate	180	450	2.5
	Jitimen Gate	107	160	1.5
	Yamen Gate and Huangmao Sea	535	730	1.4

The Pearl River Delta is one of the most populated areas in the Chinese mainland with a population over 28 million and a density of 674 people km^{-2}, excluding Hong Kong and Macau (1999 data). In total, 28 cities and 420 towns are found in this delta with a "town density" of 10 towns per 1,000 km^2. The population in the 12 largest cities of the delta is shown in Table 4; it grew from 27.02 million people in 1996 to 28.27 million people in 1999.

Table 4. *Population of the 12 largest cities in the Pearl River delta (millions of people) (adapted from Statistics Bureau of Guangdong Province 2000; Statistics Bureau of Guangxi Province 2000).*

Cities	1996	1997	1998	1999
Guangzhou	6.561	6.665	6.742	6.850
Shenzhen	1.034	1.095	1.146	1.199
Zhuhai	0.654	0.673	0.695	0.714
Foshan	3.161	3.210	3.250	3.292
Huizhou	2.601	2.665	2.700	2.718
Zhaoqing	3.615	3.683	3.725	3.813
Jiangmen	3.744	3.771	3.789	3.798
Dongguan	1.453	1.471	1.488	1.508
Zhongshan	1.268	1.284	1.301	1.320
Shunde	1.025	1.040	1.053	1.068
Nanhai	1.043	1.058	1.071	1.085
Panyu	0.867	0.880	0.896	0.911
Total	27.026	27.495	27.856	28.276

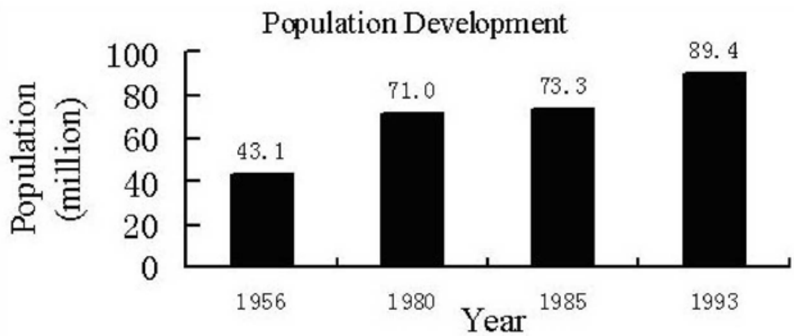

Figure 3. *Population growth in the Pearl River Region (adapted from PRWRC, 1993).*

4. ECONOMIC DEVELOPMENT

The total gross domestic product (GDP) in the basin was RMB 1330.0 billion in 2000, with a per capita GDP of RMB 7,917 or approximately US$960. Although significant economic growth has occurred across the whole region, the delta's economy has grown much faster than other areas in the basin, with a per capita GDP exceeding US$3600 in 2000, the highest in the country. From 1980 to 1994, the GDP in the Pearl River Delta increased by 17.8% annually, while that of Guangdong province and the whole country increased by 14.5% and 9.7%, respectively (Figure 4). The economy has continued to boom with GDP growth of more than 10% from 1995 till present.

In the Pearl River delta, Guangzhou, Shenzhen (Figure 2) and ten other cities are the most important economic units in terms of their economic scale, degree of economic innovation, ability to attract skilled workforce, knowledge and investment.

Table 5 lists the GDP values in these 12 cities. Four cities, Guangzhou, Shenzhen, Fushan and Dongguan (Figure 2), play a remarkable role in the delta's economic development. Nearly three-fourths of the 2003 GDP of the delta came from these four cities.

Not only did the GDP increase but the structure of the economy has also improved during this period. Figure 5 shows the rapid increase of the proportion of the "Tertiary Industry" from 29% to 44% between 1980 and 1999 and the concurrent decrease of the "Primary Industry" from 26% to 6%. The Pearl River Delta has become an important supplier of a variety of products to the world. In 1999, the exported goods from the Pearl River delta were valued at US$ 67 billion.

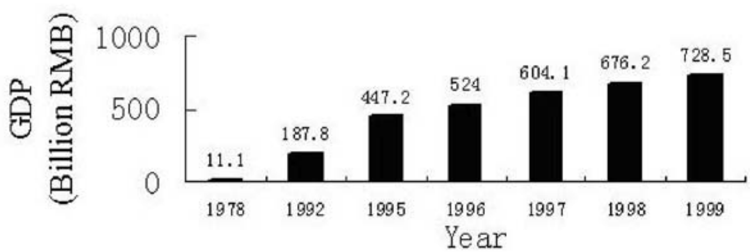

Figure 4. *GDP growth in the Pearl River Delta (adapted from Statistics Bureau of Guangdong Province 2000; Statistics Bureau of Guangxi Province,2000).*

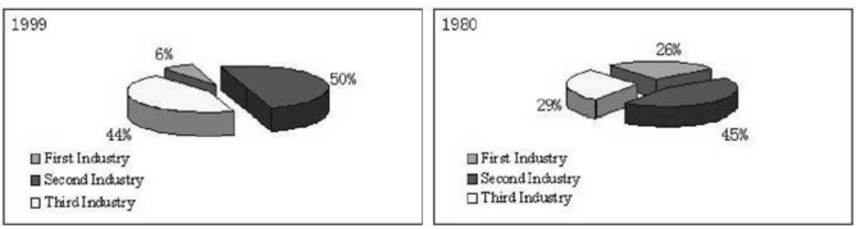

Figure 5. *Change in the economic structure in the Pearl River Delta (adapted from Statistics Bureau of Guangdong Province 2000; Statistics Bureau of Guangxi Province 2000).*

5. HARBORS AND TRANSPORTATION

The Pearl River Estuary is densely distributed with ports (Figure 6). Guangzhou, Shenzhen and Hong Kong are the principal ports in the region, while the others are lateral ports or feeding ports of the main ports.

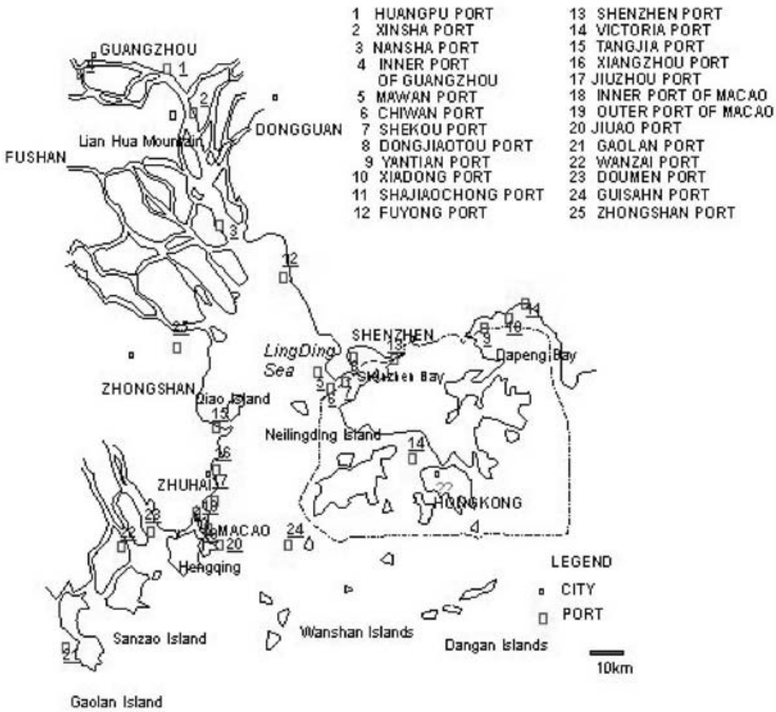

Figure 6. *Location of ports in the Pearl River estuary and surroundings.*

Table 5. *GDP (billions of RMB) of nine major cities in the Pearl River delta (adapted from Statistics Bureau of Guangdong Province 2004; Statistics Bureau of Guangxi Province 2004).*

City	1999	2000	2001	2002	2003
Guangzhou	205.674	237.591	268.576	300.169	346.663
Shenzhen	143.603	166.574	195.417	223.941	286.051
Foshan	83.379	95.720	106.836	116.866	137.993
Dongguan	41.284	49.271	57.893	67.227	94.753
Jiangmen	51.469	56.751	61.516	65.829	73.109
Huizhou	39.237	43.724	48.039	52.546	59.020
Zhaoqing	35.549	38.340	41.102	44.388	50.115
Zhongshan	27.268	31.282	36.250	41.553	49.710
Zhuhai	28.661	33.026	36.659	40.627	47.327

5.1. Guangzhou Port

Guangzhou Port is located in the center of the Pearl River delta. It is the main port of South China and has 140 berths with different functions, 9 of which have a capacity of 10,000 t, 8 with a capacity of 20,000-25,000 t and 15 with a capacity of 35,000 t. It also has 22 mooring areas, each with a berthing capacity of over 10,000 t; the maximum capacity is 300,000 t. The total length of the navigable channel of the Port is about 173 km, including the seagoing channel of 115 km. General cargo ship of 13.5 m draught and container ship of 4th and 5th generations ships are able to pass through during the flood tides. (source: Information and Communication Center of Guangzhou Harbor Bureau)

The first stage of the Guangzhou Port watercourse dredging project was completed in 2000. This resulted in deepening the navigation channel depth from 9 m to 11.5 m. Further dredging is planned to deepen the channel to 13 m by the end of 2005 (Zhou, 2004).

5.2. Shenzhen Port

Shenzhen Port comprises a total of nine port areas, namely, the Shenzhen Inner Port area, Shekou, Chiwan, Mawan, Dongjiaotou, Fuyong, Yantian, Xiadong, and Shatianchong (Figure 6). The western ports are situated along the Fanshi Channel (Figure 2) by the upper reaches of the PRE, about 35 km north of Hong Kong. The eastern port is located in northern Dapeng Bay (Figure 1) with water depth of 12 to 14 m (Zhou, 2004). Altogether Shenzhen Port has 102 berths with capacity over 500 tons, in which 23 are above 10,000 t and 65 for production berth. In 2003, Shenzhen's container handling capacity reached 10.6×10^6 TEU and the total cargo handling capacity was 87.67×10^6 t in 2002 (Statistic Yearbook of Guangzhou, 2004), making it the fourth largest container transportation base in the world.

5.3. Hong Kong Port

This is described in other chapters in this book.

5.4. Macao Port

Macao Port is situated on the west side of the PRE, 150 km south of Guangzhou and 70 km west of Hong Kong. The Port covers three sub areas: the inner part of the Port is on the west coast of the Macao Peninsula and the outer part on the southeast side of the Peninsula, while the Jiuao Port is at the southeast side of the Coloane Island (Figure 1) (Port Bureau of Macau, 2004).

Macao Port silts readily and thus not suitable for a deep water port. The major function of the port is the transportation of both passenger and cargo between Macao, Hong Kong, and the mainland.

5.5. Zhuhai Port

Zhuhai Port consists of a number of sub-ports located both on the west coast of the PRE and around the Gaolan islands (Figure 1) in the Huangmao Sea. The sub-ports are, namely, Jiuzhou, Xiangzhou, Gaolan, Hongwan, Qianshan, Jing'an, Tangjia, Doumen, Wanzai, and Guishan (Figure 6). The sub-ports can be grouped into three parts in accordance with their location and function. The Gaolan area of the Zhuhai Port is a transitory base for international freights. The middle part of the port is mainly for passenger and the west part is mainly for transportation of petrochemical products and various other goods. In total, the Zhuhai port has 8 berths of over 10,000 t capacity, with an annual total handling capacity of 28×10^6 t (Statistics Bureau of Guangdong Province 2004).

6. WASTEWATER DISCHARGES

Wastewater discharge in the Pearl River Delta originates mainly from the rapidly developing cities such as Guangzhou, Shenzhen, Zhuhai, Dongguan, and Zhongshan, on the Chinese mainland (Figure 7), as well as Macao and Hong Kong, together with from shipping and transportation activities over the estuary.

Table 6 provides data on the Pearl River delta, as well as on the percentage of the industrial wastewater treated. Much of the industrial wastewater is treated. However, there is no information available on the extent of the treatment of household wastewater.

Table 6*. Discharge of household and industrial wastewater in 10 major cities in 1997 (adapted from Statistics Bureau of Guangdong Province 1998).*

	Total discharge of wastewater (household and industrial) (10,000 m^3)	Total discharge of industrial wastewater (10,000 m^3)	Percentage of industrial wastewater treated (%)
Guangzhou	105,327	26,764	87.3
Shenzhen	44,781	2,666	98.3
Zhuhai	11,238	3,741	73.2
Shantou	16,596	4,003	81.4
Shaoguan	22,581	13,130	93.6
Dongguan	29,370	8,930	86.2
Zhongshan	10,400	5,905	88.7
Jiangmen	20,878	10,689	65.7
Foshan	32,488	12,391	93.4
Zhanjiang	22,547	8,947	60.7
Total	316,206	97,166	83.6

7. ADMINISTRATIVE MEASURES TOWARDS SOLVING ENVIRONMENTAL PROBLEMS

The PRE is facing a continued rise in pollution load and the increasing occurrence of red tides. This degrades water quality and endangers aquaculture and fishery resources. To solve these environmental problems, different strategies and measures as follows have been proposed.

7.1. Industrial pollution control

Industrial pollution control measures taken in the area are many, namely: to ensure that the effluent criteria for industrial wastewater discharge are met; to speed up the restructuring process in industrial sector; to actively promote "clean production" and to increasing the efficiency of energy use; to reinforce the implementation of environmental impact assessment system; to build central wastewater treatment plants in industrial parks and to decrease the level of disposed pollutants; to increase the recycling and re-use of water in the manufacturing industry; to prohibit the building of new thermal power plants (using coal) and to timely resolve the desulfurizing problem for the existing thermal power plants. At present the effort in pollution regulation and control is focused on the following industries that have serious pollution problems: electric power generation, construction material, chemical, pulp and paper making, metal refining, sugar, fermenting, plating, spinning, printings and dyeing.

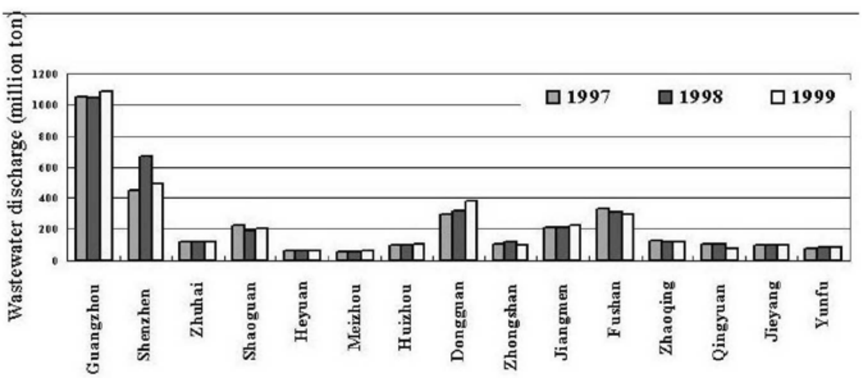

Figure 7. *Yearly wastewater discharge from 15 major cities along the Pearl River from 1997 to 1999 (adapted from Statistics Bureau of Guangdong Province 2000; Statistics Bureau of Guangxi Province 2000).*

7.2. Domestic sewage treatment

Domestic pollution control measures taken in the area are many and include: to coordinate the building of wastewater treatment plants in urban and rural areas; to promote building new and expanding existing wastewater treatment plants in a comprehensive way, and improving the operation efficiency of these plants; to reduce the discharge of nitrogen and phosphorus by raising the efficiency of the denitrification and dephosphorization treatments in sewage systems; to prevent and control the non-point pollution sources and to remove the black and odor materials in urban small rivers and creeks.

There are 67 wastewater treatment plants in urban areas of the Guangdong province with daily treating capacity of 5.083×10^6 t, out of which eight new plants were built in 2004. Among these 67 plants, 55 are located in the Pearl River delta region with the daily treating capacity of 4.738×10^6 t.

By the end of 2001 the Stonecutters Island (Figure 1) wastewater plant in Hong Kong was put into operation with the daily treating capacity of $1.70\times10^6\,m^3$. There are 16 catchments in Hong Kong collecting and treating more than 98% of the wastewater. The whole system consists of wastewater collection network with the total length of about 1,320 km and 200 wastewater treatment plants. The treated wastewater is discharged into the deep sea for further dilution and diffusion. 75% of the wastewater around Victoria Port (Figure 6) was well-treated (by secondary treatment) before being discharged. (The Drainage Services Department, Government of Hong Kong). Macao has three wastewater plants with daily capacities of 144,000 m^3, 70,000 m^3 and 20,000 m^3, respectively. About 90% of the wastewater networks are connected to the wastewater plants in Macao except for small areas like the docks in the Inner port of Macao and small islands nearby. All of the wastewater from the public works and household in Macao is treated completely before being discharged (Environmental Statistic Information Network).

7.3. Prevention and treatment of petro-waste in the sea

Oil tankers with carrying capacity over 150 ton and other cargo ships over 400 ton are required to be equipped with wastewater treatment facilities. Wastewater treatment facilities capable of treating petro-wastes have been built for each commercial port, and first and second class fishing ports. The discharge of poisonous liquid materials, engine rooms waste, sewage, and ballast water is prohibited; the principle of zero discharge of petro-pollutants is enforced. Emergent response plans/schemes have been drafted to deal with oil spills and other serious pollution incidents.

7.4. Pollution control regulations for river mouth areas

The following measures are included in the pollution control regulation for all the eight Gates (distributaries) in the Pearl River delta: to implement a system to control the discharge from aquaculture and to strengthen the diffuse (non-point source) pollution control; to dredge channels with serious polluted sediment; to prohibit the dumping of rubbish; and to regulate the exploitation of estuary sand for construction material.

7.5. Total pollutant load control

The following measures have been taken: to control the gross pollutants discharging into the sea and to control the quality of the water environment for each district on the basis of water environment capacity; to strengthen efforts in areas where the water quality is below acceptable standards, to closely monitor the water environment, and to take strong measures to enhance the water quality management for the intensive development areas; to strive for a harmonization between advanced and base water utilization.

The problem of wastewater disposal for Hong Kong is solved through 16 wastewater disposal systems covering the whole Hong Kong and through the "plan for harbours cleaning" implemented to collect all the wastewater. A sewer system and wastewater treatment plants are planned based on the pollution source areas, and an "Integrated Plan for Waste Water Collection" has been formulated. The Joint Working Panel of Environmental Protection for Guangdong and Hong Kong has developed a 15 year plan to improve the estuary's environment. The plan covers three phases with a duration of 5 years each and will be implemented from 2005.

In addition, the following measures are also taken: to establish eco-agriculture systems along the riparian areas; to emphasize awareness of protection of ecosystem/environment in the tourism industry; to reinforce management for delineated protection areas and protection of a tree belt along the shore; to ensure the proper treatment of solid wastes; to build a sophisticated system of environmental monitoring and information gathering; to reinforce the enforcement of the laws and regulations; to increase the investment for environmental protection; and to promote international cooperation in these activities.

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CHAPTER 9

PHYSICAL PROCESSES AND SEDIMENT DYNAMICS IN THE PEARL RIVER

LIXIAN DONG, JILAN SU, YAN LI, XIAOMING XIA,
AND WEIBING GUAN

1. INTRODUCTION

The Pearl River, with a total drainage area of 4.5×10^5 km^2 and an annual average discharge of 10,000 m^3s^{-1}, is the longest river in southern China. It discharges into the South China Sea (SCS) through eight distributaries (whose mouths are called Gates in Chinese). The four eastern Gates, namely, Humen Gate, Jiaomen Gate, Hongqili Gate and Hengmen Gate, discharge about half of the discharge of the Pearl River directly into the Lingdingyang Bay, called “the Pearl River Estuary (PRE)” in this section (Figure 1). The PRE, with an area of over 2000 km^2, has two deep channels separated by Lantau Island and the Neilingding Island. The east channel connects the coastal ocean through Mawan bay and Hong Kong waters and the west one passes through the west side of the Lantau Island and Neilingding Island. The two deep channels merge near the Humen Gate at the upper reach of the PRE. Except for the deep channels and the areas around the outer islands where the water depth varies between 20 and 30 m, most of the PRE is quite shallow with a water depth between 2 and 10 m (Figure 1).

The PRE is the most important estuary in southern China. The earliest hydrology study of the estuary was initiated in the 1850’s and modern comprehensive hydrodynamic studies started in the 1970's. Results of these studies, including tides, circulation, water and suspended sediment fluxes, are summarized by Xu (1985) and Zhao (1990). These early studies are mostly limited to the upstream side of the Neilingding Island. Since the 1990’s, several large comprehensive estuarine environmental research projects have been conducted in the PRE.

E. Wolanski (ed.), The Environment in Asia Pacific Harbours, 127–137

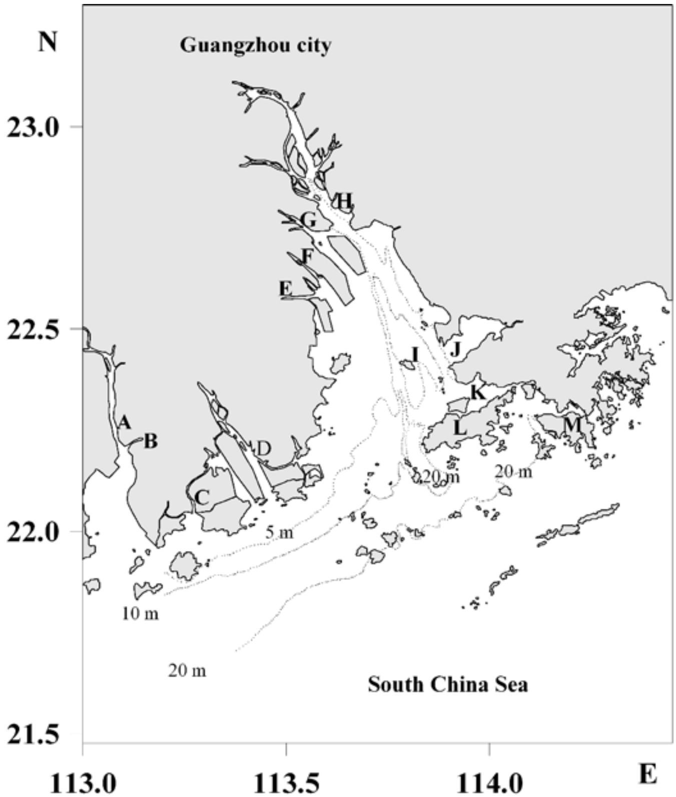


Figure 1. *Map of the Pearl River Estuary, or Lingdingyang Bay. A: Yamen Gate; B: Hutiaomen Gate; C: Jitimen Gate; D: Modaomen Gate; E: Hengmen Gate; F: Hongqili Gate; G: Jiaomen Gate; H: Humen Gate; I: Neilingding Island; J: Chiwan; K: Mawan Bay; L: Lantau Island; M: Hong Kong.*

Physical processes in the PRE are influenced by the combination of the East Asia monsoon, weak tidal dynamics, complex topography and large seasonal variation in both the runoff and coastal circulation. In this section, tides, hydrography, water circulation and suspended sediment dynamics in the PRE are described.

2. TIDES

The PRE is situated along a micro-tidal coast with a mean tidal range less than 1 m at the estuary mouth. The mean tidal range increases to 1.4 m at its middle reach (Chiwan) and to 1.6 m at the upstream end of the estuary (Humen Gate) (Figure 1). The flood tide duration decreases from 6.5 h at the estuary mouth to about 5.5 h at the upstream end of the estuary (Xu, 1985; Zhao, 1990).

Tides in the PRE are semi-diurnal with prominent diurnal inequality (Figure 2). M2 is the dominant tidal constituent, followed by K1, O1, and S2. When propagating upstream from the offshore side of the PRE, the M2 amplitude increases by approximately 80%, much larger than the increase of the amplitudes of K1, O1 and S2. The tidal constituent amplitude ratio, (K1+O1)/M2, varies from 1.8 at the estuary mouth to 0.94 at the upstream end of the PRE (Zhao, 1990). Over a spring-neap cycle, the diurnal inequality increases with the tidal range.

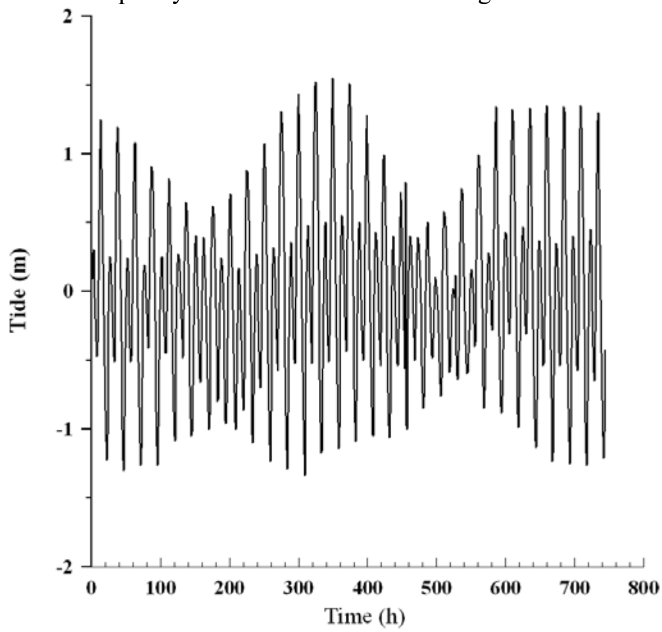


Figure 2. *Time-series plot of the tidal elevation at Humen during July, 1999.*

Except at the southeastern side of the estuary, ebb tides dominate inside the PRE. In the northwest part of the estuary the ebb duration exceeds the flood duration by about 2.8 h in July (wet season) and 1.0 h in February (dry season) (Xu, 1985). Observed flood currents reach 0.9 m s^{-1} in the east channel and the ebb currents exceed 2 m s^{-1} at the western side of the estuary (L. Dong, unpublished data).

3. HYDROGRAPHY

The PRE hydrography has distinct seasonal characteristics associated with the Asia monsoon. The northeast winter monsoon begins to appear over the northern South China Sea (SCS) in September and gradually diminishes in April. In the northern SCS, the winter monsoon has a monthly average wind speed of 7–10 m s^{-1}. The onset of the summer monsoon over the SCS usually occurs around the middle of

May. The southwesterly winds establish abruptly in the southern and central part of the SCS and soon expand to the entire SCS in June. In the northern SCS, the summer monsoon winds are actually more southerly, reaching peak values in July and generally less than 6 m s^{-1} (Su, 2004).

The Pearl River discharges into the SCS through eight distributaries. The four eastern-most distributaries, namely, Humen, Jiaomen, Hongqili and Hengmen, discharge their waters into the PRE. The annual average discharge rate through the PRE is about 5300 m^3 s^{-1}, about 53% of the total discharge rate of the Pearl River. Eighty percent of the total discharge from the Pearl River occurs in wet season between April and September. The ratio of maximum-to-minimum discharges in a year varies between 3 and 6.

3.1. Dry season

Temperature in the PRE is vertically homogenous during the dry season. It ranges over 16-20 °C, with warm water on the eastern side of the estuary (Huang and Ye, 1995, Dong et al., 2004).

Although the river runoff is reduced in the dry season, the salinity at the estuary mouth rarely reaches 32. This is because of the southeastward coastal current driven by the northeast monsoon (Dong et al., 2004). At the head of the PRE, the salt water can intrude a long way upstream into the river "Gates". For example, the salinity at Humen was observed to be about 10 and 20 at the surface and bottom, respectively (Zhang, 1984; Xu et al., 1985).

A salinity front usually exists in the PRE during the dry season. It runs seaward from the northern part of the upper reach of the PRE along the 5-6 m isobaths (Figure 3). The river plume, bounded by a salinity front, flows seaward along the west shoreline. Upon exiting from the PRE, it turns west along the coast outside the estuary. Numerical experiments show that, in winter, both the winds and river discharge have significant effects on the front. The frontal zone broadens in the surface layer with weakened northeast winds and also widens somewhat with increasing river discharge (Wong et al., 2004a). However, in the bottom layer, the front is hardly influenced by the winds.

As the runoff is greatly reduced in the dry season, the estuary is well-mixed by the strong northeast winds and the estuary is usually considered to be partially mixed (Xu et al., 1985; Zhao, 1990). Recent studies have found that stratification of the water column can be significant, the vertical salinity gradient reaching about 5 m^{-1}, in both the deep channel in the upper reach of the PRE and the region near the western shoreline in the lower reach (Dong et al., 2004). However, over most part of the PRE, the water column is vertically homogeneous (Dong et al., 2004).

3.2. Wet season

During the wet season, the southwest monsoon drives a northeastward coastal current just outside the PRE. In addition, saline oceanic water upwells close to the estuary mouth (Dong et al., 2004). The temperature and salinity are below 27 °C and over 33, respectively, in the bottom layer at the east side of the estuary mouth.

At the upstream end of the estuary, the salinity is close to zero and the temperature is about 30 °C near Humen Gate.

The freshwater flows out of the PRE largely through spreading across the entire upper layer (Figure 3). The offshore saline water intrudes into the PRE along the bottom layer of the eastern part of the estuary and moves up the estuary following the two deep channels. A strong halocline forms between the upper layer and the bottom layer. The low-salinity upper layer becomes thinner seaward, with a thickness of about 4-5 m in the upper reach, 3-4 m in the lower reach and 1-2 m near

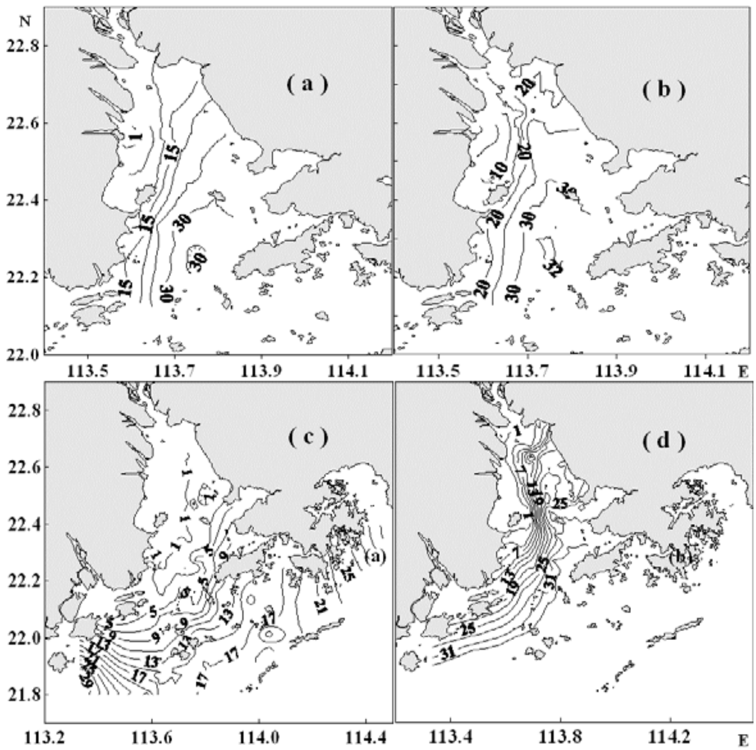


Figure 3. *Salinity distribution in the PRE, (a) near the surface in the dry season, (b) near the bottom in the dry season, (c) near the surface in the wet season, (d) near the bottom in the wet season.*

the estuary mouth (Dong et al., 2004). The salinity in the upper layer, on the other hand, increases from less than 3 in the upper reach to over 12 near the estuary mouth, as saline water from the bottom layer is entrained along the way. The bottom layer is vertically homogeneous. A bottom salinity front forms where the halocline intersects the estuarine bottom. The bottom front extends north-south along the axis of the estuary. It turns west parallel to the coastal shoreline outside the estuary after exiting from the PRE (Figure 3).

4. WATER CIRCULATION

The combined effects of a small tidal range and a large bay area result in an peculiar circulation pattern in the PRE. This pattern is different in the wet and dry seasons because of the distinct seasonal variation of both the river discharge and the East Asia monsoon.

4.1. Circulation during the dry season

During the northeast monsoon, a classical partially mixed estuarine circulation exists in the upper reach of the PRE, where longitudinal gravitational circulation dominates (Zhao, 1990; Dong et al., 2004). In the lower half of the PRE, however, the circulation assumes the typical pattern of a coastal current, where an alongshore front (parallel to the western shore of the PRE) and its associated frontal dynamics prevail (Dong et al., 2004). In that sense, in the middle reach of the PRE there is a transformation of the circulation pattern from a river plume (confined by the eastern shore of the PRE) to a coastal current (Su, 2004). The two deep channels are likely important in this transformation. The two deep channels conveniently provide offshore water to the eastern side of the middle reach of the PRE, where the transition takes place. In this connection, the ocean water entering the middle reach directly through the east channel via Hong Kong may play a more critical role. The coastal current continues to the west upon exiting from the PRE, as expected from the behavior of a density current in the northern hemisphere.

A two layer circulation develops in the stratified water between the frontal zone and the western shore of the estuary. The currents are southwestward in the surface layer and westward in the bottom layer (Dong et al., 2004). Numerical experiments demonstrate that there is a southward jet in the frontal zone, and density-driven circulation dominates the small scale circulation associated with the front (Wong et al., 2004a). During the spring tide, the sub-tidal currents driven by the tides enhance the southward surface current in the frontal zone.

4.2. Circulation during the wet season

During the wet season, the PRE is highly stratified and the circulation has a two-layer structure in the entire estuary. The measured surface-layer residual currents in July 1999 and August 1996 were all southward, with maximum speeds over 0.5 m s^{-1} and stronger currents on the western side of the front (Figure 4). Both the field data and the numerical results indicate that the plume exits the estuary in three branches (Dong et al., 2004; Wong et al., 2004a). The strongest one exits westward at the western side of the estuary mouth. The other two flow eastward, one through the eastern side of the estuary mouth and the other through the Hong Kong water (Figure 4). These plumes continue to flow eastward along the coast outside the estuary, probably driven by both the southwesterly summer winds and the eastward coastal current.

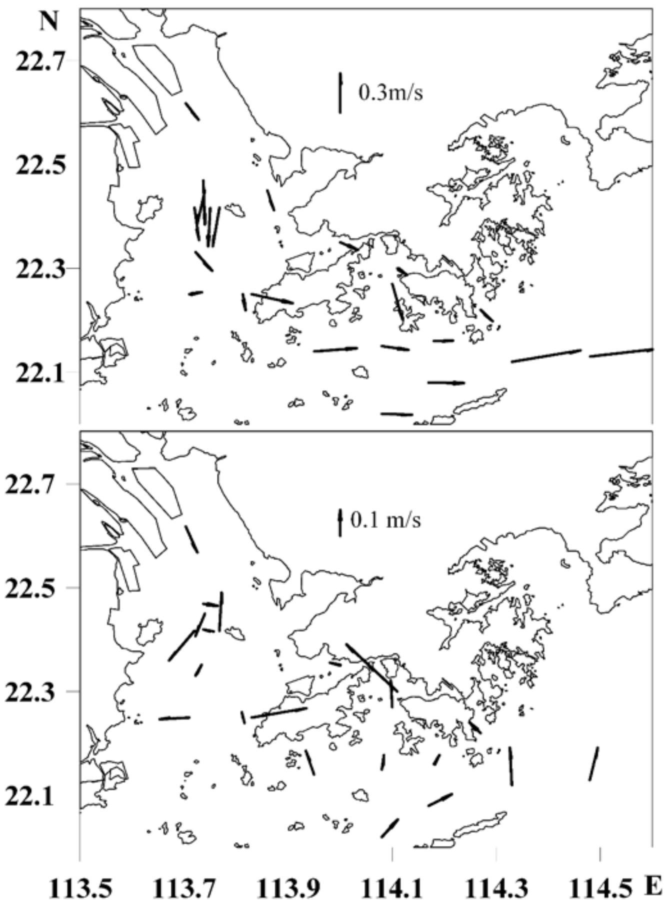


Figure 4. *Residual currents during the wet season, (upper figure) near the surface, (lower figure) near the bottom.*

Outside the estuary, the coastal currents are generally eastward during the wet season. Observations in August 1996 show that the currents outside the PRE respond readily to the variation of the local winds (Dong et al., 2004). The river plume spreads out rapidly in the surface layer over the inner shelf. Numerical experiments show that the coastal circulation, thereby the spreading of the river plume over the coastal waters, depends highly on the wind direction (Wong et al., 2004a). Westerly winds enhance seaward spreading of the plume, whereas easterly winds drive the plume towards the west and close to the coast, similar to the winter situation. If the surface plume reaches beyond the inner shelf, it may extend much further to the east across the shelf, possibly driven by both the northeastward shelf current and favorable winds (Su, 2004). Strong stratification favors the intrusion of the offshore bottom saline water into the PRE through the two deep channels. Thus, in the bottom layer of the PRE, the circulation still assumes the same coastal current structure that occurs during the northeast monsoon. The bottom residual currents on the western side of the front are either southwestward or towards the front (Figure 4). Because of the diminishing wind effects below the halocline, this bottom current continues to the west along the coast outside the PRE, at least initially, like a density current (Su, 2004; Wong et al., 2004a).

During the summer monsoon, the westerly winds induce coastal upwelling at many places along the southern coast of China (Han and Ma, 1988). Such an event usually brings to the nearshore area the SCS sub-surface water that has upwelled at the shelf break area (Su, 2004). Observations indicate that, in summer, upwelling of such water does occur along the bottom just outside the estuary mouth.

Table 1. *Distribution of the suspended sediment load for the eight tributaries of the Pearl River. 1=Humen; 2=Jiaomen; 3=Hongqili; 4=Hengmen; 5=Modaomen; =Jitimen; 7=Huttaomen; 8=Yamen.*

Tributary	1	2	3	4	5	6	7	8	Total
Annual load ($\times 10^4$ t)	658	1289	517	925	2341	496	509	363	7098
Percentage (%)	9.3	18.1	7.3	13.0	33.0	7.0	7.2	5.1	100

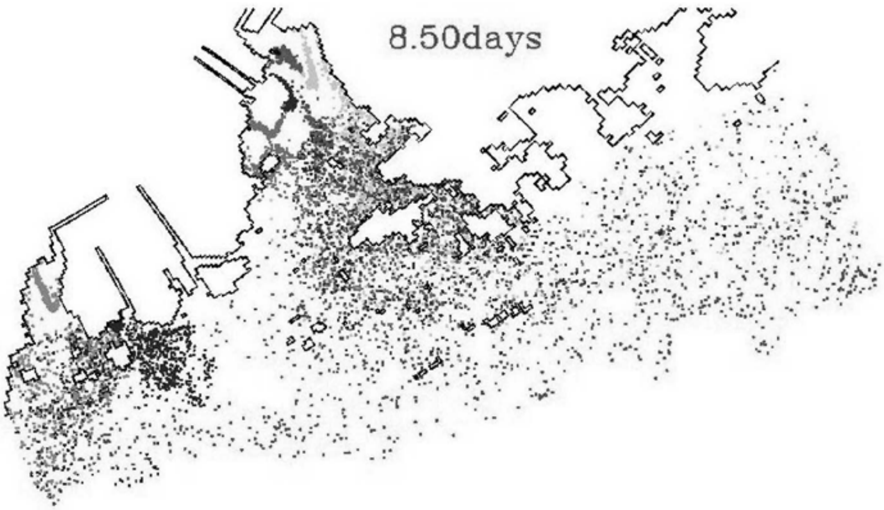
4.3. Flushing time

Using a mean salinity of the PRE derived from 4 cruises in 1987-1988 and an annual average of the total discharge from the four eastern Gates, Wang et al. (1996) estimated that the flushing time for the PRE was 16 days. This is consistent with results from Lagrangian particle tracing experiments using the results from a 3-D POM model (Wong et al., 2004b). The numerical experiments (W. Guan, unpublished data) show a flushing time ranging from about 1 week during the wet season to around 3 weeks in the dry season (Animations 1 and 2).

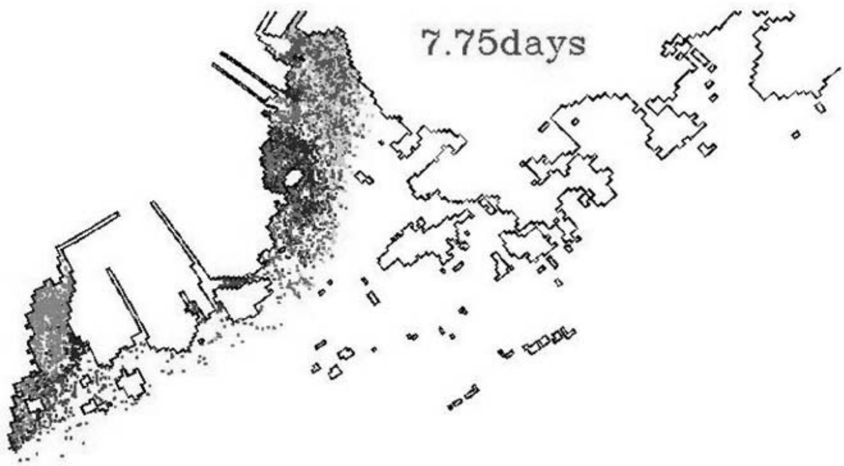
5. SUSPENDED SEDIMENT

5.1. Suspended sediment load

The Pearl River carries annually an average suspended sediment load of about 7.1 $\times 10^7$ tons (PRWRC, 1993; Zhao, 1990). Its distribution through the eight tributaries is given in Table 1. Only the first four distributaries, however, discharge their suspended sediment loads into the PRE directly. The suspended sediment load has a clear seasonal variation affected by the river discharge. About 95% of the sediment load is delivered during the wet season from April to September.



Animation 1. *Wet season tracer experiment under southwesterly monsoon. The tracer particles are released from the eight river Gates of the Pearl River. At each river Gate, there are five layers in the vertical direction from which passive particles are released. One particle at each layer is released every 6 minutes. The particles are released in a 25-hour period beginning at the neap tide and are then followed as they drift out of the estuary. Particles released from different Gates are shown in different colors.*



Animation 2. *Same as Figure 1 but for dry season under the northeasterly monsoon.*

5.2. Flocculation properties of suspended sediment

Suspended sediment in the PRE is composed mainly of clay and silt. Its median diameter (d_{50}) is 2-9 μm in summer and 6-35 μm in winter. Because of the strong seasonal variation in the river runoff and urbanization effect, the estuarine water quality has obvious seasonal and spatial variations, leading to complicated flocculation and settling characteristics (Xia et al., 2004).

During the wet season, water quality differences across the distributaries are insignificant because of the high river discharge. The suspended sediment occurs largely as coagulated flocs with their characteristics significantly influenced by salinity gradient and vertical stratification. For the plume water in the surface layer, the *in situ* median diameter of the suspended sediment is closely related to salinity, but both its *in situ* settling velocity and its apparent density vary little with salinity. The suspended particles at the halocline interface are larger in size than those above or below because of their long residence time there. Both the buoyancy effect and the circulation caused by the vertical salinity stratification easily keep the flocs in suspension at the halocline and enhance the particle collision frequency, favorable for large floc formation.

During the dry season, the seawater intrudes further upstream and the estuarine water is better mixed vertically because of strong winds and low river discharge. However, water quality differences across the distributaries are more evident because impact from urban influences is more important under the low river discharge condition. In the less-polluted freshwater zone of Jiaomen and Hengmen, the suspended matter has high percentage of fine-grained sediments because of wave induced soil erosion of the deltaic plain along the river. Its d_{50} is less than that upstream in the plume water zone of Hengmen. Both the *in situ* d_{50} and the settling velocity increase gradually with salinity. In the heavily polluted freshwater of Humen (near Guangzhou city), the suspended matter has high percentage of organic particles because of urban sewage. Its d_{50} is larger than that upstream. In the brackish water plume water near Humen, the particle size varies little with salinity, while both the settling velocity and the apparent density increase substantially with salinity.

5.3. Temporal and spatial variations of the suspended sediment concentration

The suspended sediment concentration (SSC) in the PRE exhibits large seasonal variation. SSC is higher in the wet season than in the dry season, because 95% of the suspended sediment from the Pearl River is discharged during the wet season. As the four eastern river Gates, carrying discharges from the Pearl River to the PRE, are located at the north and northwest sides of the PRE, the SSC in the PRE decreases both seaward and southeastward.

The SSC is generally less than 0.04 kg m^{-3} in the lower reach of the PRE, except close to the western side of the salinity front where the maximum SSC can reach 0.4 kg m^{-3} near the bottom (L. Dong, unpublished data).

In the wet season, the SSC in the upper and middle reaches of the PRE ranges over 0.04-0.30 kg m^{-3}, generally higher during ebb tides than that during flood (Xu, 1985; Zhao, 1990). Turbidity maximum zones with SSC exceeding 0.1 kg m^{-3} are found near the river Gates, where the SSC exceeds 0.3 kgm^{-3} (Xu, 1985; Zhao, 1990; Deng et al., 2002), and on the western side of the salinity front in the lower layer (Dong et al., 2004).

In the dry season, discharges of both the freshwater and suspended sediment into the PRE are much reduced. The SSC in the upper and middle reaches of the PRE decreases to 0.02-0.19 kg m^{-3} (Xu, 1985). Turbidity maximum zones with SSC greater than 0.1 kg m^{-3} are present near Humen Gate and also near the western channel in the middle reach of the PRE (Xu, 1985; Zhao, 1990).

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CHAPTER 10

WATER QUALITY AND PHYTOPLANKTON BLOOMS IN THE PEARL RIVER ESTUARY

YAN LI, LIANGMIN HUANG, JIANFANG CHEN,
MINGJIANG ZHOU, AND YEHUI TAN

1. INTRODUCTION

Among all rivers in China the Pearl River receives the second largest amount of wastewater (Chen and Heinke, 2002). Pollutants in the Pearl River Estuary (Figure 1) come mostly from river discharges through the four distributaries (or Gates); from wastewater discharges from Shenzhen, parts of Hong Kong, and Macau; and from the extensive shipping activities in the estuary. The Pearl River Water Resources Commission (PRWRC), the environmental Protection Agency of Guangdong, and the Oceanic and Fisheries Administration of Guangdong are the three principal agencies responsible for the water quality of Pearl River and the PRE. They carry out routine monitoring and investigation of the wastewater discharges into the Pearl River and PRE. They are also responsible for measures to protect water resources and to regulate the pollutant discharge. However, there are no regular publications on the pollutant loading of the PRE.

Table 1 lists the total and some organic pollutant loads carried to the PRE from the Pearl River in three separate years. A survey carried out in 1992 (PRWRC, 1993) found the total pollutant discharges into the PRE reached 1.4×10^6 t per year. Among the pollutants, COD is the largest and NH_3-N the second, accounting for 95.5% and 4.3% of the total load, respectively. The pollutant discharges are concentrated, the COD from five major cities together being 43.1% of the whole region. In 2002 and 2003, respectively, a total load of 1.62×10^6 and 1.76×10^6 t of pollutants was discharged into the PRE. The limited data also show an increase in petroleum and phosphate pollutants.

2. POLLUTANT LOADS

Loading of "traditional pollutants" in the PRE come from four major sources: industries and mines, domestic sewage, harbour and shipping operations, and

E. Wolanski (ed.), The Environment in Asia Pacific Harbours, 139–145.

agricultural sources. The annual loads of some of such pollutants in the PRE from the Pearl River are presented in Table 2.

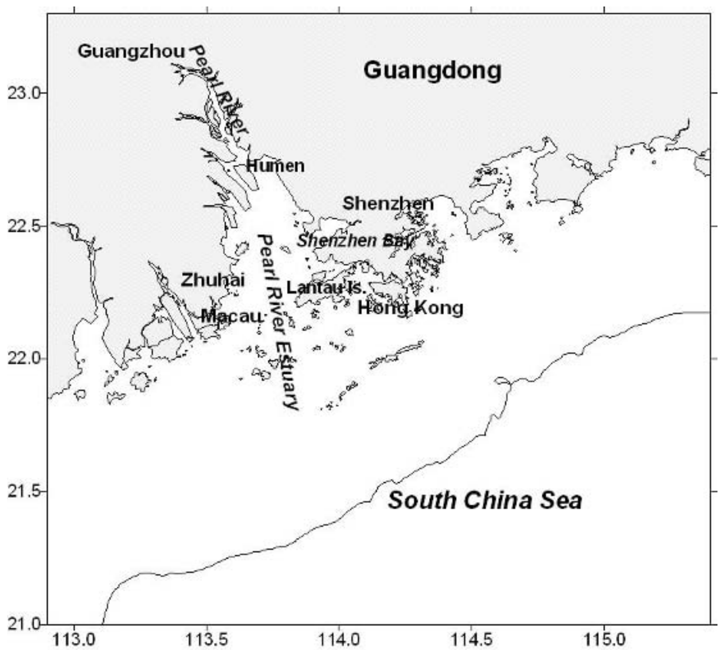

Figure 1. *Map of the Pearl River estuary.*

Table 1. *Total and organic pollutant loads, in 10^6 t y^{-1}, from the Pearl River to the PRE. (Data source—a: PRWRC, 1993; b: SOA, 2004).*

	1992[a]	2002[b]	2003[b]
Total pollutants	1.40	1.62	1.92
COD	1.34	1.15	1.76
Petroleum		14.0	47.0
NH_3-N	60.2		
Phosphate		15.0	24.0

Table 2. *Loading of pollutants from the Pearl River to the PRE (10^3 t y^{-1}). (Source: Record of Estuaries and Bays in China, Volume 14, 1998).*

Hg	Cd	Cu	Pb	Zn	Cr	As	Phe nol	Ni	Hydr o-carbo n	Chlo ride	666	DDT
0.01	0.3	0.2	3	15	1.8	1	1.5	1	49	0.8	0.6	0.2

3. HEAVY METALS

Available data on the concentration of dissolved heavy metals in the PRE were obtained from two major surveys. One survey in the period late August to late September 1996 covered a spring tide and a neap tide. The other survey was carried out in July 1999. The results are presented in Table 3.

Table 3. *Concentration of dissolved heavy metals in the Pearl River Estuary ($\mu g\ l^{-1}$).*

	1996 (later August to later September)			1999 (July)	
	range	Average		Range	Average
		Spring tide	Neap tide		
Hg	0.010-0.280	0.036	0.06		
As	0.05-2.38	1.29	1.40		
Cu	0.31-3.9	0.95	0.85	0.03 to1.48	0.39
Zn	1.0-5.4	2.8	2.1	0.88 to 44.6	20.8
Pb	0.12-0.97	0.41	0.34	0.03-3.30	0.77
Cd	0.015-0.613	0.076	0.061	0.005~0.356	0.066
Cr	0.070-0.612	0.253	0.267	0.06 to 0.46	0.23
Ni	0.23 - 3.5	0.66			
Se	0.082-0.19	1.28	0.132		

4. OIL AND GREASE

Pollution by oil and grease in the PRE derives mostly from shipping, fishing vessels, and discharge of wastewater. Due to evaporation, photolysis and biodegradation, the oil and grease concentration fluctuates seasonally (Li and Cai, 1998). The average concentration of the oil and grease for the PRE was 0.046 mg l^{-1} in the late 1970's and it increased to 0.079 mg l^{-1} in the middle 1980's (Editorial Board of "Record of Estuaries and Bays in China", 1998).

The 1996 wet season survey revealed an oil and grease concentration ranging from 0.018 to 0.108 mg l^{-1} with the average value of 0.033 mg l^{-1}. Most of the sites had concentration in the range of 0.02 to 0.04 mg l^{-1}. High concentrations were found in waters east of Lantau Island. The July 1999 survey found the concentrations of oil and grease in the PRE varying between 0.02 and 0.38 mg l^{-1} with an average value of 0.096 mg l^{-1}, higher in general than 1996 but still lower than the first category criterion for seawater quality standard in China (0.5 mg l^{-1}).

The dry season survey in November 1996 found the oil and grease concentration ranging from 0.014 to 0.202 mg l^{-1} with an average value of 0.040 mg l^{-1}. Again, higher concentrations were found over the navigation paths south of Hong Kong.

5. NUTRIENTS

The most prominent characteristic of the water quality of the PRE is the high nitrate to phosphate ratio. In the upper estuary, this ratio (N:P) can be up to 200:1 in the surface layer (Zhang et al., 1999; Yin et al., 2000; Chen et al., 2004; Cai, et al., 2004).

Concentration of the dissolved inorganic nitrogen (NO_3-N, NO_2-N and NH_4-N) in the PRE decreases dramatically from >100 molL^{-1} at the freshwater end to <1 molL^{-1} at its very sea end, while that of the dissolved inorganic silica (SiO_3-Si) also decreases dramatically from >140 molL^{-1} in freshwater area to <1□molL^{-1} in coastal water. Both the nitrate (NO_3-N) and the silicate (SiO^3-Si) show very good linear regression relationship with the salinity. The effective concentration of the river water for nitrate is found in the range of 90.6 to 115.2 molL^{-1} and that for silicate in the range of 145~169 molL^{-1}.

The dissolved inorganic phosphorus (PO_3-P), however, behaves non-conservatively and remains at a level of about 1 molL^{-1} throughout the whole PRE. This is due to a strong "buffering" action related to fast regeneration from particulate phosphorus. There is a shift in potential nutrient limitation across the estuarine plume. It changes from P limitation in the estuary to P and Si co-limitation at the frontal zone of the river plume, and then to N limitation offshore from the river plume (Zhang et al., 1999; Yin et al., 2001 and 2004). Apparently the upper PRE is dominated by aerobic respiration (Zhai et al., 2005). Within this aerobic respiration system, particulate phosphorous is released following the consumption of dissolved oxygen and the release of carbon dioxide.

6. PHYTOPLANKTON COMMUNITY

6.1. Species diversity and phytoplankton community structure

About 300 phytoplankton species have been recorded for the PRE, including about 200 diatoms, more than 70 dinoflagelates, and over 20 other species (Lin et al., 1985; Guo et al., 1994; Huang et al., 2004).

The species composition is quite different between years and seasons. A marked seasonal succession of dominant species has been observed, and the number of species is not much different. In recent years *Skeletonema costatum* is often high in density and can become the dominant species in any season of the year. The indices of species diversity and evenness vary significantly in different seasons and regions, with range from 1.2 to 3.9 and 0.3 to 0.8, respectively. The species diversity shows typical subtropical features (Huang et al., 1997).

The phytoplankton community in the PRE consists mostly of widely distributed coastal species and some estuarine species. Coastal, or marine, species accounts for more than 90% of the total species. In the upper part of the PRE, such as the waters near Humen, the dominant species are estuarine species *Melosita granulata* (accounting for more than 50% of the total amount) and the widely distributed species *Skeletonema costatum*. In the outer estuary area, besides the widely distributed species *Skeletonema costatum,* some species of the *Chaetoceros* genera often dominates. The dominant phytoplankton species in the PRE vary from one year to another, but *Skeletonema costatum* has become more dominant in recent years.

6.2. Phytoplankton abundance and seasonal succession of dominant species

The annual average of phytoplankton abundance in the PRE ranges between 5.4×10^6 and 6.0×10^9 cell m^{-3} (see Table 4). The highest abundance (10.6×10^9 cell m^{-3}) occurred in winter of 2003, and the lowest (1.2×10^5 cell m^{-3}) in spring of 1987. There is a trend of increasing abundance in recent years. High abundance normally occurs in the river plume front area, such as the waters between the Lantau Island and the middle reaches of the PRE.

Table 4. *Annual variations of phytoplankton abundance and dominant species. (Source: Lin and Yuan, 1985; Lin, 1989; Huang et al., 1995, 1997 and 2004; Guo et al., 1994).*

Year	Species number	Abundance (cells m^{-3})	Survey waters	Survey seasons
1980-1981	224	1.9×10^7	Pearl River estuary	spring, summer, autumn, winter
1987-1988		5.4×10^6	Pearl River estuary	spring, summer, autumn, winter
1989-1990	237	2.1×10^7	Pearl River estuary	spring, autumn
1996-1997	266	2.8×10^8	Pearl River estuary and Hong Kong's adjacent waters	spring, summer, autumn, winter
1999-2001		3.9×10^8	Pearl River estuary and Hong Kong's adjacent sea-waters	summer, winter
2002-2003	163	6.0×10^9	Pearl River estuary	spring, summer, autumn, winter

Because of the distinct seasonal characteristics of the hydrography in the PRE, seasonal succession of the dominant phytoplankton species in the estuary is evident. For example, in the middle reach of the PRE the estuarine species *Melosita granulata* dominates in the wet season when the area is low in surface salinity, while the widely distributed coastal species *Skeletonema costatum* dominates in the dry season when the salinity there is high (Huang et al., 1995). Over the last twenty years there is a significant change of dominant species in the PRE. In the 1980s, the dominant species were: in spring, *Skeletonema costatum* and *Coscinodiscus centralis*; in summer, *Chaetoceros affinis* and *Sk. costatum*; in autumn *Sk. Costatum* and *Ditylum brightwellii*; and in winter, *Thalassiosira subtilis* and *Thalassiothrix frauenfeldii*. The dominant species in the 1990s were: in spring, *Chaetoceros dentieulatus* and *Chaet. affinis*; in summer, *Chaet. pseudocurvisetus* and *Skeletonema costatum*; in autumn, *Skeletonema costatum* and *Bellerochea malleus*; and in winter, *Chaet. siamense* and *Chaet. pseudocurvisetus*. In recent years (2002-2003) the dominant species in the PRE have again changed: in spring, *Nitzschia delicatissma* and *Staurastrum polymorphum*; in summer, *Nitzschia delicatissma* and *Skeletonema costatum*; in autumn, *Skeletonema costatum* and *Chroococcus splendidus*; and in winter, *Skeletonema costatum* and *Staurastrum polymorphum*.

7. HARMFUL ALGAL BLOOMS (HAB)

The fast human population growth and the rapid expansion of the mariculture industry have exerted much stress on the coastal environment off the Pearl River delta. As in elsewhere of China, there is a rapid increase of the frequency of harmful algal bloom (HAB) in this area over the last two decades (Su, 2001; Zhou et al., 2001). Most of these HAB events happened in Hong Kong waters and in bays further east of Hong Kong (e.g., Yan et al., 2001). However, outbreaks of HAB in the PRE itself also happened more frequently in recent years, especially in Shenzhen Bay at the east side of the middle PRE (Table 5).

Table 5. *Harmful algal blooms in the Pearl River estuary since 2000.*

Date	Area and range	Duration (days)	Red tide/bloom species	Location
January, 2000	5 km^2	3	*Mesodinium rubrum*	Pearl River estuary
February, 2001	50 km^2	17	*Gymnodinium samtenatum*	Shenzhen Bay
February, 2001	10 km^2	6	*Mesodinium rubrum*	Pearl River estuary
March, 2001	8 km^2	9	*Thalassiosira* sp.	Shenzhen Bay
January, 2002	20 km^2	4	*Thalassiosira* sp.	Shenzhen Bay
May, 2002	12 km^2	13	*Skeletonema costatum*	Shenzhen Bay
June, 2002	150-300 km^2	5	*Skeletonema costatum* *Gyrodinium instriatum*	Pearl River estuary
April, 2003	350 km^2	1	*Skeletonema costatum* *Cheatoceros* sp	Pearl River estuary
May-June, 2003	5-12 km^2	1-3	*Skeletonema costatum*	Shenzhen Bay
June, 2003	40 km^2	1	*Heterogma akashiwo*	Shenzhen Bay
January, 2004	42 km^2	4	*Skeletonema costatum* *Gyrodinium instriatum*	Pearl River estuary
January-February, 2004	20-25 km^2	50	*Phaeocystis pouchetii*	Pearl River estuary

HABs in the PRE are mostly caused by dinoflagellate species, accounting for 72.7% of the total events. Among these the most frequent one is *Noctiluca scintillans,* responsible for 50.9% of the total events, and the next is *Gymnodinium,* accounting for 12.7%. Diatoms are also important for HABs in the PRE, accounting for 23.6% of the total. The blooms are mostly of single-species type and less of multi-species type. More than 40 species of organisms have been identified for HABs in the PRE. The principal species include *Noctiluca scintillans, Skeletonema costatum, Chattonella marina, Rhizosolenia alata f. gracillima, Gonyaulax polygramma, Pseudonitzschia pungens, Thalassiosira subtilis, Scrippsiella trochoidea, Prorocentrum sigmoides,* and *Gymnodinium mikimotoi*.

The high concentration of inorganic nitrogen, i.e., eutrophication, is the direct cause for the increase of HAB frequency in the PRE. However, phosphate and turbidity are both important factors influencing the HAB occurrence (Yin et al., 2001; Huang et al., 2004). Nutrients are high in the northern part of the estuary, but the chlorophyll *a* concentration is generally low there because of the high turbidity. There are two relatively high concentration of chlorophyll *a* in the estuary, one near the Shenzhen Bay and the other southwest of Hong Kong. The concentration in the two areas is in general relatively low. In fact, HAB events in the PRE often occur at these two sites. However, relationship between the HAB and the environmental conditions in the PRE has not yet been established.

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CHAPTER 11

POLLUTION STUDIES ON MANGROVES IN HONG KONG AND MAINLAND CHINA

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1. INTRODUCTION

Mangroves are unique inter-tidal wetland ecosystems found along the sheltered estuarine shores of tropical and subtropical regions, and provide a wide variety of goods and services to people. Mangroves act as a transit zone between terrestrial and marine environments, contaminants transported from rivers and land-based sources as well as those in tidal water are therefore accumulated in mangrove ecosystems. In the past, mangroves especially those in developing countries have been used as convenient waste disposal sites (Clough et al., 1983) and are sinks or reservoirs of various man-made pollutants (Harbison, 1986; Tam and Wong, 1995a, 1999a; Zheng et al., 2000). Since the 1970s, researchers worldwide have published more than 3,500 SCI papers on various aspects of mangroves, ranging from mangrove distribution and geographical patterns, climatic changes and sea level rise, general ecology, community structure and function, population dynamics, physiological and molecular studies of individual species, fisheries, aquaculture and other economic values, restoration, management and conservation of mangroves. However, environmental pollution and its impacts on mangrove ecosystems especially those in China have received little attention.

Mangroves are under increasing pollution pressure from human activities because of rapid industrialization and urbanization in our coastal areas. Refuse disposal, sewage discharge, urban emission, and accidental spillage of toxic pollutants are significant anthropogenic inputs. For instance, Deep Bay at the mouth of the Pearl River delta has the sixth largest mangrove forest in China, but mangroves around Deep Bay including Mai Po RAMSAR in Hong Kong and Futian in Shenzhen (Figure 1) have suffered from discharge of untreated domestic sewage, livestock and industrial wastewater in the catchment. Since the 1980s, Deep Bay region especially the Shenzhen Economic Special Zone has become one of the rapidest developing cities and the mangrove becomes the only one located at the

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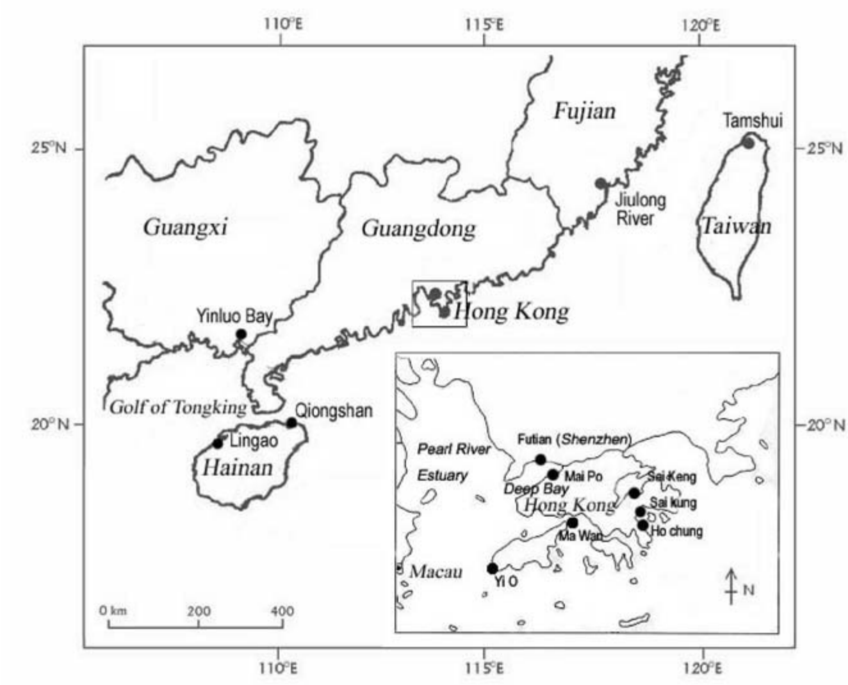

Figure 1. *Geographical location of various mangrove swamps in Hong Kong, Macau, Taiwan and Mainland China.*

heart of a modern city in China. The contaminants in the polluted water including nutrients, heavy metals and toxic organic pollutants are adsorbed onto suspended particles and subsequently accumulated in the mangrove sediments which act as a sink (Tam and Wong, 1993, 1995a, 1999b; Tam et al., 2001).

On the other hand, evidence suggests that mangrove forests are generally nutrient deficient, and mangrove plants such as *Bruguiera gymnorrhiz*a and *Avicennia marina* have shown positive growth responses to added nitrogen (e.g. Naidoo, 1990; Wong et al., 1997a; Ye and Tam, 2002). Recent studies suggest that mangrove wetlands could be employed for treating domestic, municipal, livestock, and shrimp pond effluents, and solve water pollution problem. The potential of developing a sustainable mangrove wastewater treatment facility should not be overlooked. In this paper, the degree of pollution including nutrients, heavy metals and persistent organic pollutants in mangrove ecosystems in China, in particular, Hong Kong, Shenzhen and Fujian were reviewed. The feasibility of using mangrove wetlands as a sustainable wastewater treatment facility was discussed.

2. NUTRIENT CONCENTRATIONS IN MANGROVE ECOSYSTEMS

Mangroves in various geographical locations in China have accumulated different amounts of organic matter, total nitrogen and phosphorus in their sediments (Table 1). The anthropogenic inputs of nutrients are obvious in cities around Pearl River Estuary such as Mai Po in Hong Kong and Futian in Shenzhen (Figure 1). Domestic, industrial and agricultural inputs from the Pearl River significantly enhanced the nutrient concentrations in mangrove sediments (Richardson et al., 2000). Similar findings emerge from studies of the Tamshui Estuary mangroves in Taiwan that have been polluted by municipal sewage, and the ammonium and nitrate concentrations, at the range of 0.15 to 17.10 and trace to 2.54 mg N kg^{-1} sediments respectively, were much higher than values reported elsewhere (Chiu and Chou, 1991; Chiu et al., 1996). It is also clear that the sediment pollutant concentrations in Mai Po tidal shrimp pond were significantly lower than that in the foreshore mangrove areas. The pond operates with a sluice gate that is opened only during high tide to recruit juvenile shrimps, and the pond is drained at harvest to collect the shrimps. The pond was thus shielded by limited exposure to the polluted water from the Pearl River Estuary resulting lower degree of contamination.

In addition, differences along tidal positions are also significant within a mangrove swamp. Organic matter, nitrogen and phosphorus concentrations in surface sediments were higher landward than seaward of Sai Keng mangrove swamp in Hong Kong (Figure 1). Total N in mangrove sediments at the seaward site was <0.03% while that landward was >0.1%; organic matter content landward was >30% while seaward was 7% (Tam et al., 1993; Tam and Wong, 1998). Similar variations were found in the mangroves of Futian (Tam et al., 1995). The relatively high concentrations of nutrients at landwards were due to less frequent tidal exchange so litter was more likely to be accumulated and decomposed, and nutrients were returned back to the sediments. Freshwater input, terrestrial runoff from the abandoned paddy fields, leaching of fertilizers from agricultural fields, discharge of untreated domestic and livestock waste, as well as leachate from dumping sites at the back of mangroves also increased the organic carbon and nutrient content. Localized "hot spots" of contamination due to anthropogenic inputs have been found within a mangrove swamp near human settlements. Significant positive correlations found among organic matter, nitrogen and phosphorus in mangrove sediments give further evidence that the input sources of nutrients and organic matter were anthropogenic (Tam et al., 1993; Tam and Wong, 1998).

Nutrients entering a mangrove ecosystem are quickly assimilated because of the high productivity and demand of nutrients for plant growth, rapid microbial activities and turnover. The turnover periods of nitrogen and phosphorus in mangrove communities dominated by *Kandelia candel* (20 years old) were 7 and 10 years, respectively, which were shorter than that in *Bruguiera sexangula* community with 12 years turnover periods for both N and P (Lin, 1999). The concentrations of nutrients and organic matter in sediments reflect the net results of interactions among many biogeochemical factors, including pollution inputs, plant uptake, released from litter decomposition and microbial transformation, tidal flushing, and leaching.

Table 1. *Concentrations in % dry sediment (mean and range in bracket) of organic matter and nutrients in surface sediments of mangrove swamps in China (Kc: Kandelia candel, Ac: Aegiceras corniculatum, Am: Avicennia marina, Bg: Bruguiera gymnorrhiza, Bs: Bruguiera sexangula, Rs: Rhizophora stylosa; Sa: Sonneratia apetala; NR: not reported).*

Location	Dominant species	Organic matter	Total nitrogen	Total phosphorus	References
Taiwan					
Tamshui Estuary	Kc	(5-10)	(0.02-0.27)	NR	Cheng, 1995; Chiu et al., 1996
Fujian					
Overall	NR	3.15	0.12	0.132	Lin ,1999
Jiulong River Estuary	Kc	1.56 (0.19-3.24)	0.34	0.050	
Guangdong					
West	NR	3.18 (1.24-11.69)	0.09 (0.02-0.32)	0.075 (0.02-0.18)	Lin, 1999
East	NR	4.37 (0.71-4.98)	0.14 (0.11-0.17)	0.057 (0.04-0.064)	
Pearl River Estuary	NR	3.05 (2.49-3.60)	0.19 (0.16-0.24)	0.141 (0.135-0.15)	
Futian, Shenzhen	Kc, Ac, Am and Bg	3.27 (0.53-6.29)	0.16 (0.02-0.29)	0.137 (0.021-0.198)	Tam et al., 1995; Zhang et al., 1998
Hong Kong					
Mai Po: tidal ponds	Kc and Ac	2.10 (1.92-2.33)	0.10 (0.09-0.11)	0.049 (0.047-0.050)	Lau and Chu, 1999
Mai Po: overall	Kc, Ac and Am	9.48 (7.85-11.05)	0.21 (0.19-0.23)	0.139 (0.128-0.168)	Tam and Wong, 1999a
Sai Keng	Kc, Ac and Am	7.61 (4.37-14.2)	0.17 (0.09-0.25)	0.034 (0.016-0.067)	Tam and Wong, 1998
Ho Chung	Kc and Ag	12.46 (10.78-15.57)	0.38 (0.35-0.56)	0.075 (0.066-0.081)	Tam and Wong, 1999a
Guangxi					
Overall	Am, Kc, Ag, Bg and Rs	2.92 (4.38-0.67)	0.10 (0.02-0.25)	0.059	Lan et al., 1994
Yinluo Bay, Guangxi	Rs	13.43	0.41	0.062	Lin, 1999
Hainan Island					
Solonchak	Bs	4.48	0.31	0.053 (0.016-0.096)	Liao, 1990
Qiongshan	NR	9.88	0.20	0.029	Lin, 1999
Lingao	NR	1.64	0.08	0.023	

3. MANGROVE AS SINKS FOR HEAVY METALS

Heavy metal contamination in aquatic environments is of critical concern due to their toxicity, persistence and bioaccumulation problems. The impact of human perturbation is most strongly felt by estuarine and coastal environment adjacent to urban areas, in particular, the inter-tidal mangrove wetlands. Heavy metals from incoming tidal waters and freshwater sources are rapidly removed from the water body and deposited onto the sediments. Mangrove sediments being anaerobic and reduced, rich in sulfide, organic matter and iron acted as sinks for heavy metals, resulting in elevated concentrations (e.g., Harbinson, 1986; Silva et al., 1990; Lacerda et al., 1993; Tam and Wong, 1993, 1995a and b, 1999a, 2000). Studies in China show that the concentrations of heavy metals in mangrove sediments vary spatially and are closely related to the degree of pollution (Table 2). The anthropogenic inputs contributed more to the variations of heavy metals in mangroves than factors such as the characteristics of sediments, frequency and duration of tidal flooding. For instance, the heavy metal concentrations in western side (Mai Po and Inner Deep Bay) were higher than those obtained in the eastern side of Hong Kong. The mangroves located in the mudflat region near the polluted rivers in Futian had accumulated more heavy metals than other locations in the same mangrove swamp. The vertical distribution of heavy metals in sediment cores also reveals that the anthropogenic pollution in Futian is acute; the concentrations in the sediments are the highest near the surface (Table 3). The rapid urbanization and population growth of Shenzhen City, the industrialization and development of other cities around Pearl River delta have led to increasing discharges of contaminated wastes (Wang et al., 2003).

The heavy metal contamination in most mangrove sediments in China varied from slight to moderate with some hot spots where severely contaminated. In terms of toxicity, most mangrove sediments had Cu, Zn, Pb and Cd concentrations higher than the ER-L (Effects range-low) but lower than the ER-M (Effects range-moderate) values suggested by Long et al. (1995). This implied that heavy metal contamination might pose some adverse effects but the effects should not be frequent and may also not be at high risks.

Most metals accumulated in mangrove sediments are not bio-available (Tam and Wong, 1993, 1995a). The amounts of heavy metals in the exchange and carbonate portions are small compared to that of the total metal concentration. For instance, Tam and Wong (1996a) found that only 2.36% Cu, 3.79% Zn, 0.89% Pb and 0.09% Cr in Futian mangrove sediments are bio-available (i.e. could be extracted by 1M ammonium acetate at pH 4). Chiu and Chou (1991) also reported that the exchangeable fraction of Cu, Zn, Pb, Cd, Ni and Cr in Tamshui mangrove sediments is 0.24, 1.38, 1.72, 3.17, 0.43 and 0.09% of total heavy metals, respectively. The anaerobic nature of the mangrove sediment, usually at negative redox potential and the presence of high amounts of sulfide, iron and organic matter decrease the metal solubility. Moreover, when the sediments are submerged at high tide, Fe and Mn oxides are converted into hydrated forms and provide a large surface area for reaction with metal ions, leading to a reduction in the bio-availability of heavy metals.

4. ORGANIC POLLUTION IN MANGROVE ECOSYSTEMS

The discharge and dumping of wastes not only cause eutrophication and heavy metal pollution, high levels of anthropogenic toxic organic pollutants including polycyclic

Table 2. *Concentrations (mg kg^{-1} dry sediments) of heavy metals in surface sediments of mangrove swamps in China (mean and range in bracket are shown) (same legends as in Table 1).*

Location	Cu	Zn	Pb	Cd	Ni	Cr	References / Remarks
Tamshui Estuary, Taiwan	44.1 (17.3-61.3)	148.8 (99-265)	66 (58-71)	3.62 (3.15-3.90)	102.5 (91.8-109.2)	NR	Chiu and Chou, 1991
Jiulong River Estuary, Fujian							
Kc	29.7	111.0	18.3	0.09	16.9	4.73	Lin, 1999
Ac	26.6	138.0	101	0.12	NR	NR	
Futian, Shenzhen							
Mudflat (near polluted river)	80.3 (44-124)	207.7 (129-333)	37.9 (22-50.)	1.07 (0.7-1.5)	NR	NR	Zhang et al., 2004
Mixed mangrove: Kc, Ac, Am & Bg	41.1 (16-308)	146.1 (53-423)	35.4 (0.1-63)	2.96 (0.3-8)	NR	34.2 (6.8-57)	Tam et al., 1995
Am	38.3	114.0	28.7	0.14	25.0	7.97	Lin, 1999
Sa	45.6	125.1	69.9	NR	62.8	58.1	Wang et al., 2003
Hong Kong							
Tidal ponds, Mai Po	36.5 (32-41)	155.5 (125-183)	NR	0.65 (0.6-0.76)	27.1 (20-34.4)	NR	Lau and Chu, 1999
Western side (Deep Bay)	64.7 (45-82)	221.7 (147-247)	75.4 (65.3-82.0)	1.95 (0.4-3.2)	30.1 (24.3-36.9)	41.3 (39.2-43.5)	Tam and Wong, 2000
Northeast and Sai Kung	15.4 (0.5-58)	59.3 (12-140)	28.9 (7.9-45.6)	0.27 (0.01-1.9)	6.41 (0.9-11)	3.52 (0-17)	
Yinluo Bay, Guangxi (Rs)	18.9	46.6	10.0	0.07	14.6	9.27	Lin, 1999
Classification of contaminated sediments in Hong Kong (EPD, 1992)							
A	<10	<70	<25	<0.1	<15	<25	Uncontaminated
B	10-54	70-150	25-65	0.1-1.0	15-35	25-50	Slightly contaminated
C	55-64	150-200	65-75	1.0-1.5	35-40	50-80	Moderately contaminated

Table 3. *Vertical distribution of heavy metals in different sediment layers of mangrove swamps in China (concentrations in mg kg^{-1} dry sediments are shown; NR: Not reported; Ac: Aegiceras corniculatum; Am: Avicennia marina; Sa: Sonneratia apetala).*

Location and depth in cm	Cu	Zn	Pb	Cd	Ni	Cr
Tamshui, Taiwan (Chiu and Chou, 1991)						
0-10	17.5	59.9	17.3	0.23	4.53	NR
10-20	15.7	50.6	20.3	0.27	4.37	NR
20-30	11.8	32.3	19.0	0.27	4.20	NR
30-40	10.3	22.5	15.7	0.22	3.20	NR
40-50	6.1	15.8	13.7	0.22	2.70	NR
Jiulong River, Fujian, Ac community (Zheng et al., 1996; Lin, 1999)						
0-30	26.6	138	101	0.12	NR	NR
30-60	27.3	143	89	0.12	NR	NR
60-90	26.0	147	88	0.17	NR	NR
Futian, Shenzhen, Sa community in early 2000s (Wang et al., 2003)						
0-30	45.6	125.1	69.9	NR	62.8	58.1
30-60	25.1	95.4	56.4	NR	49.9	34.2
60-90	10.7	81.8	51.9	NR	33.6	23.8
Futian, Shenzhen, Am community in mid 1990s (Lin, 1999)						
0-30	38.3	114	28.7	0.14	25.0	7.97
30-60	35.7	106	33.8	0.13	19.3	6.66
60-90	23.6	76	31.8	0.12	18.4	6.05
Mai Po mudflat, Hong Kong (Ong Che, 1999)						
0-10	67.1	222.0	135.3	1.2	70.8	33.0
21-30	60.3	213.3	120.3	1.2	66.9	24.6

aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), and organochlorine pesticides (e.g. DDTs) are also found in coastal environments. Other human activities such as oil spills, ship traffic, atmospheric fallout of vehicle exhaust and industrial stack emission have caused significant accumulation of PAHs and PCBs in marine environments (Simpson et al., 1996). Trace toxic organic contamination have received less attention than the trace metals in mangrove sediments especially in China. The available data on the concentrations of toxic organic pollutants including PAHs, PCBs and DDT in mangrove swamps and coastal environments in China show a large scatter (Table 4). Within the same swamp, the concentration ranges of total PAHs (summation of 16 USEPA priority PAH compounds) were very wide and hot spots of contamination in a relatively clean area were often recorded. The local sources of contamination of PAHs in Sai Keng, Sai Kung, Ho Chung and Ma Wan mangroves in Hong Kong (Figure 1) included leachate from a garbage station, livestock waste from a nearby village, and oil spillage from fishery activities (Tam et al., 2001; Ke et al., 2005). The amounts of PAHs detected in Xiamen mangrove sediments are related to urban runoffs, sewage outfalls and wastewater discharges in addition to non-point sources of contamination from petroleum related activities including heavy commercial boating

Table 4. *Concentrations of toxic organic pollutants (ng g^{-1} dry weight) in surface sediments of mangrove swamps and other coastal habitats (NR: not reported; ND: not detected).*

Location	Total PAHs	Total PCBs	Total DDTs	References
Mangrove sediments				
Mai Po and Deep Bay, Hong Kong	1945 (685-4680)	2.90 (1.47-8.13)	NR	Liang et al., 1999; Tam et al., 2001
Contaminated site, Hong Kong	6186 (1273-11098)	7.21 (5.50-25.10)	NR	Tam et al., 2001; Tam and Yao, 2002
Clean site, Hong Kong	1044 (356-1811)	0.88 (0.11-1.60)	NR	Tam et al., 2001
Futian, Shenzhen	409 (238-726)	NR	NR	Zhang et al., 2004
Jiulong River, Xiamen	334 (59-1177)	NR	NR	Maskaoui et al., 2002
Coastal sediments other than mangroves				
Hong Kong Mai Po mudflat	770 (630-960)	5.24 (4.7-6.4)	NR	Zheng et al., 2002; personal communication
Pearl River Estuary	630 (323-1006)	11.37 (5.5-22.3)	26.88 (2.6-115.6)	Yang, 2000; Mai et al., 2002
Daya Bay, Guangdong	481 (115-1134)	NR	NR	Zhou and Maskaoui, 2003
Jiulong River, Xiamen	111 (424-1520)	8.37 (1.60-14.29)	45.12 (8.61-73.70)	Yuan et al., 2001
Hsin-ta, Taiwan	98-2048			Fang et al., 2003
Sediments in other countries				
Puerto Rico mangrove sediments	1820 (500-6000)	NR	NR	Klekowski et al., 1994
Caribbean Island mangrove sediments	502 (103-1657)	NR	NR	Bernard et al., 1996
Low to high polluted coastal sediment in Gulf of Mexico, USA	3-3230	2-134	ND-24.1	Wade et al., 1993
Low to very high polluted coastal sediment in Pacific Coast, USA	0.1-20000	0.1-2000	NR	Brown et al., 1998
Toxicity guidelines (Long et al., 1995)				
ER-L	4092	22.7	1.58	Effects range-low
ER-M	44792	180	46.1	Effects range-median

and industrial shipping activities, oil spill and release (Maskaoui et al., 2002). Similarly, the highest concentration of PAHs recorded in Daya Bay sediments is caused by the large amount of soil runoff and sewage discharges from a densely populated area with intense agricultural activities and a thriving fish mariculture (Zhou and Maskaoui, 2003).

In terms of the PAH composition in Hong Kong mangrove sediments, all of the 16 USEPA priority target PAH compounds are present, and the high molecular weight PAHs (with 4 and more benzene rings) are more dominant (Bernard et al., 1996; Yang, 2000; Tam et al., 2001; Maskaoui et al., 2002). The diagnostic ratios of individual PAH compounds including the phenanthrene / anthracene, fluoranthene / pyrene and benzo[*a*]pyrene / benzo[*ghi*]perylene suggested that the PAH burden in mangrove sediments in Xiamen as well as in Hong Kong came from several sources, a combination of fuel-combustion (pyrolytic) and crude oil (petrogenic) contamination (Tam et al., 2001; Maskaoui et al., 2002). These results suggest that there are potentially many different sources of contamination in mangrove habitats, and local deposition is more important inputs than long-range atmospheric transportation.

Data on the degree of PCB and DDT contamination in mangrove sediments in China are scarce (Table 4). Published information on the concentrations in sediments collected from the two Estuaries, Pearl River and Jiulong River, suggested that PCB contamination at those sites was not a major environmental concern. The low PCB contamination was probably the result of limited use of PCB in China as compared with the developed industrial countries. However, some hotspots had very high PCB concentrations suggesting localized contamination were found in Hong Kong and Mainland China. DDT is the major chlorinated pesticide found in China mangrove sediments with concentrations higher than other countries. The use of DDT was heavy and widespread in China during the 1960s and 1970s. Although its usage was banned in the early 1980s, residual DDT in the sediment from previous usage is still dominant as degradation is slow under the anaerobic condition in estuarine sediments.

The PAH and PCB contamination in mangroves in China, similar to heavy metals, is considered as slight to moderate. The total PAHs and PCBs concentrations in mangrove sediments were lower than the ER-L values that are deemed to show toxic biological effects for benthic organisms (Long et al., 1995). However, exceptionally high concentrations of total PAHs and PCBs were detected in some sediment samples. The limited information on total DDT concentrations in coastal sediments in China showed that they were above the ER-L value, suggesting that the DDT in the sediments were likely to pose detrimental biological effects. More attention must be paid on the hotspots of contamination as well as DDT in the sediments.

5. POLLUTION IN SOUTH CHINA: PEARL RIVER ESTUARY

The Pearl River is one of the largest rivers with a great number of tributaries and streams runs through Guangzhou city and the delta, extending to Deep Bay in Hong Kong. Every year, approximately 3.11 x 10^{11} m^3 y^{-1} freshwater flows into the South

China Sea via eight major outlets (Mai et al., 2002). The Pearl River delta is the most developed region in China. Its population has increased from 50.6 millions in 1979 to 69.0 millions in 1996, while the GDP experienced a ~13% annual increase since 1997, leading to significant pollution. Moreover, only a small portion of the sewage generated from the densely populated cities around the Pearl River such as Hong Kong, Shenzhen, and Guangzhou is completely treated, the untreated and partially treated sewage effluents become major contamination sources. The untreated wastewater from industrial and municipal activities, upstream runoff from mining sites and agricultural land, and deposition of air pollutants all contributed to the increase of pollutants, including nutrients, heavy metals and toxic organic pollutants in sediments. The concentrations of heavy metals and trace organic pollutants in the coastal and bottom sediments, including mangroves, of the Pearl River Estuary has increased over the last 20 years (Yang et al., 1997; Kang et al., 2000; Li et al., 2000; Yang, 2000).

The depth profiles of heavy metals and the lead $^{206}Pb/^{207}Pb$ isotope ratios in marine sediment cores collected at different regions in the Pearl River Estuary showed that anthropogenic inputs of heavy metals were from recent rapid industrial development in the surrounding region (Li et al., 2001) except Hong Kong. Kang et al. (2000) reported that the highest PCB concentration (485 μg Kg^{-1} dry weight) and some hotspots were found in the sediments taken from the Pearl River section running through the industrial district of Guangzhou. Hong Kong was heavily industrialized since1970-1980 and had over 2,000 major and 200,000 minor industries, in particular the metal-related industries such as electroplating, printed circuit board, ship building and boat yards, can production, enamelware, batteries, paints, etc. These industries produced significant pollution inputs. Many of these industries have moved to cities around the Pearl River delta since 1980s where land and labor costs were cheaper. The vertical distribution of PAHs in sediment cores collected from Ma Wan mangrove swamp in Hong Kong (Figure 1) showed that the highest concentration was in the bottom layer (10-20 cm) reflecting serious contamination that occurred from 1960 to 1980, and the concentrations in the 5-10cm sediment layer (representing the period of 1980 to 1990) decreased substantially (Ke et al., 2004, 2005). These results confirmed that the source of pollution from industrial discharges in Hong Kong has migrated to other cities in the Pearl River delta.

The heavy metal and trace organic pollution in Shenzhen and Deep Bay regions, although classified as slight to moderate contamination and the values were lower than the ER-M guideline, has led to a significant reduction in shrimp and other fisheries production. The accumulation of heavy metals and PCBs in seafood also exceeds the limits making them not suitable for human consumption (Liang et al., 1999). To maintain the sustainability of mangrove wetlands in Deep Bay, an important habitat of migratory birds and water fowls, pollution from discharges of domestic, livestock and industrial wastes must be controlled.

Similar to the Pearl River delta, the mangroves in Jiulong River have accumulated many pollutants discharged from Xiamen city (Hong et al., 1995; Maskaoui et al., 2002; Xue et al., 2004).

6. MANGROVE AND WASTEWATER TREATMENT: FEASIBILITY AND SUSTAINABILITY

Mangrove wetlands have long been used as convenient sites of waste disposal and often inadvertently receive untreated wastewater of both human and animal origins (Clough et al., 1983). Mangrove sediments have been acted as sinks of nitrogen and phosphorus (e.g. Corredor and Morell, 1994; Tam and Wong, 1996b; Rivera-Monroy et al., 1999), heavy metals (e.g. Harbison, 1986; Silva et al., 1990; Tam and Wong, 1993, 1995a, 1996a, 1999a) and total organic pollutants (e.g. Tam et al., 2001; Maskaoui et al., 2002). Mangrove plants are specially adapted to stressed environments and are able to tolerate various kinds of pollutants in wastewater. Their high productivity also indicates a great demand on nutrients in sediments. The large diversity of microorganisms associated with the sediments and mangrove roots suggest that chemical and biological transformation of pollutants, in addition to immobilization as insoluble precipitates or bound with organic matter, would take place in mangrove sediments (Harbison, 1986; Lacerda et al., 1993; Tam and Wong, 1995a). All these facts and findings lead to the possibility of employing mangrove wetland, similar to the constructed wetland, as a wastewater treatment facility.

Mangrove wetlands have function as buffer zones to purify upland runoff prior to its entering the sea, and also are used as filters to strip nutrients from shrimp pond effluent (Robertson and Phillips, 1995; Rivera-Monroy et al., 1999). The use of mangroves to treat effluents from shrimp farming in tropical regions has not been widely practiced as the shrimp producers are not required to treat any pond effluents. With the severe impacts of shrimp pond effluents on the water quality of adjacent estuarine and the increase in public awareness on wastewater reuse and recycle, the need to treat pond effluent becomes more and more pressing. Integrated pond-mangrove farming systems with the wetland to shrimp pond ratios varied from 2 to 22 have been proposed (Robertson and Phillips, 1995). The ratio of wetland to shrimp pond for the removal of nitrogen could be reduced to a range of 0.04-0.12 if denitrification occurs in the wetland system (Rivera-Monroy et al., 1999).

In addition to shrimp effluent, the feasibility of using mangroves to remove pollutants from municipal and livestock wastewater has been examined since 1990. Sediments in a fringe mangrove forest were capable of removing nitrate added in the effluent from a sewage treatment plant via nitrification and denitrification processes (Corredor and Morell, 1994). Results from a 3-year long field study at Futian mangroves showed that primarily settled domestic sewage was purified if intermittently discharged to the landward region of the mangrove wetland during the low tide period, tidal water was not contaminated and negative impacts on plant growth were not detected (Wong et al., 1995, 1997b). The species diversity and abundance of macro-algae and benthic invertebrates colonizing the mangrove floor were also not affected by sewage discharge (Liu et al., 1995; Yu et al., 1997). Greenhouse experiments demonstrated that the discharge of simulated municipal sewage at high nutrient concentrations and of livestock wastewater promoted growth of the dominant mangrove plants in China, namely *Kandelia candel*, *Aegiceras corniculatum*, *Avicennia marina* and *Bruguiera gymnorrhiza*, while nutrients in wastewater were removed (Chen et al., 1995; Wong et al., 1997a; Ye and Tam, 2002).

Mangrove wetlands, in general, have a large nutrient assimilative and dissimilative capacity. A field study in Futian mangroves showed that at the end of 3-years discharge of municipal sewage, only the surface sediments in the first two meters from the discharged points had 20% increases in total Kjeldahl nitrogen and 38% increases in phosphorus concentrations, while no significant change in nutrient concentrations was found in other sediments (Wong et al., 1997b). Around 27% of the nitrogen and 85% of the phosphorus from wastewater was retained in mangrove sediments. The amounts of nitrogen retained in mangrove sediments were much less than that of phosphorus, despite the fact that more nitrogen was present in wastewater; this is because the demand for nitrogen by plants is higher than that for phosphorus. Moreover, nitrogen is lost via nitrification and denitrifcation processes. The role of plant uptake in nitrogen removal is still debatable. Chiu et al. (1996) showed that the ^{15}N labelled ammonium added to the experimental pots disappeared rapidly, and around 20% N was taken up by *Kandelia candel* after three months. Greenhouse experiments demonstrated that the percentage of nitrogen lost from the mangrove ecosystem was around 40% and the plant uptake varied from 12 to 68%, dependent on the plant species and salinity (Table 5). On the other hand, most researchers agree that the most important pathways to remove N in a wetland system are ammonia volatilization, nitrification and denitrification. Ammonia volatilization is negligible in mangrove wetlands because of its acidic to neutral pH. On the contrary, nitrification and denitrification processes are very significant because mangroves are periodically flooded by tidal water with alternating aerobic and anaerobic conditions. Mangrove plants are also known to be able to translocate oxygen from the atmosphere to the roots and create an oxygenated zone for nitrification around the roots while the surrounding sediments are reduced thus favor denitrification (Chiu and Chou, 1993; Tam et al., 2002). The population sizes of various bacterial groups, in particular those related to the nitrogen cycle, and the microbial activities in mangrove sediments were found to increase as a result of the addition of nutrient-rich sewage (Table 6). With continuous losses of nitrogen from the system, mangrove ecosystems could be maintained as a sustainable wastewater treatment facility without saturation.

Phosphorus and other pollutants such as heavy metals in wastewater were mainly immobilized in sediments with little loss (Table 7). Although the mangrove ecosystem had a high potential to act as a sink for these pollutants, the continuous accumulation of these pollutants decreases its retention capacity and poses long-term effects on the mangrove ecosystem. The dynamic nature of the mangrove ecosystem, which is strongly influenced by factors like tidal flow, wave action, climates, salinity, redox potential, and various biotic components, may cause the retained pollutants to be re-suspended back to the aquatic environment under extreme environments. Preliminary laboratory studies show that the heavy metals and phosphorus retained in the mangroves were not released when the sewage

Table 5. Distribution of nitrogen (N) and phosphorus (P) from discharge of livestock wastewater in mangrove microcosms planted with different plant species at different salinities (in ppt; Ye et al., 2001).

Components	Nitrogen				Phosphorus			
	K. candel		***B. gymnorrhiza***		***K. candel***		***B. gymnorrhiza***	
	0	30	0	30	0	30	0	30
% in treated effluent	15.7	7.3	4.5	2.0	20.8	12.0	8.2	2.2
% retain in sediment	18.6	39.0	17.3	23.9	70.5	84.1	88.2	95.9
% uptake in plants	18.7	12.0	63.4	30.3	2.6	3.8	3.6	1.9
% losses	47.0	41.7	14.8	43.8	6.1	0.1	0	0

Table 6. Percentages of increases in population sizes of various bacterial groups in surface mangrove sediments receiving wastewater compared to the control (without wastewater addition) (NW: synthetic wastewater with chemical composition simulated municipal sewage in Hong Kong; 4NW: synthetic wastewater with chemical composition four times of that in NW; ND: not determined) (Tam, 1998; Tam et al., 2002).

Parameters	NW sewage discharged to microcosm with *K. candel*	4NW sewage discharged to microcosm without plants	
	0 ppt	0 ppt	15 ppt
Aerobic heterotrophs	20.1	10.8	35.8
Anaerobic heterotrophs	ND	2.5	3.2
Ammonium oxidizers	20.5	7.2	25.8
Nitrite oxidizers	22.6	7.5	10.3
Denitrifiers	24.4	352	7009

Table 7. Distribution of heavy metals from discharge of electroplating wastewater in mangrove microcosms planted with Kandelia candel.

Components	Zn	Cd	Pb	Cr
% in treated effluent	7.0	3.4	2.5	5.3
% retain in sediment	89.5	94.8	95.5	90.4
% uptake in plants	0.2	0.2	0.3	0.1
% losses	3.3	1.6	1.7	4.2

discharge was suspended (Chu et al., 2000; Tam et al., 2002). However, the situation in natural environments and under extreme weather conditions is difficult to evaluate. The possibility of the mangrove sediments becoming a secondary source of pollution has not been addressed and deserves in-depth studies. More research must be conducted to understand the treatment mechanisms, the maximum capacity and saturation, and the long-term adverse effects before employing mangrove wetlands as a wastewater treatment facility.

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CHAPTER 12

FIELD AND MODEL STUDIES OF WATER QUALITY IN HONG KONG

KWOK-LEUNG PUN

1. INTRODUCTION

Hong Kong (Figure 1) is a densely populated coastal city with a population of 6.9 million in 2004. The population is predicted to increase to 9.2 million in 2030. The total land area of Hong Kong is approximately 1,103 km^2 comprising the Hong Kong Island and adjacent islands, Kowloon and New Territories. Many large-scale infrastructure developments in Hong Kong involve reclamation to provide new land for residential and commercial developments. Since 1887 there has been an increase in land area of 67.2 km^2 from reclamation. Most of the reclamations were carried out in Victoria Harbour and are gradually shifting towards the western part of Hong Kong and at Lantau Island.

The growth in population not only poses a need to create new land area but also increases the sewage flow and load discharging into the Hong Kong waters. Comprehensive sewerage studies have been conducted to plan the collection and treatment of sewage generated from the 16 catchment areas in Hong Kong. There are more than 400 major discharge points defined within the whole of Hong Kong waters in water quality modelling studies. Upgrading of the existing sewage treatment works and exporting of treated effluent to less sensitive water bodies have been implemented. The first stage of the Harbour Area Treatment Scheme (HATS), which was formerly named as Strategic Sewage Disposal Scheme (SSDS), was fully implemented in late 2001 using deep tunnels to collect about 1.4 million m^3 of sewage (~75% of the sewage) generated in the harbour area for chemically enhanced primary treatment. The treated effluent is discharged into the near shore water through a submerged outfall. The primary aim of the HATS is to improve the water quality in Victoria Harbour. The HATS Stage 2 is underway to further improve the harbour water quality by collecting all sewage from the harbour area and upgrading the treatment process to biological treatment with disinfection.

After 1997, the economic link between Hong Kong and the Pearl River delta is increasingly important. To shorten the traveling distance between Hong Kong and the major cities in the Pearl River delta. The first bridge linking Hong Kong and Shenzhen is being constructed and is scheduled to complete in 2005. The second

E. Wolanski (ed.), The Environment in Asia Pacific Harbours, 165–186.

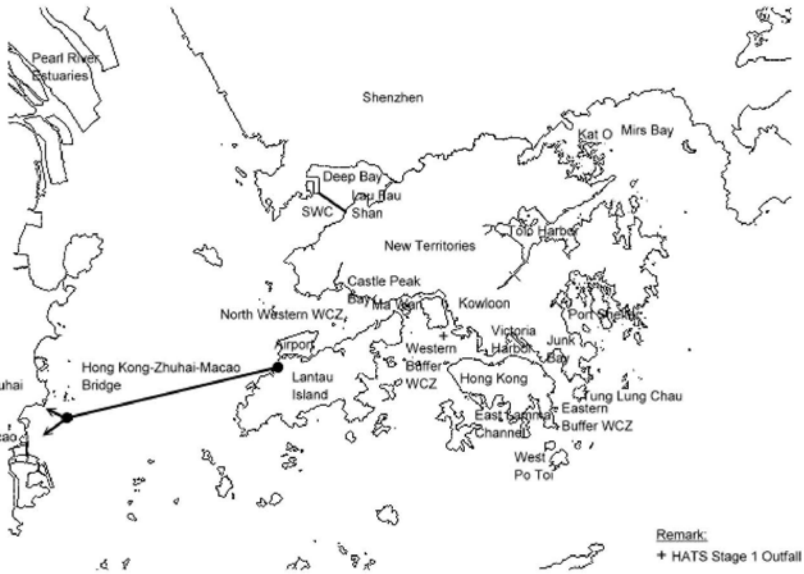


Figure 1. *A location map of Hong Kong and sites mentionned in the text.*

bridge connecting Hong Kong, Zhuhai and Macao is in the planning stage. These two bridges provide a strategic link between both sides to expedite financial cooperation, logistics and transport, tourism, trade, communications and a wide range of services. From an environmental viewpoint, the bridges built across the Hong Kong and Mainland waters may to a certain extent affect the existing coastal environments on both sides. Cross-border environmental issues and the differences in environmental regulations and standards add a challenging component to the projects.

Provision of large-scale infrastructures is required to support all these developments. During the construction and operational stages of the reclamation, bridge construction and sewage disposal projects, water quality impacts may arise from various types of construction activities associated with the projects. These include marine sediment dredging, use of fill materials, and construction of sewage outfalls, seawalls, breakwaters, bridge piers and artificial islands. The construction activities may have a short-term impact on the local water quality whilst the operation of the projects may create a long-term impact on the coastal environments. The continuous discharge of treated effluent will have a long-term impact.

The assessment of short-term and long-term effects on water quality as a result of the infrastructure developments relies on the application of mathematical models. Water quality modelling provides an effective way for checking compliance with relevant standards and determining the feasibility of the engineering works. This

Chapter presents the experience on water quality modelling of the reclamation, bridge construction and export of effluent projects in Hong Kong. The influences in relation to fish culture zones and changes in water quality conditions are addressed.

2. CHARACTERISTICS OF THE HONG KONG COASTAL ENVIRONMENTS

Both the oceanic and estuarine water masses affect the Hong Kong waters throughout the year. The months of January and February (winter) are generally the dry season whereas the months of July and August (summer) are the wet season.
The Pearl River, which is one of the largest rivers in China and is located at the western side of Hong Kong (Figure 1). The average discharge is 19,405 $m^3 s^{-1}$ in the wet season and 4,116 $m^3 s^{-1}$ in the dry season (EPD, 2003). Pearl River freshwater is heavily polluted by sewage and industrial discharges.

The oceanic currents interact with the Pearl River plume. A large amount of brackish water can be found in the western part of Hong Kong waters in the wet season, and the temperature stratification also prevails. The eastern part of Hong Kong waters is less influenced by the Pearl River discharge. The influence of the Pearl River on Hong Kong waters is smaller in the dry season when Hong Kong waters are fairly well-mixed vertically. Both the Kuroshio oceanic current and the coastal Taiwan Current influence the inshore Hong Kong waters in winter. The Kuroshio Current brings the warm water from Pacific Ocean and interacts with Taiwan Current to trigger algal blooms in the eastern part of the Hong Kong waters and spread these blooms southwestward (Wong 2003).

The Agriculture, Fisheries and Conservation Department (AFCD) of the HKSAR (Hong Kong Special Administrative Region) is a coordinating center for algal bloom or red tide outbreaks; it undertakes a programme using a computerised system together with the Geographic Information System for phytoplankton monitoring to disseminate information on red tides in real time. There are about 20 to 30 red tide incidents in a year. Most of the algal blooms occurred in Hong Kong are harmless but a few harmful algal blooms (HAB) still can be found in the Hong Kong waters.

The red tides are linked to the eutrophication of coastal waters. The increases in red tides in Tolo Harbour, a semi-enclosed and poorly flushed water body, between 1976 and 1986 coincided with the rapid growth in population around that area. Between 1980 and 1992, about 50% of the red tide incidents (189 incidents over 13 years) occurred in Tolo Harbour. A serious red tide outbreak occurred at Kat O and widely spread in the northern waters in 1988 causing major fish kills. The bloom was later found in Long Harbour, Tolo Harbour, Port Shelter and the southern waters with a total of 88 incidents reported. The fish culture zones in Tolo Harbour were seriously affected, with very large economic losses.

There are 26 designated fish culture zones occupying a total of 209 hectares of sea area. The fish culture zones at Ma Wan and Tung Lung Chau are located near a number of reclamation sites in the harbour area. These fish culture zones may also be affected by cooling water discharges in the harbour. Within Victoria Harbour, there is no fish culture zone.

AFCD has promoted the use of pellet feeds to replace the traditional trash fish for fish feeding. In 2003, about 47% of mariculture farms had adopted the use of pellet feed for fish feeding. Regular water quality monitoring at fish culture zones has been undertaken to ensure that the water quality is in a good condition and is suitable for fish culture. In collaboration with the City University of Hong Kong, a study of using biofilters to lower the organic waste concentrations in the sediments within the fish culture zones is ongoing.

In the past, oyster cultivation was one of the major industries in Hong Kong. Lau Fau Shan supported most of the oyster cultivation activities in Deep Bay. Oyster farming has been practiced in Deep Bay for about 200 years. Due to the shallow water depth and the Pearl River discharge, siltation created a large area of inter-tidal mudflat in Deep Bay making this bay suitable for oyster cultivation. Raft culture is now the most common method for growing oysters.

Nowadays, water pollution in the inner bay seriously affects the oyster industry in Deep Bay. The sediments contain high concentrations of heavy metals and the water also contains high concentrations of suspended solids, *E. coli* and organic matters. The oyster production has declined from about 1,200 tonnes y^{-1} in the late 1950's to about 76 tonnes y^{-1} in 2000 and 210 tonnes y^{-1} in 2004.

3. WATER QUALITY STANDARDS

The Water Pollution Control Ordinance (WPCO) (Cap.358) was enacted in 1980 to provide the statutory framework for the protection and control of water quality in Hong Kong. Under the Ordinance, water control zones (WCZs) and water quality objectives (WQOs) were declared and established.

3.1. Water Control Zones

In accordance with the WPCO and its subsidiary legislation, the whole of Hong Kong waters is divided into ten water control zones and four supplementary water control zones. In 1987, the Tolo Harbour and Channel Water Control Zone was firstly appointed. The other water control zones were subsequently declared.

3.2. Water Quality Objectives

Maximum levels of pollutants and minimum levels of essential constituents are defined under the WPCO in the water quality objectives for each of the water control zones. These maximum and minimum levels are slightly different in applying to different water control zones depending on the nature of the water body to be protected. The parameters specified in the water quality objectives include odour, colour, floating matters, *E. coli.*, dissolved oxygen, pH, salinity, temperature, suspended solids, turbidity, unionised ammoniacal nitrogen, inorganic nitrogen, 5-day biochemical oxygen demand, chemical oxygen demand and toxic substances.

The water quality objectives for general marine waters specify the depth-averaged dissolved oxygen in the water column not less than 4 mg l^{-1} for 90% of the year and within 2 m from the seabed not less than 2 mg l^{-1} for 90% of the year. In fish culture zones, dissolved oxygen level shall be higher than 5 mg l^{-1} for 90% of occasions. The pH value of the water shall be in the range between 6.5 and 8.5. Any change from natural pH range due to human activity shall not exceed 0.2. The temperature and salinity changes from natural daily temperature range and ambient salinity level shall not exceed 2 ^{o}C and 10% respectively. The increase in suspended solids level due to human activity shall not exceed 30% of the natural ambient suspended solids level. The annual arithmetic mean of unionised ammonia shall not exceed 0.021 mg l^{-1} and the annual arithmetic mean of the depth-averaged inorganic nitrogen level in the water column shall not exceed 0.7 mg l^{-1} for inner Deep Bay, 0.5 mg l^{-1} for North-western Waters and outer Deep Bay waters, 0.4 mg l^{-1} for Western and Easter Buffer Zones and Victoria Harbour, and 0.3 mg l^{-1} for Castle Peak Bay Sub-zone within the North-western Waters. There is no specific requirement on *E. coli* for general marine waters. For bathing waters, the geometric mean of *E. coli* level shall be less than 180 per 100 mL between March and October. In secondary contact recreation zones and fish culture zones, the annual geometric mean of *E. coli* level shall be less than 610 per 100mL.

The water quality objective for temperature change due to human activity in the Victoria Harbour Water Control Zone specified that the variation in temperature should not exceed 2 ºC. This is a limiting value used for assessing the increase in ambient water temperature due to the cooling water discharges. There are no statutory requirements for residual chlorine and biocides in seawater in Hong Kong. The residual chlorine concentration of 0.02 mg l^{-1} is harmful to the aquatic life. A lower value of 0.01 mg l^{-1} is used as the USEPA standard for residual chlorine and this value is generally adopted as the assessment criterion in Hong Kong. The limiting value of 0.1 mg l^{-1} (Ma et al., 1998) is used as the criterion for assessing the concentration of C-Treat-6 in the seawater.

All discharges are controlled by a licensing system within each water control zone. A Technical Memorandum sets limits for discharge of effluent into the water control zones. Specific limits apply for different areas and are different between surface water and sewers. The limits vary with the rate of effluent flow.

For the cross-border projects, both the standards of the HKSAR and mainland China shall be adopted. Under the National Standard of the People's Republic of China UCD 551463, the Sea Water Quality Standard GB3097-1997 has been implemented since 1st July 1998. It specifies water quality objectives for different beneficial uses of marine water in mainland waters.

4. WATER QUALITY MODELS

A number of water quality modelling exercises have been conducted in Hong Kong to assess the water quality impacts associated with infrastructure developments and sewage disposal studies. Assessment of impacts on fish culture zones and redistribution of nutrient load leading to reduction in eutrophication problems forms part of the objectives in some major studies.

In 1982, the first set of comprehensive mathematical models for simulation of hydrodynamics, water quality, waves, and sediment movement in the Hong Kong waters was developed under the Study of Water Quality and Hydraulic Modelling in Victoria Harbour (WAHMO) and were later upgraded in 1987. In 1990, the Danish Hydraulic Institute's MIKE 21 and later the MIKE 3 models for two- and three-dimensional flows were introduced to Hong Kong and were applied in bridge pier protection, drainage master plan, and the Strategic Sewage Disposal Scheme studies in Hong Kong. Since 1998, the Delft3D models have been widely used in reclamation, sewage disposal, bridge construction, cooling water discharge, and the Harbour Area Treatment Scheme projects. The models are used to model 2-D and 3-D flows, water quality, ecology, wave propagation, and morphology. The hydrodynamic model is based on the finite difference method. The water quality model computes the transport of substances numerically by the advection-diffusion equation. The modelling described in this Chapter is mainly based on the framework of Delft3D models.

4.1. Model Set-up and Validation

The selection of model for the prediction of water quality impacts depends on the nature of the project. A refined grid model is generally required to provide a higher level of detail for water quality impact assessment. The model is calibrated against field data obtained from a water sampling program. Monthly water quality data measured by the Environmental Protection Department (EPD) are also used for checking the accuracy of the model. In several environmental impact assessment studies, the refined grid model was verified by comparison with the model prediction from a well-calibrated larger model, the Update model that covers the whole of Pearl River estuaries and the Hong Kong waters. The Update model is commonly used to provide boundary conditions to the refined grid model. Through the link between the larger model and the refined grid model, the influences on hydrodynamic and water quality conditions from the areas outside of the Hong Kong waters are transferred into the refined grid model at its open boundaries.

A general approach for hydrodynamic and water quality modelling is shown in Figure 2.

4.1.1. Coastline Configuration

The very first step in setting up a water quality model is to define a specific time period. Water quality data from field measurement should be available within this period to verify the model. The model coastline configuration is then fitted to the actual land boundary for that period. The only modification to the model set-up is to update the coastline to match with the time horizon for a particular scenario. This approach gives a direct comparison of the changes in water quality conditions between the baseline scenario and the future scenarios avoiding the differences in tidal stages for different simulation periods.

4.1.2. Grid Schematisation

Curvilinear grids are applied to fit the natural coastline within the study area improving the smoothness and orthogonality of the grid cells, hence minimising errors associated with the finite difference method. The total area covered by the refined grid model is much smaller than that of the larger model. A high grid resolution of less than 75 m × 75 m is applied in the vicinity of the project site and

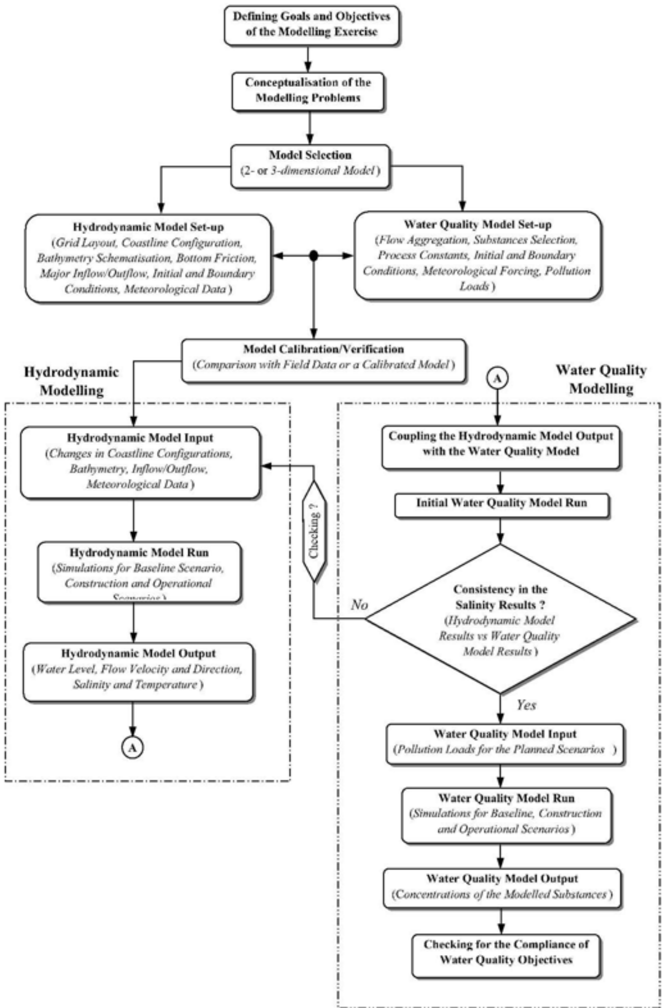


Figure 2. *A general approach for hydrodynamic and water quality modelling.*

the locations where the water sensitive receivers are situated. The grid sizes increase gradually towards the open boundaries.

For a 3-D hydrodynamic model set-up, the vertical column of water body is evenly divided into 10 layers. Depending on the coverage of the modelling area and the stratification condition of the water body, the thickness of each layer can be adjusted to give a better representation of the level of stratified layer.

4.1.3. Bathymetry

The Hydrographic Office, Marine Department of HKSAR, measures and collates bathymetry data in the Hong Kong waters and produces nautical charts. These data provide the basic information for defining the water depths in the model. The depth data are defined at nodal points of the grids. The reference level of the model is Hong Kong Principal Datum.

4.1.4. Simulation Period

Representative spring-neap tidal cycles for dry and wet seasons are used for hydrodynamic computations. The model runs for each season cover a 7-day period for model spin up and 15 days to represent a spring-neap cycle. For the annual water quality simulation, transitional seasons between the wet and dry seasons are defined by interpolating the dry and wet season conditions. The hydrodynamic data of the representative spring-neap cycle are repeatedly used in the water quality model to cover a full annual simulation. Model inputs such as the meteorological forcing and pollution loads from river discharges, storm drains and sewage outfalls vary from month to month over the year.

4.1.5. Flow Aggregation and Coupling

The vertical hydrodynamic structure of the model is aggregated from 10 layers to 5 layers generating a vertical distribution of 10%, 20%, 20%, 30% and 20% of the hydrodynamic layers from surface to bottom in the water quality model. A 2×2 flow aggregation is also applied in the spatial level. The main purpose of flow aggregation is to optimize the computational time and data storage without a significant influence on the quality of the modelling results.

4.1.6. Flow and Pollution Loads

A pollution load inventory is required to provide flow and load data for all existing discharge points including major rivers, storm drains, nullahs, sewage outfalls, landfills, typhoon shelters, and fish cultural zones. New discharge points from the proposed development are added in the model with flow and load data derived from the relevant project information. Different sets of flow and load data are compiled to represent the years for baseline, construction and operational scenarios.

The inventory is compiled based on the field measured flow and load data, planning and development statistics including population, industrial, commercial and agricultural data, and projection for discharges from outfalls, rivers and nullahs. The basic relationship between the pollution loading and flow is expressed as:

$$\text{Loading } (g\,s^{-1}) = \text{Concentration } (mg\,l^{-1}) \times \text{Flow Rate } (l\,s^{-1}) \times \frac{1g}{1000mg} \tag{1}$$

The flow and load data for the existing discharge points include the sources from sewage, sewage flow interception, stormwater runoff and livestock. The sewage flows and loads are mainly derived based on the territory population and employment data for planning study. The data are distributed into the sewerage catchments and are multiplied by the global unit flow and load factors specified in the Sewerage Manual (DSD, 1995) to estimate sewage flows and loads for the catchments. The derived pollutants include suspended solids, biochemical oxygen demand, chemical oxygen demand, total Kjeldahl nitrogen, ammonia nitrogen and *E. coli*. Assumptions are made based on the relevant monitoring data, design standards and relevant studies for the pollutant factors not included in the Sewerage Manual such as total phosphorus, ortho-phosphate, silica, copper, organic nitrogen and total oxidized nitrogen. Pollution load reduction factors are applied to the sewage treatment works. The pollutant removal in sewage treatment works is calculated by:

$$\text{Pollutiant Removal (\%)} = \left[1 - \frac{\text{Pollution Load in Treated Effluent}}{\text{Pollution Load in Raw Sewage}}\right] \times 100\% \tag{2}$$

The flows and loads from stormwater runoff are calculated based on the catchment areas, monthly rainfall data and the load factors from the EPD river monitoring data and the relevant urban stormwater pollution studies. The pollution load measured in watercourses is higher in the wet season when compared with the measured data in the dry season. The compiled flows and loads from stormwater runoff therefore consist of two sets of data to cover the dry season and wet season cases.

A number of major storm outfalls receive livestock discharges. Estimation of livestock loading makes use of data on population of pigs and chicken, and actual biochemical oxygen demand (BOD) load discharged into streams.

The total load from each catchment area is distributed into a number of point sources represented by the sewage outfalls and storm drains. These point sources are distributed along the coastline of each catchment area.

Storm outfall discharges are made in shallow water near the water surface. Therefore, the flow and load data for storm outfall discharges are specified only in the surface layer of the model. Sewage outfall discharges are mostly at the sea bottom due to the use of submerged outfall pipes. Buoyancy effects lift up the pollutants. The sewage outfall discharges are therefore allocated in the middle layer of the vertical water column to take into account this effect.

4.1.7. Boundary and Initial Conditions

The open boundary conditions of the refined grid model can be defined either using field data or the modelling results from the larger model. It is a time consuming and expensive process to obtain field data to define the open boundary conditions. Some specially developed refined grid models covering different modelling areas may have different open boundaries. In view of this, it is a common practice to use the larger model to generate open boundary conditions for the refined grid model. The boundaries are forced by water level and velocity data generated by the larger model. Salinity and temperature data are also transferred to the refined grid model at the boundaries.

An initial model run is performed to generate a restart file at the last time step of the simulation period. The restart file gives the initial conditions to rerun the model at the first time step. By repeating the runs, the model can start the simulation at a realistic condition eliminating the influence of the arbitrary defined initial conditions at the first run.

4.1.8. Meteorological Forcing

The water quality processes link closely to the ambient environmental conditions. Meteorological data including wind, solar surface radiation and water temperature are obtained from the weather monitoring stations of the Hong Kong Observatory and the EPD's marine water monitoring data. Monthly averaged values are used in the model. It is assumed that the data are constant over the whole modelling area. An average wind speed of 5 ms^{-1} is usually applied in both the dry and wet seasons. The wind directions are from the north-east during the dry season and from the south-west during the wet season.

4.1.9. Modelled Substances

The key water quality parameters covered in the water quality model include salinity, water temperature, dissolved oxygen, suspended solids, biochemical oxygen demand, *E. coli*, phytoplankton, organic and inorganic nitrogen, phosphorus, silicate, air-water exchange and benthic processes.

Phosphorus and nitrogen are good indicators of potential algal bloom problems. Sunlight, salinity, water temperature, current condition and trace elements are important factors contributing to the algal growth. For dredging and filling activities, the increase in suspended solids in the water body affecting the fish culture zones and other water sensitive receivers is a key factor to be examined by the model.

4.1.10. Verification of Model

The Study Brief issued by the government authorities for a development project specifies the criteria for hydrodynamic model calibration/verification. The validation criteria specifying the differences between simulated results and the actual field measured data are root-mean-square values of tidal elevation < 8 %, maximum

phase error at high water and low water < 20 minutes, maximum phase error at the peak speed < 20 minutes, maximum current speed deviation < 30%, maximum direction error at peak speed < 15 degree, and maximum salinity deviation < 2.5.
Verification of the water quality model is made by comparing the prediction with the field data. Figure 3 gives an example of the comparison between the predicted annual mean depth-averaged unionised ammonia nitrogen (UIA) results and the EPD field data in 1998. The model predicts high UIA levels (> 0.021 mg l^{-1}) in Deep Bay and Victoria Harbour, and decreasing concentrations seaward. The predicted results are in good agreement with observations. Another example of the prediction of *E. coli* in Victoria Harbour is shown in Animation 1.

After a successfully model verification, the refined grid model is then applied to predict water quality impacts associated with the development project.

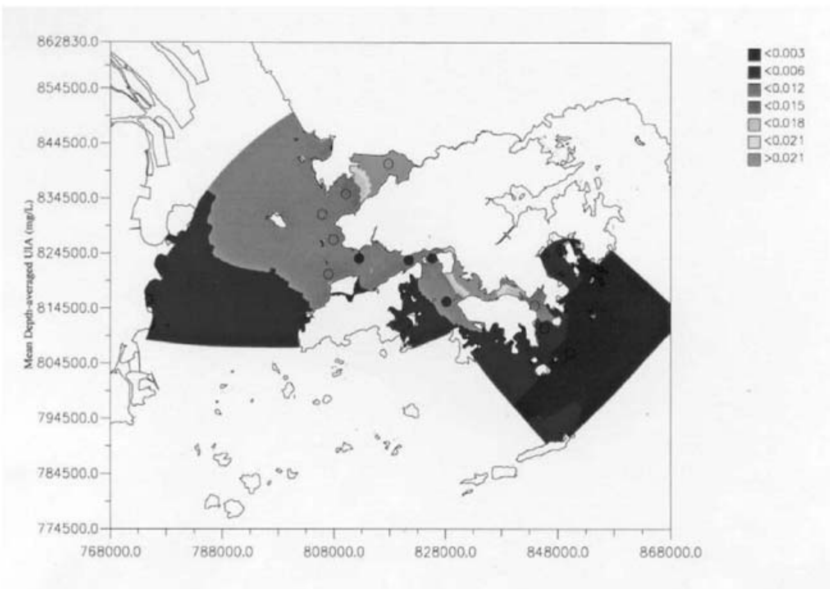
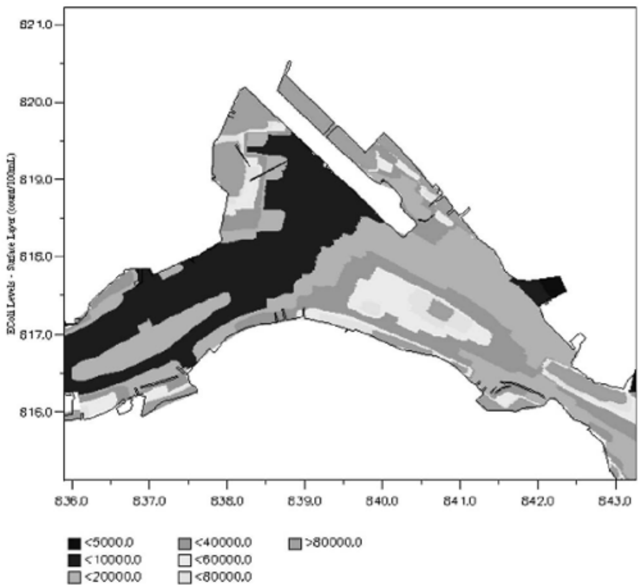


Figure 3. *Comparison of the predicted UIA with field data.*

5. APPLICATIONS OF WATER QUALITY MODELS

5.1. General

A large-scale development project may have a construction period lasting several years. Reclamation of new land to support the development can be divided into several phases involving a range of dredging and filling activities. Modelling scenarios generally cover the baseline, construction, and operational phases. A direct comparison of the model results between the construction phase scenario and the



Animation 1. *Predicted E. coli Concentrations in Victoria Harbour.*

baseline scenario can check for compliance with the water quality objectives and relevant requirements, identifying the key issues that need to be avoided or mitigated. A similar comparison can be made between the operational scenario and the baseline scenario to examine the long-term water quality impacts associated with the development.

5.2. Reclamation Projects

The high population density and limited developable land area in Hong Kong causes a strong demand on new land from reclamation. The natural coastline and seabed conditions have been significantly changed. Since most of the reclamation sites are located within Victoria Harbour, the cross-sectional area of the Victoria Harbour channel is continuously reduced creating short-term and long-term impacts on the flushing capacity and water quality of the harbour water. During the construction stage, massive dredging and filling activities are carried out in the reclamation sites. Generation of sediment plumes and disturbance to the seabed pose a potential risk to the nearby water sensitive receivers.

Examples of some of the major reclamation projects and studies with reclamation sites in Victoria Harbour include Central Reclamation Phase I/II/III (58 hectares), Wanchai Reclamation Phase I/II (35 hectares), West Kowloon

Reclamation (340 hectares) and South East Kowloon Development (previously planned new land of 133 hectares).

For Lantau, major reclamation projects include reclaiming 112 ha of land after completion of the new airport at Chek Lap Kok, and the Tung Chung New Town. Penny's Bay Reclamation and Yam O Reclamation involve reclamation of about 290 hectares of land.

Public fill, and marine sand fill are the major fill materials for reclamation in Hong Kong. Crushed rock from land sources is also an alternative fill but is less commonly used when compared to public fill and marine fill. Since 1990 more than 270 million m^3 of sand have been extracted from the seabed within the Hong Kong waters for use in reclamation projects. Marine borrow areas such as West Po Toi and East Lamma Channel are the specified areas to provide sand fill. Public fill is generated from construction and demolition (C&D) materials, which are inert rock and soil. In some public filling areas, the C&D materials are used for reclamation reducing the reliance on sand fill.

The major concerns on water quality from reclamation are the dispersion of sediment and release of heavy metals and organic micro-pollutants from the disturbed seabed during the construction stage and the long-term changes in hydrodynamics and water quality after the completion of the project. The impact on aquatic life depends on the location of the reclamation sites. There are 26 designated fish culture zones in Hong Kong. Local fish farmers have reported fish-kills that they blame the sediment plumes generated from dredging and filling activities on reclamation sites. The other water sensitive receivers located within the harbour and in the adjacent regions including seawater intakes for cooling and flushing purposes, enclosed water bodies and gazetted beaches are also subject to the influence from dredging and filling activities.

The development projects in Victoria Harbour may also involve relocation or provision of cooling water discharges from water-cooled air conditioning systems. A District Cooling Scheme has been proposed by the Government to explore more economical and environmental attractive air conditioning systems for Hong Kong. The temperature of the extracted seawater for cooling can be heated up to 3 to 7 ^{o}C. In addition to the temperature rise, the discharges may also contain anti-fouling and anti-corrosion chemicals like residual chlorine and biocides causing harmful effects to the aquatic life.

For a large-scale reclamation, dredging and filling are required to carry out in phases. Therefore, a number of construction scenarios can be determined based on the planned construction programme to identify the worst-case scenarios where the sediment load releasing into the water body is expected to be critical. There are different construction methods for the dredging and filling activities on reclamation sites. One of the methods is to carry out the filling behind seawall. Construction of external seawall or barrier is conducted to enclose the inner area where massive filling will be subsequently carried out. In this case, there will be almost no release of sediment during filling. Sediment plume dispersion modelling may not be required. The impact on the nearby sensitive receivers including the fish cultural zones in this case is minimal. An alternative method is to install silt curtain surrounding the dredging and filling sites to confine the sediment plumes within the works areas. The effectiveness of using silt curtain depends on the management and operational control of the dredging and filling works, and the tidal current conditions. The workable tidal current speeds are in the range between 0.3 and 0.5 $m\,s^{-1}$.

Under a fully controlled operating condition, a reduction factor of 80% (silt curtain efficiency) can be applied in calculating the sediment loss rate, which is inputted into the model for simulation of sediment plume dispersion. The third method is not to execute any mitigation measures such as provision of external seawall or silt curtain. In simulation this situation, it is considered to be the worst-case scenario. Figure 4 shows a time series plot for suspended solids at Ma Wan Fish Culture Zone, with and without reclamation activities. In this case the increase in suspended solids appears to be within acceptable levels (< 30% increase of the background level).

The required model inputs for modelling the cooling water discharges in the harbour include:

- Maximum discharge flow rate
- Number of discharge points, locations and submerged/surface discharge
- Maximum temperature excess of the cooling water discharges: 6 – 7 ^{o}C
- Residual chlorine concentration at discharge: 0.3 – 0.5 mg l^{-1} with no decay for a conservative approach or with a decay factor of T_{90} = 0.021 day
- Biocide (C-Treat-6) concentration at discharge: < 2 mg l^{-1} with a decay rate of 64% (decayed amount) in 8 days

The required dilution rates to comply with the WQO for temperature are about 3 – 3.5. This is rather easy to achieve in the open waters with fast flowing tidal currents. To meet the recommended standards for residual chlorine and biocide, the

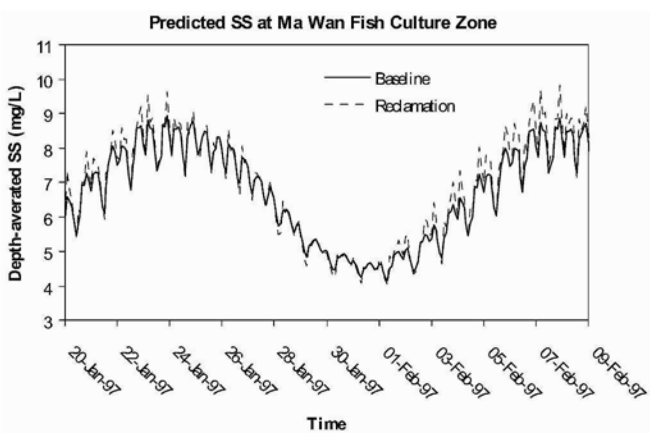


Figure 4. *Time series plot for suspended solids with and without reclamation activities.*

required dilution rates are 50 and 20, respectively. This implies that the area influenced by the high residual chlorine and biocide concentrations is larger than that under the thermal impact. Based on the water quality modelling results from various reclamation studies, no significant impacts in terms of sediment plume and thermal plume dispersion have been identified.

A rigorous environmental process is undertaken to safeguard the water quality that is sensitive to pollution. In the environmental impact assessment stage, water quality modelling is required to assess the impacts from sediment plume dispersion and changes in water quality conditions. All the assessment results shall comply with the relevant water quality objectives before the actual implementation of the project. Mitigation measures to avoid and minimize water quality impacts shall be proposed and implemented in accordance with an implementation schedule.

During the construction and operational stages of the project, water quality monitoring is required to ensure that any unforeseen situations affecting the water quality in the receiving water body and the water sensitive receivers can be detected and rectified. This provides a second level of protective measure to safeguard the coastal environments. An example of the monitoring results for suspended solids of a reclamation project is shown in Figure 5. If the suspended solids (SS) levels exceed the action or limit level, an Event and Action Plan will be implemented. For depth-averaged suspended solids, the action level defines 95%-ile of baseline data or 120% of the SS levels measured at the upstream control station at the same tide of the same day and the limit level defines 99%-ile of baseline or 130% of the SS levels measured at the upstream control station at the same tide of the same day.

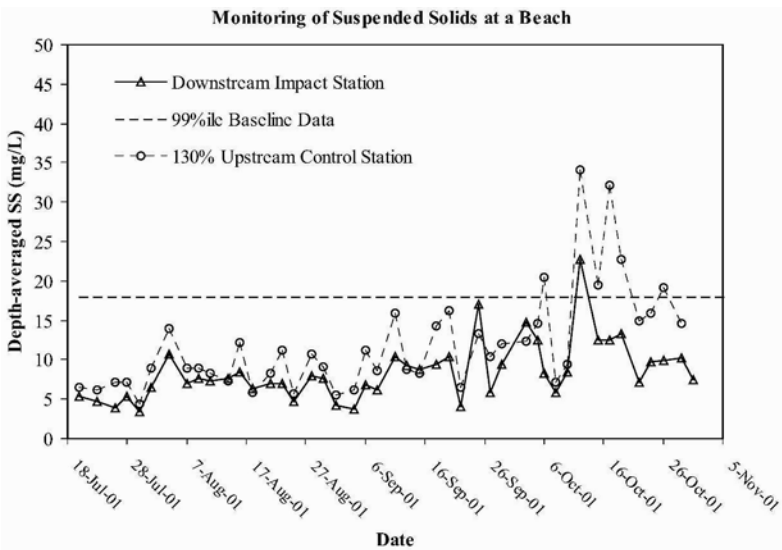


Figure 5. Monitoring of suspended solids concentrations at a beach.

5.3. Bridge Projects

In 1997 to 1999, the construction of three major bridges to support high-capacity transport links in the western part of Hong Kong were completed serving the new airport at Che Lap Kok and the new industrial and residential developments at Lantau. These three bridges are the Tsing Ma bridge (1,377 m), the Ting Kau bridge (448 m) and the Kap Shui Mun bridge (430 m). In addition, two bridges linking Hong Kong and the mainland China will be constructed. These are the Shenzhen Western Corridor (SWC) and the Hong Kong-Zhuhai-Macao Bridge (HZMB).

These two bridge projects share some common features with the needs for bridge pier and pile cap construction, dredging of marine sediment and sand filling, construction of major structures in the navigation channels (cable-stayed bridge/submerged tunnel) and reclamation. All these activities during the construction stage may cause impacts on the exiting aquatic environment. The presence of the bridges may also have long-term water quality impacts. One of the major water quality issues of these projects is the reduction in flushing capacity leading to degradation of water quality. The major water sensitive receivers are the Chinese White Dolphin feeding ground, finless porpoise area, a marine park, horseshoe crab areas, and Special Site of Scientific Interest (SSSI). These projects may also impact Deep Bay, which is a semi-enclosed water body supporting many special areas with high ecological value, e.g. Ramsar Site, Mai Po Nature Reserve Area, oyster farms, seagrass beds, SSSI and horseshoe crab areas, possibly leading to the disappearance of mud flats and the loss of feeding grounds for birds and many animals.

The water depth in Deep Bay is shallow. In the near shore region, the water depth is mostly below 2.5 m. During ebb tides, a large area of mud flats is exposed and provides a feeding ground for many species of birds. Deep Bay has been known as a highly polluted water body with nutrient levels often exceeding the corresponding water quality objectives. Water quality in the outer sub-zone of Deep Bay is in general better than that in the inner sub-zone. Exceedances of the WQOs for dissolved oxygen, total inorganic nitrogen and unionised ammonia are commonly recorded in the inner sub-zone. Ortho-phosphate and total phosphorus levels are also high in inner sub-zone.

The environmental impact assessment of the SWC project was completed in 2002 (Highways Department 2002). The model grid matches with the varying seabed conditions in Deep Bay. The small channels in the shallow water region and the areas near the Inner Deep Bay contain finer grid sizes. The grid sizes vary from 25 m in the small channels to 800 m near the open boundaries. There were 2,700 computational grid cells. The horizontal eddy viscosity and diffusivity were 1 $m^2\ s^{-1}$. The k-ε model was used to compute the vertical eddy viscosity and diffusivity. Minimum values for the vertical eddy diffusivity and vertical eddy viscosity were set at $10^{-7}\ m^2 s^{-1}$ and $5 \times 10^{-5}\ m^2 s^{-1}$ respectively.

The bridge piers create friction to the tidal flows reducing the flushing capacity in the bay. An additional quadratic friction term is added to the momentum equations to simulate the frictional effect along the bridge alignment (Hyder, CES and Delft Hydraulics 1998, and Netherlands Marine Technological Research 1980). Loss coefficient in the x- and y-directions (C_{Lx} and C_{Ly}) are expressed as:

$$C_{Lx} = \frac{nD}{2} \cdot C_d \cdot \left[\frac{A^2}{A_b^2} \right] \frac{1}{\Delta y} \tag{3}$$

$$C_{Lx} = \frac{nD}{2} \cdot C_d \cdot \left[\frac{A^2}{A_b^2} \right] \frac{1}{\Delta x} \tag{4}$$

where,
n is the number of bridge piers in the control grid cell;
C_d is the drag coefficient;
D is the diameter of the bridge pier (m);
A is the total cross-sectional area of the pier (m^2);
A_b is the effective area and is the difference between the total cross-sectional area and the area blocked by the pier (m^2); and
Δx and Δy are the grid distances in the x- and y-directions (m).

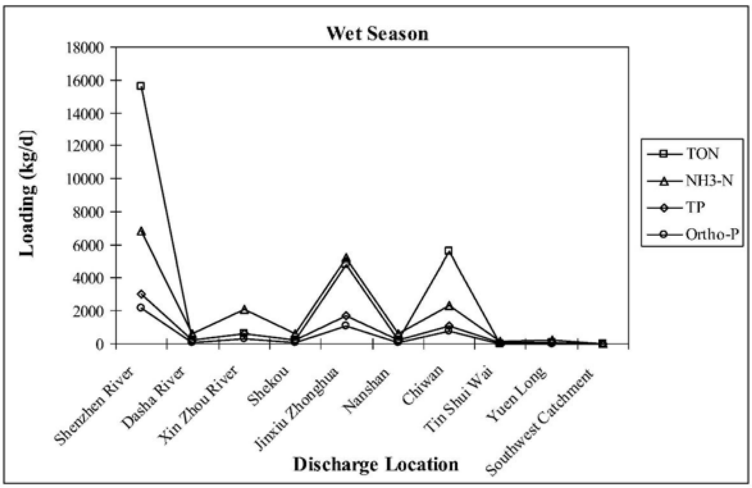


Figure 6. *Nutrient loadings in Deep Bay – wet season.*

There are ten major locations discharging pollutants into Deep Bay. Figure 6 shows the total loadings of total oxidized nitrogen (TON) (27,455 kg d^{-1}), ammonia nitrogen (NH_3-N) (18,779 kg d^{-1}), total phosphorus (TP) (7,216 kg d^{-1}) and ortho-phosphate (Ortho-P) (4,738 kg d^{-1}) at the major discharge locations in the wet season. Shenzhen River, catchment of Jinxiu Zhonghua and Chiwan appear to be the major nutrient sources. The total *E. coli* (1.12×10^{17} no. d^{-1}) and biochemcial oxygen demand (248,860 kg d^{-1}) loads are also high. The pollution loads for the wet season are higher than those pollution loads for the dry season. The differences in nutrient loads for organic nitrogen, ammonia nitrogen, total phosphorus and ortho-phosphate are approximately 13.5%, 1.5%, 5.2% and 1.6% respectively. The highest differences are in biochemical oxygen demand (16.8%) and suspended solids (33.2%).

Using the same model set-up to perform a water quality simulation, the predicted total inorganic nitrogen (TIN) results for dry and wet seasons were compared with the field data (Figure 7) . The predicted TIN levels are higher in the wet season and are within the natural fluctuations of the field data.

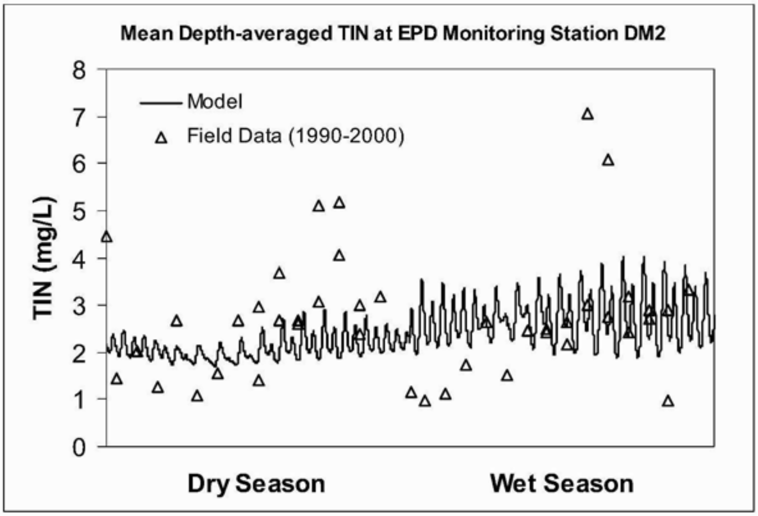


Figure 7. *Water quality model prediction and field data showing natural fluctuation of total inorganic nitrogen in Deep Bay.*

In the water quality impact assessment of the SWC project (Highways Department 2002), the predicted reduction in flushing capacity to be caused by bridge piers alone was 0.76% and by the reclamation at the landing point togather with the bridge piers was 3.3%. The overall change in water quality conditions in Deep Bay due to the project was approximately 2.5%. No significant deviation of water quality conditions from the baseline conditions was found. The predicted cumulative impacts for different scenarios due to sediment dredging along the bridge alignment and filling at the reclamation sites on the mainland side increased the suspended solids levels at the oyster beds near Lau Fau Shan to about 4.1 – 5.95% (1.24 – 1.8 mg l^{-1}) in the dry season and 5.06– 5.12% (1.5 – 1.52 mg l^{-1}) in the wet season. All the predicted increases were below the WQO for SS (< 30 mg l^{-1} increase of the background value).

5.4. Export of Effluent Projects

To cope with the growing population and increasing sewage flows, the Hong Kong Government initiated a sewage disposal strategy in 1989. In the northern district and Tolo areas of Hong Kong, expansion and upgrading of sewage treatment works (STW), and export of treated effluent to the less sensitive water bodies have been studied by the government authorities. The Tolo Harbour Effluent Export Scheme (THEES) has been implemented in phases since 1995-96 to reduce nitrogen loadings on Tolo Harbour. Treated effluent is discharged into Victoria Harbour.

The North District effluent is discharged into the Deep Bay waters. The water quality monitoring data (Figure 8) shows that total nitrogen and total inorganic nitrogen levels in the inner bay of Tolo Harbour were high before 1998 and gradually reduced after the full implementation of the THESS. The nutrient levels in the bay were still relatively high. The frequency of red tide incidents also remained high although the nutrient levels in the harbour dropped.

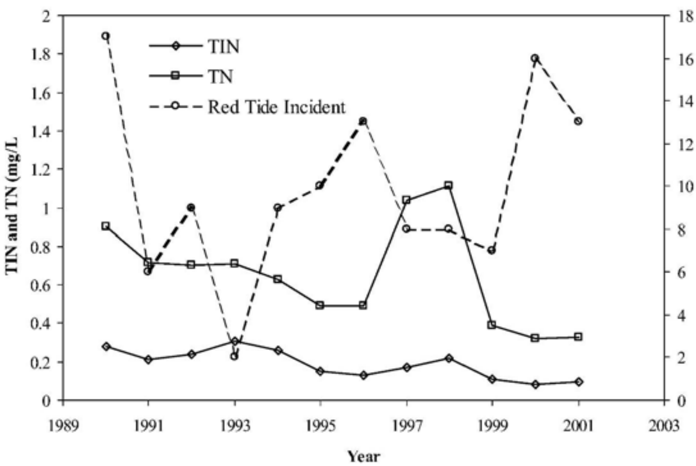


Figure 8. *Measured total inorganic nitrogen and total nitrogen in inner bay of Tolo Harbour Water Control Zone and red tide indicents.*

There is a need for the sewage from the North District to be exported out of Deep Bay. Victoria Harbour, North Western, Western Buffer and Eastern Buffer waters appear to be the more suitable receiving water bodies for the discharge of exported effluent from sewage treatment works. In fact, the Tolo effluent from Sha Tin STW and Tai Po STW has already been collected and discharged via Kai Tak Nullah to the Victoria Harbour.

Water quality modelling for the study of effluent export schemes aims to cover all the possible discharge locations of the exported effluent. Depending on the export location, the potential water quality impact zones and water sensitive receivers that may be affected by the dispersed pollutants arising from different export options include marine parks, fish culture zones, typhoon shelters, cooling and sea water intakes, beaches and secondary contact recreational zones. The long-term water quality impacts over all seasons in a year are the major concern for the case of effluent export. The requirements for modelling of effluent exported from sewage treatment works are outlined as follows:

- Define all possible scenarios including treatment levels of the sewage treatment works and effluent export routes
- Identify potential export locations and water sensitive receivers
- Calibrate a 3-dimensional hydrodynamic model to cover a real sequence of representative 15-day spring-neap tidal cycle in both dry and wet seasons
- Calibrate a 3-dimensional water quality model to run for a complete year using repeating tides of full spring-neap cycles in both dry and wet seasons with the incorporation of monthly variations in flows and loads from Pearl River estuaries and meteorological factors
- Define time horizons for different water quality modelling scenarios
- Compile a pollution load inventory to distribute accurate flows and loads in all the discharge points within the modelling area for all the time horizons under consideration
- Identify water quality objectives and relevant criteria for water quality impact assessment
- Conduct hydrodynamic and water quality modelling of the baseline and all export scenarios
- Perform sensitivity tests of the preferred option for different flows and loads from sewage treatment works
- Model emergency discharge cases as a result of the shut-down of sewage treatment works by allocating the flows and loads at the discharge point during neap tide to represent a worst-case situation
- Carry out cumulative impact modelling by incorporating the flows and loads from the concurrent projects
- Assess the compliance of water quality objectives and relevant criteria of all the cases and identify the scenario with minimal water quality impacts.

In the case where the discharge of the exported effluent is through a submerged outfall (for location of major outfalls, see Figure 1), near-field modelling is required to determine the compliance with mixing zone criteria and to assess the water quality impacts in the near-field region. The JETLAG and CORMIX models have widely been used for near-field modeling in the Hong Kong projects. JETLAG, which was developed by Lee and Cheung (1991), has been designed for modeling outfall plume behaviour from single port discharges to rosette diffusers and is capable of calculating an arbitrarily inclined buoyant jet with a three-dimensional trajectory. CORMIX, which was developed by USEPA, can simulate single diffuser discharges (CORMIX 1), multi-port diffuser discharges (CORMIX 2) and surface discharges (CORMIX 3). The model is capable of predicting the outfall discharge behaviour in the near-field region and the subsequent mixing zone. Extensive laboratory and field data have been used to verify the applicability of the model.

In order to compare the near-field model predictions with the water quality objectives for annual, depth-average values, the pollutant concentration at the edge of the initial dilution zone (C_Z) is calculated by the following equation:

$$C_Z = C_B + \frac{C_0 - C_B}{S} \tag{5}$$

Where,
C_B = background concentration of the pollutant in the ambient water;
C_0 = initial concentration of the discharged effluent;
S = dilution at the edge of the initial dilution zone.

The pollutant concentration C_Z is compared with the corresponding water quality objective to check for compliance. A selected near-field model predicts the dilution under different discharge and ambient conditions. Hourly simulation is performed using the near-field model for 12 months to give the annual results and to generate statistics of the pollutant concentration. For the depth-averaged requirement of the WQOs, the following equation calculates the depth-averaged concentration of the pollutant ($C_{\text{depth-averaged}}$):

$$C_{depth-averaged} = \frac{T}{H}\left[\frac{C_0 - C_B}{S} + C_B\right] \tag{6}$$

where,
T = plume thickness at the edge of the initial dilution zone; and
H = total water depth.

The near-field model results provide details of the vertical structure of effluent plume in the mixing zone. The predicted results of plume height and dilution can be inputted into the water quality model as the loading from the submerged outfall for prediction of water quality changes in the far-field. The allocation of pollution load in the far-field model may affect the accuracy of the model prediction. The link between near-field and far-field models needs to be accurately established in order to ensure that reliable model prediction can be provided for decision-making. At this stage, there is no standard method to couple the near-field model and the far-field model. Bleninger and Jirka (2005) recommended a coupling procedure using CORMIX and Delft3D to improve the quality of the model results in the far-field.

6. CONCLUSIONS

This chapter presented the general characteristics of the Hong Kong coastal environments and the potential impacts associated with large-scale developments. The approach for water quality model validation and application of the validated models for prediction of water quality changes in reclamation, bridge construction and export of effluent projects are briefly described.

Dredging and filling activities are of major concern in the reclamation and bridge construction projects. Examples have been given of the increase in suspended solids in fish culture zones and oyster beds in Deep Bay. Export of effluent from sewage treatment works to less sensitive water bodies on one hand reduces the eutrophication in the originally affected water body but on the other hand introduces additional loads into the receiving water body. Through the application of water quality modelling, the potential impacts and benefits of the project can be examined providing accurate data for decision making on the feasibility of the development projects.

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CHAPTER 13

EUTROPHICATION DYNAMICS IN HONG KONG COASTAL WATERS: PHYSICAL AND BIOLOGICAL INTERACTIONS

JOSEPH H.W. LEE, PAUL J. HARRISON, CUIPING KUANG, AND KEDONG YIN

1. INTRODUCTION

Hong Kong is a mega-city of 6.7 million people that contributes a high nutrient load through its sewage discharge and it is one of the busiest ports in the world. Hong Kong waters are relatively unique because of its intensive utilization of marine resources and frequent occurrence of red tides. Within a small area there is a complexity and richness in eutrophication dynamics, with a sharp gradient from potential phosphorus limitation in western/southern waters to nitrogen limitation in the eastern waters in the summer (Yin at al., 2000 and 2001). Hong Kong waters are sub-tropical, with a clear wet season from May to August, accompanied by southwest monsoon winds and a November to March dry season with northeast monsoon winds, and transitional months in between.

The Pearl River is China's third longest river and the second largest river in terms of discharge volume. It forms the Pearl River Estuary (PRE) as it flows into the northern shelf of the South China Sea, near Hong Kong (Figure 1a). Its average annual flow is approximately 10,500 $m^3 s^{-1}$, and 80% of the total flow occurs in the wet season due to the high rainfall during this period (annual rainfall of 2100 mm).

In the summer, river water moves into the western waters of Hong Kong due to the southwest monsoon winds. During the dry season, the northeast monsoon winds cause the surface river plume to move to the western side of the estuary, away from Hong Kong. An important feature of Hong Kong waters that lie on the eastern side of PRE is that they are shallow (mainly 10 to 20 m) and interlaced with several hundred of islands and inlets (Figure 1b). This topography and bathymetry increases the complexity of the hydrodynamics, which in turn influences the occurrence of episodic events, red tides and other algal blooms.

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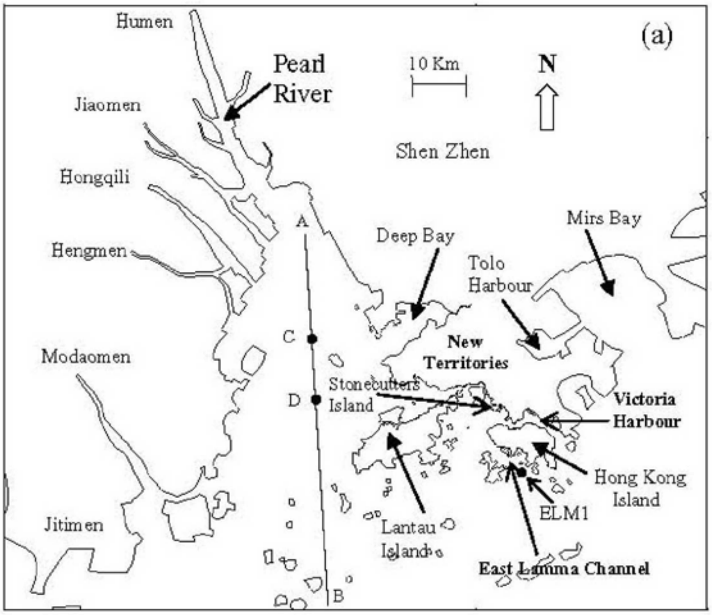


Figure 1a. *Pearl River estuary with various stations various stations noted.*

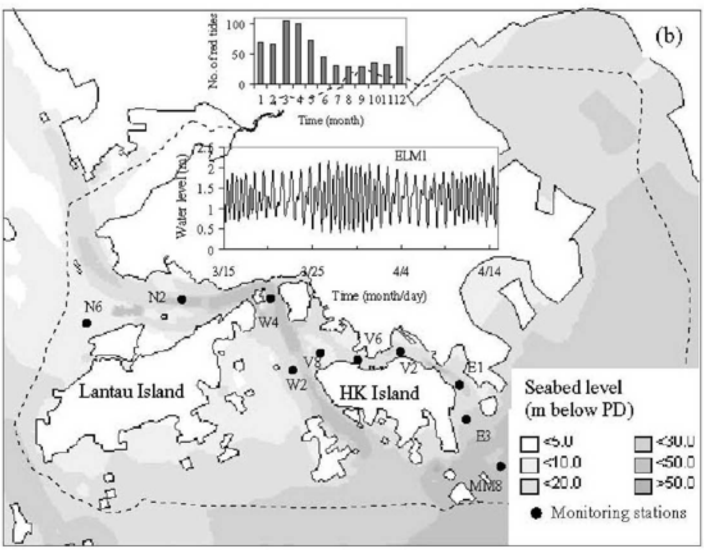


Figure 1b. *Bathymetry of Hong Kong waters (inside the dashed line). Inset figures are the seasonal distribution of red tides in Hong Kong during 1975-2003 (top) and predicted water level during March to April 1998 at station ELM1 in East Lamma Channel (middle inset).*

Rapid urbanization and industrialization has taken place during the past several decades in the Pearl River Delta, and in Hong Kong; the PRD region is now one of the largest and fastest growing manufacturing bases in the world. The total population in the Delta including Hong Kong and Macau is now about 35 million people, and over 100 million people live in the entire watershed of the Pearl River and this has increased the potential for eutrophication impacts. However, these impacts are much less than expected, given the relatively large nutrient and organic loads that are received by these waters. Over the last three decades, nitrate concentrations in the river have increased two to three times (up to >100 μM during high river discharge in summer), while phosphate has remained relatively constant at about 1 μM (Yin 2002). Up to 2001, most of Hong Kong's sewage only received preliminary treatment and the sewage was discharged through a number of submarine outfalls in Victoria Harbour and surrounding waters, resulting in high nutrients and BOD loads. In 2002, as part of the Harbour Area Treatment Scheme (HATS), a central sewage tunnel collection system and chemically enhanced primary treatment (CEPT) plant came into full operation at Stonecutters Island (Figure 1a). Currently 70% of the sewage receives CEPT with about 7 tonnes of sludge removal per day and the treated sewage is discharged through a short outfall. There are two main sources of nutrients for Hong Kong waters: (i) nitrogen loads from the Pearl River and non-point sources in local catchments that fluctuate seasonally with large loads in summer; and (ii) relatively constant inputs of nitrogen from sewage discharge in Victoria Harbour (Li et al., 2003; Yin and Harrison in press). Figure 2 shows the monthly variation of total inorganic nitrogen (TIN) at a number of stations from west to east (N6, W4, V8, E1, MM8; see Figure 1b); the increase in TIN due to the Pearl River flow in the wet season is apparent at Stations N6 and W4. The nitrogen input from sewage discharge into Victoria Harbor is evident at Stations V8, V6 and V2 (Figure 2).

The potential eutrophication impacts of concern in Hong Kong are excessive algal blooms leading to low dissolved oxygen in the bottom waters, beach closures, and red tides. The concern over red tides (colored waters) is mainly associated with fish kills (likely due to low oxygen stress) and there is less concern associated with toxic red tides (Lee et al., 1991b). For example, in April 1998, a devastating red tide due to the dinoflagellate *Karenia digitatum* resulted in the worst fish kill in Hong Kong's history and the estimated loss was more than HK$ 312 million (Dickman, 1998; Yang et al., 2000). An analyses of long term data (1980-2002) obtained by the Hong Kong Environmental Protection Department (EPD, 2003) showed that the maximum number of red tides occur in spring (March-April; see Figure 1b) and these are mostly dinoflagellates such as *Noctiluca* that occur in eastern waters such as Mirs Bay (Yin, 2003). The lowest number of red tides occurred in the western waters, near the Pearl River Estuary (PRE). Therefore, most red tides occur in spring when the Pearl River discharge is relatively low (temporal disconnect from the PRE) and in the eastern waters far from the PRE (spatial disconnect). The dominant red tide species are *Noctiluca scintillans, Mesodinium rubrum, Gonyaulax polygramma, Skeletomema costatum, Prorocentrum minimum, Ceratium furca,* *Prorocentrum triestinum, Thalassiosira* spp. *Scrippsiella trochoidea,* and *Prorocentrum sigmoides* (Yin, 2003).

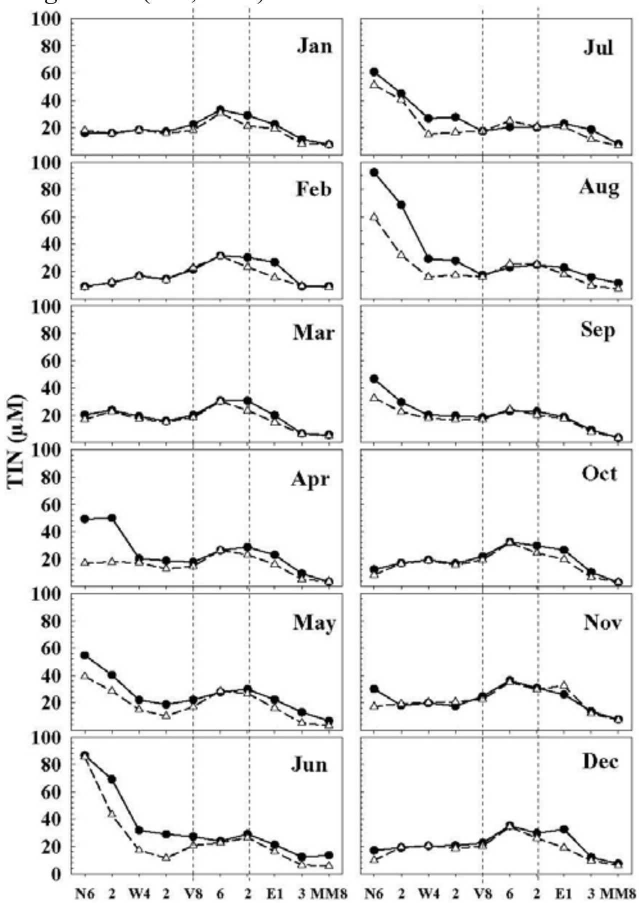


Figure 2. *Annual cycle of total inorganic nitrogen (TIN) at various stations (see Figure 1b for station locations), with N6 and N2 representing western waters, V8, V6 and V2 (between the vertical lines) representing Victoria Harbor and MM8 representing more coastal/oceanic conditions in eastern waters (from Yin and Harrison in press).*

An extensive water quality monitoring program has been in operation since 1982 and this has provided a very valuable data set for characterizing the eutrophication dynamics of the Hong Kong waters. The Environmental Protection Department has divided Hong Kong waters into 10 water quality zones (WQZ) with each zone having its own set of beneficial uses and water quality standards. Monthly sampling for a wide range of water quality parameters is routinely carried out by EPD at 86 monitoring stations. The Agriculture, Fisheries and Conservation Department (AFCD) monitors fish and shellfish culture zones and marine conservation sites for phytoplankton species, with an emphasis on red tides. These red tides are usually detected when the public reports coloured water and then a sample is obtained to determine the phytoplankton species involved. Therefore visual red tides such as *Noctilua* that are bright red (and occur as a bloom right at the surface) may be reported more frequently than high biomass diatom blooms that do not have a marked visual appearance.

While the long-term data sets from EPD and AFCD have provided excellent information on seasonal and inter-annual trends of water quality, there have been very few observations on the temporal and spatial variability of algal blooms in the sub-tropical Hong Kong waters. We have a limited understanding of the importance of physical factors such as monsoon winds, tidal cycles, rainfall and stratification that affect algal bloom dynamics. In particular, we have little information about how short-lived episodic events triggered by hydro-meteorological factors such as storms and typhoons, longer-lived El Nino events and inter-annual and inter-decadal variations in river discharge (e.g. due to climate change) control the environmental assimilative capacity of Hong Kong waters.

This review provides an insight into the complex temporal and spatial dynamics of physical, chemical and biological factors that influence the capacity of the Pearl River Estuary to assimilate anthropogenic inputs such as nutrients with surprisingly minor eutrophication impacts. This review will focus on Hong Kong waters in general, and specifically on Victoria Harbour and southern waters. It will exclude the infamous Tolo Harbour, which is a poorly flushed tidal inlet with a high incidence of red tides (Wear et al., 1984; Ho and Hodgkiss, 1991; Lee et al., 1991a,b; Hodgkiss and Ho, 1997), especially in the 1980s before nutrients were exported away from the Tolo catchment into Victoria Harbour. Based on long term water quality monitoring data and several intensive field and modeling studies, an integrated understanding of eutrophication dynamics in Hong Kong waters is beginning to emerge. A model is presented that shows how various factors interact to make the Hong Kong waters more robust to eutrophication impacts than one would otherwise expect, based on the high nutrient concentrations in these waters. Section 2 reviews the dry and wet season physical hydrography, including tidal characteristics, tidal currents, salinity structure, vertical mixing, dilution, horizontal net transport, and monsoon winds. Sections 3 and 4 document physical and biological interactions on two different time scales, the seasonal (dry and wet) cycle and episodic events such as the hydrodynamic tracking of the massive red tide in 1998. This review concludes by emphasizing the point that algal bloom dynamics in estuaries cannot be understood without a thorough understanding of coastal hydrodynamics and its coupling with phytoplankton ecology. This information has implications for ecosystem management.

2. PHYSICAL HYDROGRAPHY

Physical processes are driving forces for biological processes. The general physical hydrography, flushing rates and algal bloom transport patterns in Hong Kong waters can be elucidated using a calibrated three-dimensional hydrodynamic model

(Delft3D) for Hong Kong waters and the Pearl River Estuary. The Delft3D model solves the shallow water equations based on the hydrostatic approximation and a standard k-ε model for turbulence closure (Delft Hydraulics, 1998); the model has been extensively validated against field data and numerical experiments (Postma et al., 1999; Lee and Qu, 2004). The computational domain (shown in part in Figure 1a) includes eight Pearl River outlets (Humen, Jiaomen, Hongqili, Hengmen, Modaomen, Hutiaomen, Aimen and Jitimen), Hong Kong waters (Deep Bay, Victoria Harbour, Lamma Channel, Mirs Bay), and part of the South China Sea.

Along an open boundary, the water level is specified by a time history of water levels using nine tidal constituents (O1, P1, K1, N2, M2, S2, K2, M4, MS4) derived from long-term data; the mean sea level is calibrated against observed monsoon-driven currents in the open sea. Salinity boundary conditions for the dry and wet seasons are derived from long term measurements. The seasonal mean freshwater flows of 4120 $m^3 s^{-1}$ and 19,422 $m^3 s^{-1}$ are prescribed at the inflow boundary for the dry and wet seasons respectively. A constant NE wind (5 $m s^{-1}$) is assumed for the average dry season, and a SW wind (5 $m s^{-1}$) for the average wet season. The average grid size varies from about 100 m in Victoria Harbour to 7 km in the southeast boundary. A time step of Δt = 2 min is adopted. The model is run from an initial state obtained by running the model for two months from a cold-start to get reasonable initial salinity distributions in both horizontal and vertical directions. Further model implementation details can be found elsewhere (Delft Hydraulics, 1998).

2.1. Tidal Currents

The hydrography of Hong Kong waters is mainly influenced by three factors: tidal currents, the Pearl River discharge, and monsoon-induced coastal currents. The general character of the tide can be expressed by the F-factor (ratio of tidal amplitudes for (K1+O1)/(M2+S2). In general, tides would be considered semi-diurnal if $F < 0.25$ and diurnal if $F > 3.0$ (Bowden, 1983). In Hong Kong, F varies from 0.89 in Deep Bay (Tsim Bei Tsui) on the west to 1.27 in Mirs Bay (Kau Lau Wan) in eastern waters (Lee, unpublished; http://www.hko.gov.hk/tide/etide). Since $0.5 < F < 1.5$, tides in Hong Kong can be characterised as mixed and mainly semi-diurnal. The mean tidal range is 1.7 m; corresponding values for spring and neap tides are typically 2.0 and 1.0 m respectively. A typical spring-neap tidal variation in East Lamma Channel is shown in Figure1b (inset); it can be seen that the tide can vary from semi-diurnal tide (late March) to practically diurnal tide (early April) within a spring-neap cycle.

Tidal currents are determined by the interaction of ocean tides with the local topography and bathymetry; in general, the flow is from E/SE to W/NW through Victoria Harbour and East Lamma Channel which are deeper than the surrounding area, up towards the Pearl River estuary during flood, and from W/NW to E/SE during ebb. Figures 3a and 3b, and animation 1a, show the typical surface flow field during flood and ebb in the dry season. In the main tidal stream, significant peak surface velocities can be found in the northwestern waters north of Lantau Island (up to 2 $m s^{-1}$ in narrow channels). During spring tide, the peak ebb velocity in Victoria

Harbour is typically about 0.35 m s^{-1} near Stonecutters Island, 0.85 m s^{-1} in central Victoria Harbour, and increases to about 1.05 m s^{-1} at the eastern harbour entrance; the peak flood velocity is generally smaller at about 70-90% of ebb velocities.

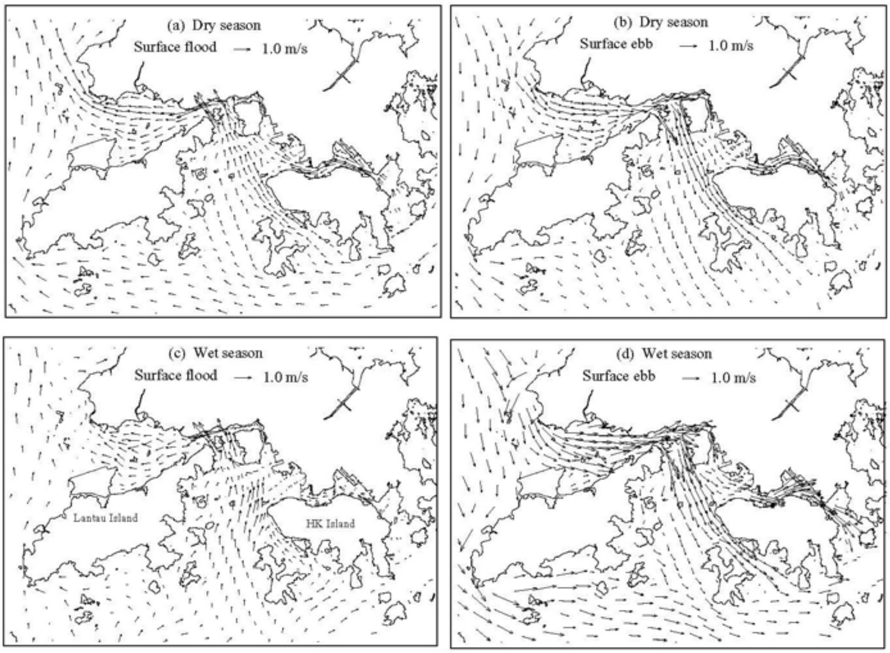
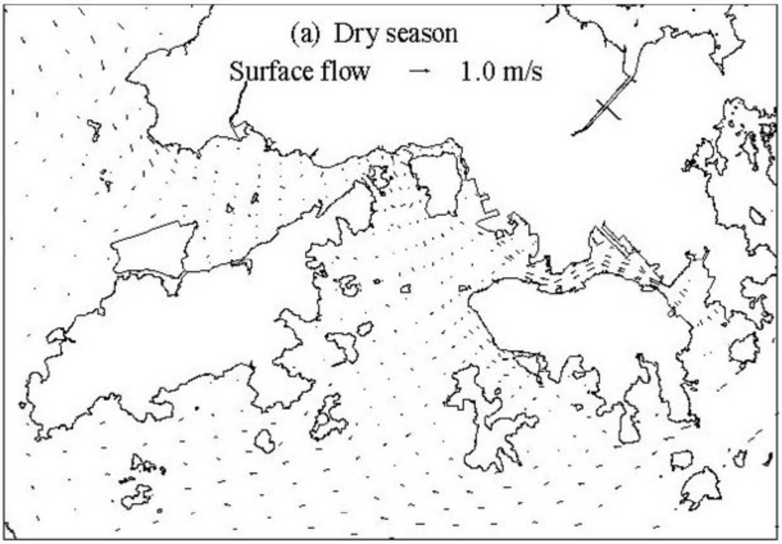


Figure 3. *General surface flow during: a) flood (HHW-3 hr); and b) ebb flow (HHW + 4 hr) in the dry season; and c) flood (HHW-3 hr); and d) ebb flow (HHW + 4 hr) in the wet season.*

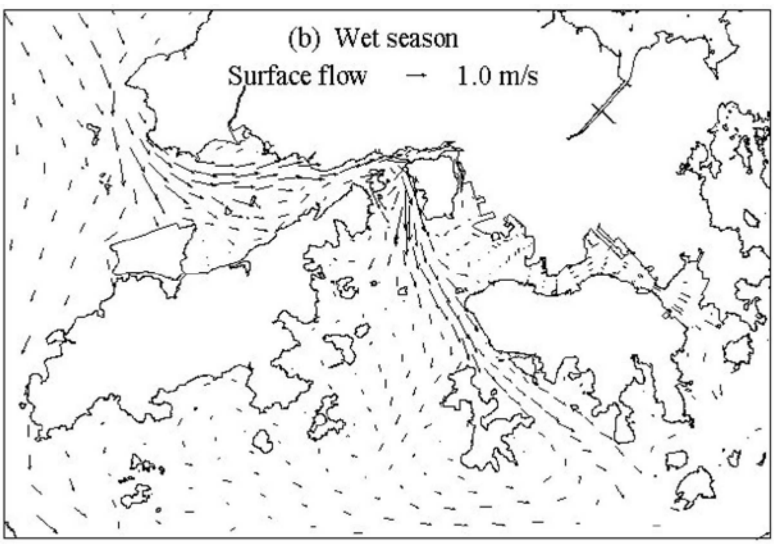
During neap tide, the peak flood and ebb surface velocities are about 77-86% of the spring tide values. The tidal velocity decreases towards the eastern waters; peak ebb velocity in Mirs Bay is about 0.2-0.3 m s^{-1} near the entrance, and <0.1 m s^{-1} inside the bay. In some landlocked tidal inlets (e.g. inner Tolo Harbour), very weak currents of only a few cm s^{-1} are observed. Surface currents in the southern waters (south of Hong Kong Island, Lamma Island and Lantau Island) flow in accordance with the wind generated by the monsoon system. Figures 3c and 3d, and animation 1b, show the surface flow field for the wet season. The predominant wind direction is E/NE in the dry season and W/SW in the wet season. In the wet season, during spring tide, the peak surface ebb velocity in Victoria Harbour is about 0.49 m s^{-1} near Stonecutters Island, 1.2 m s^{-1} in central harbour, and increases to about 1.26 m s^{-1} at the eastern harbour entrance; the peak flood velocity in the harbour is much smaller at about 30-40% of ebb velocities. The peak flood and ebb surface velocities in neap tide are about 68-75% of the spring tide values.

Figures 4a and 4b show the residual currents in the surface and bottom layers in the dry season. The residual flow is mainly from SE to NW through Victoria

Harbour, while residual currents in both directions can be seen in East Lamma Channel and northwestern waters. The coastal current flows from NE to SW in the



Animation 1a. *General surface flow in the dry season. (N.B. Animation 1a corresponds to Figures 3a and 3b).*



Animation 1b. *General surface flow in the wet season. (N.B. Animation 1b corresponds to Figures 3c and 3d).*

southern waters, and the Pearl River turns west at the mouth of the estuary due to the NE monsoon winds and the Coriolis effect (see later discussion).

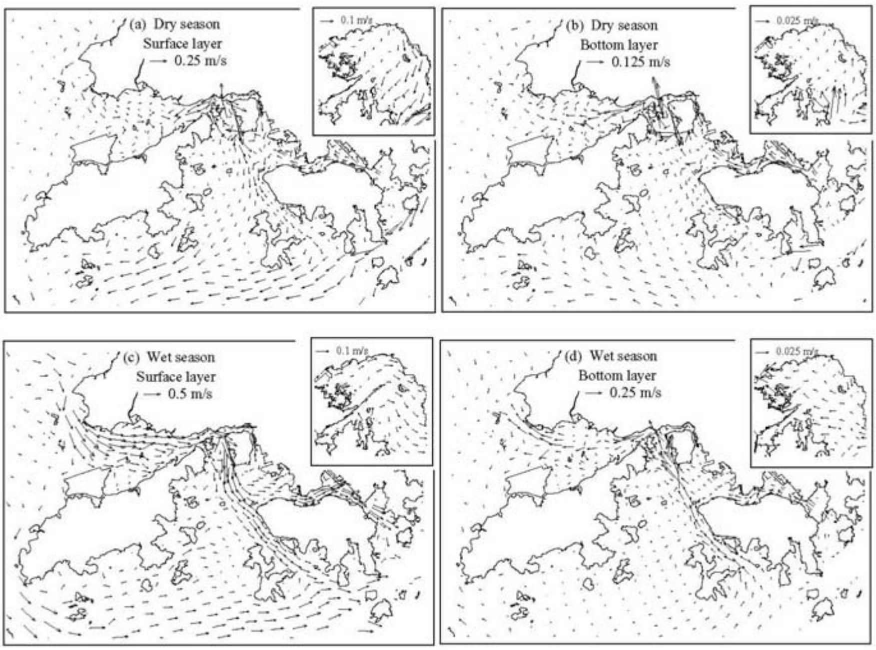


Figure 4. *Surface and bottom residual currents in the dry season (a and b); and in the wet season (c and d). Upper right inset figures show the corresponding residual currents in Mirs Bay. Note the scale difference of current speeds in each figure.*

The maximum depth-averaged residual current is about 0.2 m s^{-1} in Victoria Harbour, 0.1 m s^{-1} in East Lamma Channel, 0.08 m s^{-1} in the western waters, and 0.3 m s^{-1} in the southern waters respectively. There is no apparent vertical structure to the residual flow, although a landward bottom flow opposite to the surface flow can be noted in Mirs Bay and in East Lamma Channel. In the wet season, (Figures 4c and 4d), the flow is strongly influenced by the Pearl River discharge. In Victoria Harbour and northwestern waters, the surface and depth-averaged residual flows are mainly from NW/N/W to SE/S/E, while a residual current in both directions can be found in East Lamma Channel. A bottom residual flow in the opposite direction (from SE to NW), indicative of a two-layer exchange flow, can be clearly noted in Victoria Harbour and East Lamma Channel (note the change in velocity scale for the bottom and top layers). In the southern waters, the coastal current flows from W/SW to E/NE. The maximum depth-averaged residual current varies from 0.25 m s^{-1} in

Victoria Harbour, 0.15 m s^{-1} in East Lamma Channel to 0.3 m s^{-1} in the southern waters. For the wet season, the residual current in the surface layer is much stronger than in the bottom layer due to freshwater inputs from the river and rainfall.

2.2. Salinity structure

The hydrography is also intimately related to the salinity structure of the coastal waters. The coastal water is a mixture of fresh water derived from the river discharge (with low salinity) and oceanic shelf-water with relatively high salinity. In the dry season, Hong Kong waters are vertically well-mixed due to very low Pearl River discharge, strong tidal mixing and winds. In the dry season, the salinity (and temperature) is approximately vertically homogenous; salinities may vary from a low of around 15 at the mouth of the Pearl Estuary to 34 in the eastern waters (Figure 5a). Salinity intrusions may reach to beyond Deep Bay (Figure 5a), and Mirs Bay in the eastern waters are clearly unaffected by the Pearl River estuarine plume.

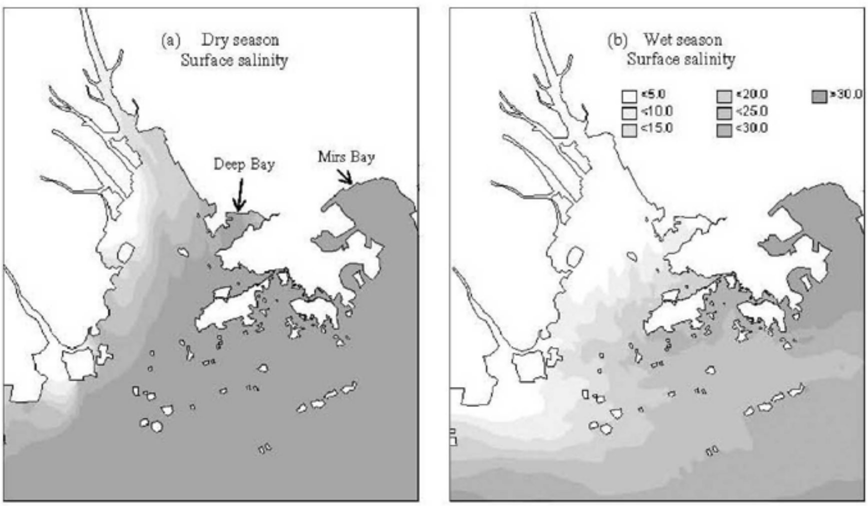


Figure 5. *General surface salinity field during flood at HHW-3hr in (a) dry season; and b) wet season.*

In the wet season, however, the interaction of the Pearl River discharge and tidal currents creates significant vertical and horizontal salinity gradients (Figure 5b). The western waters are strongly influenced by the Pearl River. For example, the surface salinity in the northwestern waters can be as low as 15, while the bottom salinity exceeds 32. By comparison, Victoria Harbour and East Lamma Channel are relatively more affected by tidal mixing, and the salinity differential is much less pronounced (< 7). The eastern waters are relatively sheltered from the Pearl River, and hence they are more oceanic with a typical salinity of 32-34 and much weaker vertical salinity differentials. In the southern waters, the salinity differential is about 1 to 4, except for the fresh water main path in the deeper channels such as East Lamma Channel (with differentials in the range of 6-10).

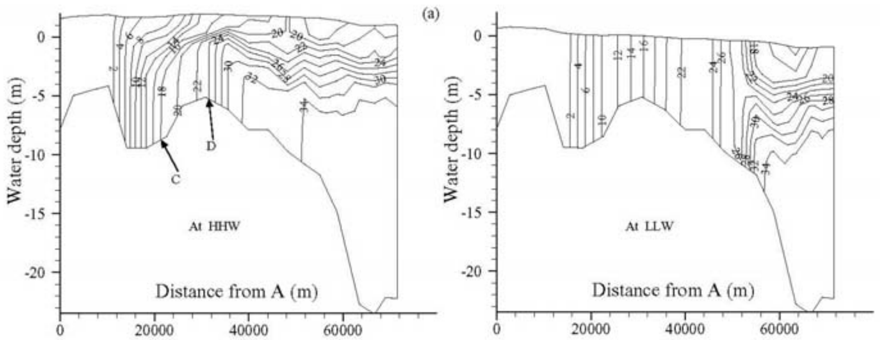


Figure 6a. *Computed salinity contours in vertical section along a transect down the axis of the Pearl river estuary from A to B (see Figure 1a) at Higher High Water (HHW) and Lower Low Water (LLW) of a spring tide in the wet season for a Pearl River discharge of 19422 m^3 s^{-1}.*

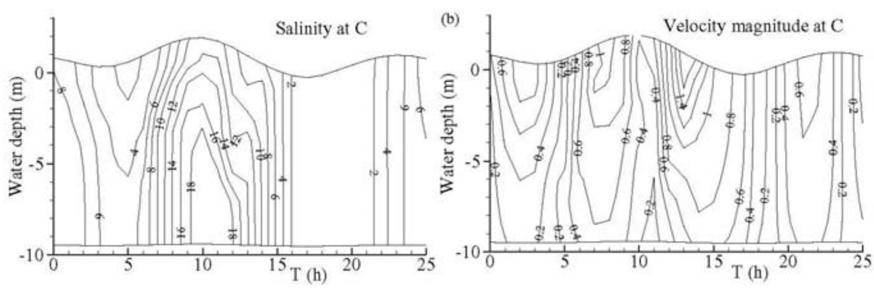


Figure 6b. *Computed depth-time variation of salinity and velocity magnitude during a spring tide at location C in the wet season.*

In the wet season, the fresh river water spreads over the sea surface; the turbulent entrainment (mixing) of the more saline lower layer into the surface layer sets up a saline wedge intrusion that moves shoreward. Figure 6a shows the salinity contours in a vertical section along a transect down the axis of the Pearl River, from A to B

(Figure 1a). Under an average wet season river discharge (Q = 19,422 m^3 s^{-1}), the bottom salinity of 18 may intrude to the west of Deep Bay (point C, in Fig. 1a) at higher high water (HHW), where the surface salinity is about 7. The bottom salinity of 18 moves southward with the ebb flow, and reaches point D at LLW, after a horizontal tidal excursion of about 12 km. At lower low water (LLW), the vertical salinity distribution in the shallow estuary area is vertically well-mixed. Figure 6b shows the temporal changes in water level, salinity, and velocity at point C, where the salinity can change significantly within a tidal cycle. Pronounced vertical gradients can be observed around high tide, when the velocity is the weakest (high Richardson number and limited vertical mixing). In general, the vertical salinity gradients are much weaker in Victoria Harbour and adjacent waters.

2.3. Flushing Time

The complicated coastal currents and turbulent mixing determine the tidal flushing of algae, nutrients, or any pollutant in the coastal water – i.e. the rate of exchange of the coastal waters with the “clean” open sea. The flushing rate in a water body of interest (e.g. a tidal inlet or fish farm) can be precisely determined using 3D hydrodynamic and transport models via a numerical tracer experiment (Choi and Lee, 2004). A unit tracer concentration is initially prescribed inside the entire bay and zero elsewhere; the subsequent decrease of tracer mass in the region of interest is then tracked. The mass removal process can be well-approximated by a double-exponential decay curve – from which the flushing time (inverse of flushing rate) can be analytically determined. In general, assisted by the density-driven estuarine circulation, the flushing rate in the wet season is substantially greater than in the dry season. For example, the flushing time in Victoria Harbour is about 1.5-2.5 days in the wet season and 5-7 days in the dry season (Kuang and Lee, 2004). By comparison, the flushing times of inner and outer Deep Bay are 20.9 and 3.8 days respectively (Qian, 2003); in view of the vertically well-mixed nature of the shallow bay (average depth of 3 m), the flushing characteristics are similar in the dry and wet season. In contrast, the flushing in many of the semi-enclosed tidal inlets in the eastern waters (Tolo Harbour, Mirs Bay, Port Shelter) is very weak; the flushing times for inner Tolo Harbour are 38 and 14.4 days for the dry and wet seasons respectively (Choi and Lee, 2004).

3. PHYSICAL – BIOLOGICAL INTERACTIONS DURING THE WET AND DRY SEASONS

Biological processes are coupled with the hydrodynamics. During the winter dry season, when the river water moves out of the estuary, the northeast monsoon wind and the Coriolis force move the estuarine water to the west side of the estuary (Figure 5a). There is little contribution of the Pearl River to Hong Kong waters. Instead, the NE monsoon wind advects coastal-shelf surface water with relatively high salinity and low nutrients into Hong Kong (Yin, 2002). These coastal waters are N-poor relative to P (N:P ~10:1 or less). Using the Redfield ratio of 16N:1P as a guideline for the requirement of phytoplankton growth, these winter surface waters with a N:P<16:1 could potentially become nitrogen-limited, especially in the eastern waters (Yin, 2002). Due to the tidal mixing and low river discharge, vertical gradients in temperature, salinity, and nutrient concentration are negligible. Since nutrient concentrations are low (NO_3 < 10 μM), chlorophyll concentrations should not exceed 10 μg l^{-1} (assuming 1 μM NO_3 produces 1 μg chl l^{-1}). In general, winter chlorophyll concentrations in Victoria Harbour and southern waters are usually < 5 μg l^{-1}. The relatively low chlorophyll concentration can be attributed to the strong flushing in these relatively open waters. In addition, in regions with strong tidal currents, vertical mixing can transport phytoplankton cells to depth and probably reduces the light availability for phytoplankton growth; this effect can be enhanced by strong wind and/or heavy navigation traffic. Other factors such as zooplankton grazing may also play a role (Yin, 2002). On the other hand, algal blooms have been observed in the winter in poorly flushed and eutrophic tidal inlets in the eastern waters (Lee et al., 1991a; Lee and Lee, 1995).

In spring, during the transition between the wet and dry monsoon seasons, red tides (coloured waters) occur most frequently, especially in the eastern waters (e.g. Mirs Bay) (Yin, 2003). The species are mostly dinoflagellates, and especially visible is the bright red, surface-forming blooms of *Noctiluca,* a heterotrophic dinoflagellate that consumes bacteria, phytoplankton and even fish eggs. Unlike the high biomass diatom blooms that occur in the summer, these *Noctiluca* blooms do not contain chlorophyll. Low silicate concentrations (often < 3 μM Si) may allow dinoflagellates to out-compete diatoms at this time of year (Yin, 2003). In general, the flushing rate, water transparency, transport by tidal currents, aggregation by wind and/or vertical migration are the most likely contributing factors that may increase the biomass beyond concentrations that could be supported by the ambient nutrient concentrations (Yin, 2003; Lee and Qu, 2004; Wong, 2004). Unlike most temperate regions, pronounced large-scale spring blooms that last for an extended period of time are not common in Hong Kong waters.

In the summer, the Pearl River carries large nutrient loads derived from anthropogenic sources. Nutrient sources include sewage, river discharge and rainfall. The 10-year EPD dataset (1991-2000) in Figure 2 clearly shows a marked increase in concentrations in the western waters. Yin and Harrison (in press) reported elevated NH_4 and PO_4 concentrations in Victoria Harbour due to sewage input. When surface water temperatures are nearly 30 ^{o}C, blooms can develop quickly (<1 week) in the stably stratified water under low wind conditions. Many high biomass blooms can occur during this biologically dynamic period. Total inorganic nitrogen (NO_3 + NH_4) is >100 μM in the PRE (Yin et al., 2000 and 2001), yielding a N:P ratio that is about 7 times higher than the normal Redfield N:P ratio of 16N:1P. Similarly, rainfall contributes much more N than P (N:P = 50:1) (Yin, 2002). In contrast, sewage is N poor relative to P (N:P = 8 to 10:1). The natural nutrient sources are relatively nutrient poor deep water from the coast that is brought to the surface by the two-layer flow due to summer upwelling and estuarine circulation (Figure 4). While nutrient ratios may indicate which nutrient may become limiting for algal growth, it is the concentration of the nutrient that determines the amount of algal biomass produced (the algal yield) and hence the amount of oxygen consumed during algal decomposition in the bottom waters (Yin, 2002). Since the N:P > 16:1 in summer in the western and southern waters of Hong Kong, then P will potentially limit the amount of algal biomass produced (Yin et al., 2000 and 2001), similar to other coastal areas in China (Harrison et al. 1990). Therefore, reducing the concentrations of P should reduce the amount of algal biomass produced (additional Biological Oxygen Demand, BOD) and decrease the potential occurrence of hypoxia (Yin, 2003).

While nutrient ratios indicate which nutrient may potentially limit algal growth, an algal bioassay or nutrient addition bioassay is one common way to confirm which nutrient will actually become limiting to growth. In a series of experiments with bioassays taken from areas with different N:P ratios, it was shown that an addition of N produced the largest increase in chlorophyll when N:P = 5:1 for Mirs Bay (Yin et al., 2001). In contrast, for the southern waters where N:P = 200:1, an addition of P produced the largest increase in chlorophyll. Hence, there was a shift from potential P limitation in the southern waters to N limitation in the eastern waters over a relatively short distance of 40 km (Yin et al., 2001).

During the summer, the winds reverse direction and the SW monsoon winds cause upwelling (Li, 1993); coastal deep water is brought into the Hong Kong area (Yin, 2002). The estuarine plume extends across most of the PRE and Hong Kong waters (Figure 5b). Nutrients increase as salinity decreases, and there is a sharp surface salinity gradient from the western low salinity waters to the eastern Mirs Bay area, where there is little influence of the Pearl River (Figure 5b). Through the process of estuarine circulation, this seaward flow of nutrient-rich surface river plume entrains a near bottom return flow (deep coastal water) that is relatively low in nutrients and dissolved oxygen (Yin, 2002). The mixing set up by this estuarine circulation (Figure 6) has the effect of diluting the nutrient-rich river water, resulting in lower nutrient concentration and hence potentially lower algal biomass produced. Considering the high nutrient loads in summer, the efficient estuarine mixing may explain why eutrophication impacts in Hong Kong waters are not more serious than presently observed (Yin, 2002).

4. PHYSICAL-BIOLOGICAL INTERACTIONS – EPISODIC EVENTS

An understanding of physical processes can aid greatly in the interpretation of the myriad of observations of harmful algal blooms (HAB). It can also directly benefit fisheries management. First, harmful algal blooms are often observed in weakly-flushed tidal inlets (Tolo Harbour) and in the eastern waters of Hong Kong (Mirs Bay), where many open cage fish farms are located (Lee et al., 2003 and 2005). The fish farms are threatened by severe dissolved oxygen depletion associated with algal blooms and red tides (Lee et al., 1991b). On the other hand, mariculture activities also contribute to pollution caused by unconsumed fish feed, leaching, as well as direct waste discharge from the fish farms. The sustainable management of mariculture requires proper siting of the fish farms and stocking density control. Both of these are related to the carrying capacity of the water body concerned, which is mainly governed by its flushing characteristics. Based on the flushing rate, a simple and effective water quality model can be developed to determine the carrying capacity – i.e. the maximum allowable fish stock density so as not to exceed a minimum target dissolved oxygen or chlorophyll *a* level (Lee et al., 2003).

In the eutrophic waters around Hong Kong, light availability rather than nutrient concentration is often the limiting factor for algal growth. For example, only a few red tides have ever been reported in Deep Bay (Figure 1a) in the west, which receives significant nutrient input from the heavily polluted Shenzhen River and Hong Kong. This is in contrast to a record 280 red tide occurrences (1975-2001) in Tolo Harbour that is relatively free from the influence of the Pearl River. This can be attributed to the low flushing rate in Tolo (of the order of 0.04 per day) and higher water transparency (Qian, 2003). Compared to Tolo Harbour, the suspended solids concentration in Deep Bay is significantly higher (typically 30 mg l^{-1} as opposed to 3 mg l^{-1}), resulting in much greater light extinction and hence light limitation to phytoplankton. Coupled with a stronger flushing rate (0.063 day^{-1}), it can be shown that the potential maximum net algal growth rate (in the absence of nutrient and other limitation) is on the order of 0.4 day^{-1} for Tolo Harbour and negligible for Deep Bay (Qian, 2003).

4.1. Hydrodynamic Tracking of the Spring 1998 Massive Red Tide in Hong Kong

When an algal bloom is formed in a weakly flushed tidal inlet, it can be transported to other parts of Hong Kong waters where it may cause fish kills and interfere with beneficial uses (e.g. beach closures). Catastrophic events sometimes occur due to a combination of wind and tidal conditions. A case in point is the devastating massive dinoflagellate (*Karenia digitatum*) red tide in March-April 1998 (Lee and Qu, 2004). According to newspaper reports based on scattered observations of fishermen, the 1998 red tide first appeared in the Mirs Bay area on March 18, 1998 and was later transported southwards into East Lamma Channel, where the most severe fish kill occurred (Figure 1). Red tide events were reported in Port Shelter, Victoria Harbour, East Lamma Channel, and south of Hong Kong Island.

A surface drogue tracking method has been used to study algal bloom transport (Lee and Qu, 2004). Drogues are floats that are released and recovered at a specified time, which are used to monitor the path of a particle moving with the surface current. The trajectory of an algal bloom initiated in the northeast Hong Kong waters (as frequently observed) under different seasonal, wind and tidal conditions was modelled. Mirs Bay was divided into four regions; about 100 numerical drogues were released from each of these regions and tracked for 40 days. The experiment was performed for 16 release times corresponding to different tidal conditions (covering spring-neap cycles and tidal stages). Figures 7a and b show the model-derived bloom tracks for the general wet and dry seasons respectively. In the wet season, any HAB initiated in Mirs Bay will move in a clockwise direction out of the bay, and be transported away from Hong Kong due to SW monsoon winds which drive the SW to NE coastal current. This concurs with the observation of low HAB events in the summer (Yin, 2003). In the dry season, a bloom initiated in northeast Mirs Bay tends to move in an anti-clockwise direction and to by-pass Victoria Harbour and be carried away into southern waters due to the NE to SW coastal current that is driven by the NE monsoon winds; the bloom typically flows past the east edge of Port Shelter, Victoria Harbour and East Lamma Channel (Figure 7). Numerical experiments show that for the dry season on average, only about 2% of the tracks enter East Lamma Channel. However, in Spring 1998, a bloom released from northeast Mirs Bay around March 18-19 found its way into East Lamma Channel (Figure 8b); the travel time was on the order of 12-17 days. The calculations show that the strong NE to E wind during March 21- April 5 (Figure 8a) plays a key role in explaining the surface movement of the observed massive red

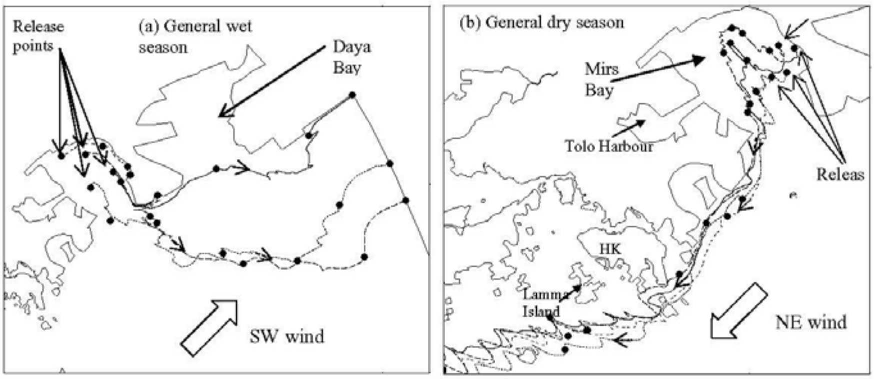


Figure 7. *Algal bloom transport patterns for: a) general wet season; and b) general dry season with model release points in Mirs Bay in both cases.*

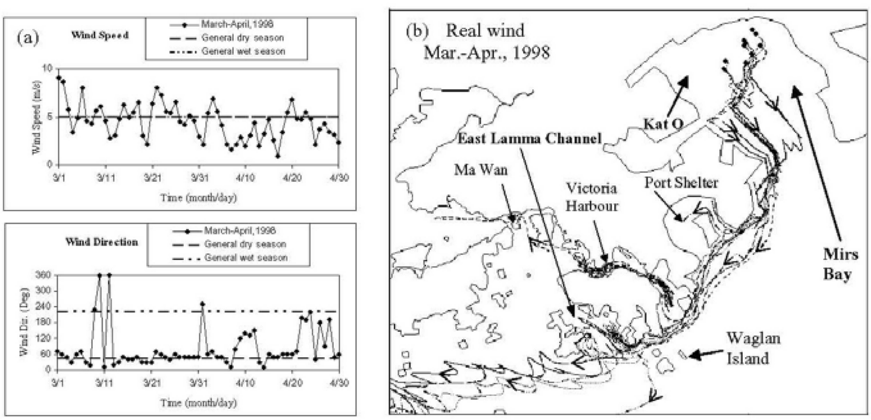


Figure 8. *Daily prevailing wind speed and direction (10 m above Mean Sea Level (MSL)) at Waglan Island in March-April, 1998; b) Model derived algal bloom transport patterns for Spring 1998 conditions.*

tide and the subsequent fish kill in Port Shelter and Lamma Island. The diurnal tide at the beginning of April (see Figure 1b inset) increased the flood duration and transported the bloom into East Lamma Channel, resulting in the massive fish kill (Figure 8). Under these wind and tidal conditions, the percentage of bloom tracks (or probability) moving into East Lamma Channel, Victoria Harbour and Port Shelter is significantly increased from 4.9 to 25.7%.

4.2. Wind Mixing and Stratification

Finally, vertical mixing as a function of wind and tidal conditions, can play a decisive role in the formation of algal blooms. A real time field monitoring system has been developed to continuously track the changes in chlorophyll fluorescence, dissolved oxygen and other hydro-meteorological variables at two weakly flushed sites in Mirs Bay and Lamma Island (Wong, 2004; Lee et al., 2005). During 2000-2003, the system successfully tracked 19 algal blooms. In the shallow weakly-flushed coastal water (depth 7-10 m, tidal current 0.05-0.19 m s^{-1}), the bloom is short-lived, typically lasting a few days to over a week, with chlorophyll and DO concentrations in the range of 20-40 mg m^{-3} and 2-15 g m^{-3} respectively.

Based on the field data and three-dimensional hydrodynamic calculations (Wong, 2004), it was demonstrated that the vertical stability of the water column plays a key role in bloom formation (Wong and Lee, 2004). It was also shown that the effect of turbulent mixing is important in the selection of bloom formation species at these two weakly flushed locations. The first order theory considers the algal dynamics that occur in a vertical water column, and incorporates the essential processes of algal growth and settling, vertical mixing, light penetration, nutrient availability and species competition for nutrients. Essentially, the model reveals that vertical turbulent diffusivity is a key controlling factor in the occurrence of algal blooms. Given a minimum threshold level for nutrient availability, the vertical turbulent mixing must be below a threshold level for blooms to develop. Further, for very calm waters, dinoflagellates would be able to out-compete diatoms by virtue of their ability to obtain nutrients through vertical migration. The model has been validated against field observations (Wong, 2004; Wong and Lee, 2004).

5. CONCLUDING REMARKS

It is well documented that in many estuaries in the world that receive high nutrient loads, extensive and sustained algal blooms occur in the summer months with the subsequent development of sustained hypoxia in the bottom waters (NRC, 2000). Off the mouth of the Mississippi River, an extensive dead zone ($O_2 < 2$ mg l^{-1}) develops during most summer months due to the decomposition of algal blooms at depth (Rabalais et al., 2002; Wiseman et al., 2004). Similarly, off the mouth of the Yangtze River (China's largest river), a monospecific dinoflagellate bloom develops during the summer along with hypoxia in the bottom waters (Lu et al., 2002). In many respects, the Pearl River's nutrient loads are similar to those of the Mississippi and Yangtze Rivers. Therefore we posed the question at the outset: "Why aren't the impacts from eutrophication worse than they are"? Hong Kong waters have no extensive and sustained algal blooms (chlorophyll *a* seldom > 10-15 μg l^{-1}) and only very episodic events of hypoxia occur in late summer (Yin et al., 2004a).

Recent field observations and modelling studies have facilitated an integrated understanding of eutrophication impacts. In this paper, we have given a general overview of temporal and spatial patterns of algal blooms in Hong Kong waters. Key features of the estuarine hydrography are summarized, and algal dynamics have been shown to depend strongly on physical and biological interactions. Physical processes drive biological processes. Despite the fact that nutrient concentrations are generally above the threshold required to trigger a bloom, large-scale and sustained blooms are not a common occurrence. First, tidal flushing in Victoria Harbour and most of Hong Kong waters is efficient in both the dry and wet season. Any algal species introduced into a given water body (e.g. from ship ballast water) will not have sufficient time to develop into a bloom before one of the growth limiting factors reduces growth (e.g. favourable hydro-meteorological conditions may not prevail). Second, wind and tidal mixing (barotropic and baroclinic) can mix phytoplankton cells downwards out of the photic zone, and discourage the formation of blooms. High light extinction due to high sediment concentrations from the river in summer is another impediment to bloom formation.

Eutrophication impacts can also be traced to a number of causes ranging from a chance combination of wind and tidal conditions, to a possible El Nino influence (Yin et al., 1999) and variability of nutrient limitation across the PRE. However, potential impacts in summer may be reduced by upwelling since the nitrogen-rich surface water from the PRE mixes with and is diluted by the upwelled water with relatively lower nitrogen concentration. This effect of upwelling is quite the opposite in other areas where upwelling increases the nitrogen concentration in the surface layers. In general, intense algal blooms can develop in semi-enclosed and poorly-flushed tidal inlets with good water transparency and long residence times, and under low wind conditions. Other factors that require further study may include the effects of temperature and turbulence on algal growth, algal sedimentation, and zooplankton grazing.

6. ACKNOWLEDGEMENT

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CHAPTER 14

MARINE COMMUNITIES AND INTRODUCED SPECIES IN PEARL HARBOR, O‘AHU, HAWAI‘I

STEVE L. COLES

1. INTRODUCTION

Harbours are protected inlets or embayments that provide shelter and anchorage or port facilities for vessels from small craft to giant super tankers or bulk-cargo carriers. Often, but not always, harbours are restricted from the open ocean by narrow entrance channels, and the water quality within the harbours may differ dramatically from the open ocean, usually with higher turbidity, organic particulates, dissolved organic matter, inorganic nutrients and primary productivity. Also, because harbours are usually surrounded by industrial developments and urban areas, a harbour often receives a wide variety of man-related effluents that may remain confined within the harbour because of limited mixing with the surrounding ocean.

These environmental characteristics often favor the spread and establishment of introduced marine species from one organic-rich harbour to another. Even before recorded history, vessels moving from one place to another provided a mechanism for the spread of marine organisms from their original habitats to new areas. Known variously as “nonindigenous” “introduced”, “exotic” or “non-native” species, these organisms may become a normal or even dominant component of the harbour communities into which they are introduced. Vessel movement of introduced species on a world-wide scale probably began at least 500 years ago, with long distance transport on slow-moving ships of fouling organisms that had accumulated while the vessels remained at anchor for months or even years in harbours or bays.

Pearl Harbor (Figures 1 and 2), called *Ke-awa-lau-o-Pu’uloa* (The Many Harbors of Pu’uloa), or *Awawa-lei* (Garland of Harbors) by Hawaiians before European contact (Handy and Handy, 1972), is the largest and most enclosed harbour in the state of Hawai‘i, and, for approximately the last hundred years, has been the site of the largest U.S. Navy base in the central or western Pacific. Hawai‘i and especially the Island of O‘ahu has been a major crossroads of Pacific ship traffic

E. Wolanski (ed.), The Environment in Asia Pacific Harbours, 207–228.

__Figure 1.__ Landsat image showing Pearl Harbor with Honolulu and the Ko'olau Mountains to the right. For a horizontal scale see Figure 2. Source: http://www.terrainmap.com/

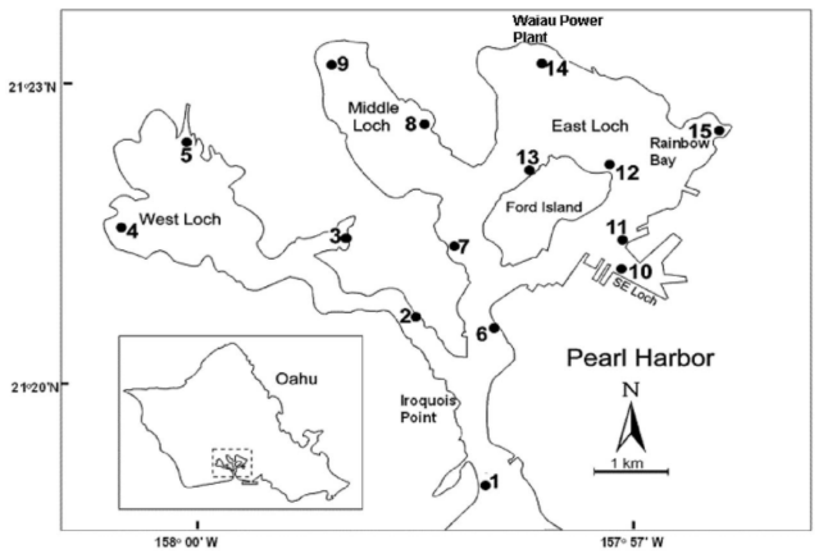

__Figure 2.__ Map of Pearl Harbor showing locations of major lochs, Ford Island and sampling locations of 1996 introduced species study.

for approximately 200 years, first as whaling port in the first half of the nineteenth century, later for the export of sugar that was the mainstay of the Hawaiian economy until the last few decades. Although most of this shipping activity was focused on nearby Honolulu Harbor, military ship traffic passing through Pearl Harbor, particularly during and following World War II, provided ample opportunity for marine introductions to occur from ships, barges and dry docks that were brought into the harbour from various ports throughout the Indo-Pacific. Moreover, Pearl Harbor has historically been the site of numerous intentional introductions of non-native species for commercial purposes.

Species introductions have been proposed to rank among the most serious potential disturbances of marine ecosystems (Carlton, 1994), and have resulted in reduced populations of native organisms, extensive alterations of communities and degraded habitats. However, detailed knowledge of the identity and composition of marine introductions relative to native species has only become available for Hawai‘i and a few other areas in the tropical Pacific in the last decade. The series of studies that has provided this information began with surveys that were conducted in Pearl Harbor from 1996 to 1999. This chapter will summarize the findings of that study within the context of historical and biological information available for Pearl Harbor and compare the findings with similar studies conducted elsewhere in Hawai‘i and the tropical Pacific.

2. ENVIRONMENTAL SETTING OF PEARL HARBOR

The main Hawaiian Islands are among the most isolated land areas in the world, lying more than 4300 km from the nearest major landfalls in North America and the South Pacific and more than 6400 km from Japan, the nearest Asian land mass. Pearl Harbor is a coastal plain estuary located between the Ko‘olau and Wai‘anae mountain ranges in central O‘ahu, Hawai‘i (Figure 1). The harbour is the most landlocked large estuarine body of water in Hawai‘i and has about 21 km^2 of surface water area with a mean depth of 9.2 m and about 58 km of shoreline. It is composed of three main coves or lochs, which are remnants of drowned river valleys joined together by a narrow main channel that connects the harbour with the open ocean. With this constricted opening the harbour is isolated from oceanic circulation, and water exchange of the harbour with the open ocean is slow. Residence time of water within the harbour has been estimated as about six days maximum for bottom water and one to three days for surface water (Grovhoug, 1992).

Water temperature in the harbour varies annually from 23 to 29 ^oC, (Evans et al., 1974) and salinities from 10 to 37 (mean 33). Salinity is highly influenced by terrestrial and groundwater runoff, especially at the heads of the three main lochs called East, Middle and West Loch, respectively, with a smaller area sometimes called Southeast Loch adjoining East Loch east of Ford Island (Figure 2). This area is the site of most U.S. Navy ship operations in Pearl Harbor. The harbour receives five perennial streams and three intermittent streams draining approximately 285 km^2 of watershed and also receives the discharges from five large springs along the lochs' shorelines. Warming of surface water and freshwater discharge contribute to the development of a pronounced vertical density stratification of water in the harbour, which in turn promotes differing current conditions between surface and bottom water and relative isolation between surface and bottom water masses. Surface water circulation is primarily driven offshore by northeast tradewinds. Weak tidal flood and ebb flows of 0.15-0.3 m s^{-1} control the movement of bottom water in and out of the harbour, driven by a mean annual tidal range of approximately 0.5 m (Grovhoug, 1992).

Vegetation along the shoreline is dominated by the introduced red mangrove (*Rhizophora mangle*) at the heads of the three main lochs, forming dense growths of bushes and trees up to 10 m high. Elsewhere the shoreline vegetation is cultivated grass, trees and plants in populated areas and kiawe trees (*Prosopis* sp.) along channels.

The water of Pearl Harbor has apparently always been relatively turbid from stream runoff and other sources of sediment. A traditional Hawaiian chant recites "Ewa's lagoon is red with dirt/…A plumage red on the taro leaf/ An ocherous tint in the bay" (Emerson, 1972). However, runoff-related sedimentation undoubtedly increased dramatically in the nineteenth century with deforestation, ranching and grazing of hillsides, declining use of taro ponds which would act to retain storm water, and development of sugar cane cultivation. S. Bishop (1901, in Sterling and Summers, 1978) described her memories of Pearl Harbor of 1836: "The lochs or lagoons of Pearl Harbor were not then as shoal as now. The subsequent occupation of the uplands by cattle denuded the country of herbage and caused vast quantities of earth to be washed down by storms into the lagoon". This runoff resulted in the harbour historically being a highly turbid environment, with thick deposits of fine silt on the bottom throughout most of the lochs. Stream input of sediments have been estimated to exceed 97 thousand metric tons annually, and maintenance dredging of about seven million m^3 has been required by the Navy on four to five year cycles (Grovhoug, 1992). Turbidity measurements indicated by Secchi disk readings in 1990 averaged only 2.5 m harbour-wide (range 0.5-3.5 m), resulting from suspended sediments from land runoff and organic material produced by eutrophic conditions (Grovhoug, 1992).

Pearl Harbor has been the center of Pacific Naval Operations and the location of the Pearl Harbor Naval Base since early in the twentieth century, with berthing and maintenance facilities for military vessels of all types. In addition, two recreational marinas are located in the harbour, one at Iroquois Point near the channel entrance and one at Rainbow Bay at the head of East Loch. Development of the naval base and urbanization of watershed areas greatly altered the shoreline and quality of water entering the harbour in twentieth century. At one time more than 100 treated or untreated sewage discharges were estimated to enter the harbour, and coliform bacterial levels indicated extremely polluted conditions (Cox and Gordon, 1970). Heavy metals and pesticides in sediments indicated further environmental degradation. These conditions were largely abated in the 1980s with the removal of sewage effluents from the harbour and changes in naval operations (Grovhoug, 1992).

Early reports described an abundance of fish and shellfish in Pearl Harbor and the importance of the area as a major Hawaiian population center with numerous and extensive fish ponds providing a major source of food. According to Handy and Handy (1972) the bays of the harbour "offered the most favorable locality in all the Hawaiian islands for the building of fish ponds and fish traps into which deep sea fish came on the inflow of tidal water... (the bays) provided a greater variety and abundance of edible shellfish, and were famous as the summer home of mullet". Like many aspects of Hawaiian culture, fishponds and fishing in the harbour declined in the nineteenth century. However, more than 30 fishponds still existed by the early 1930s (Figure 3) and oysters introduced in the 1920s thrived for a time (Costa-Pierce, 1987). By 1972 the number of fishponds had decreased to four, and 99% of the oysters in Pearl Harbor died that year from an undetermined cause that appeared related to a fungal infection (Kawamoto and Sakuda, 1973). Even at that time, however, an extensive survey of the harbour's biota revealed a diverse and abundant estuarine community (Evans et al., 1974), and abundant fish and invertebrates continued into the 1990s when water quality improved (Grovhoug, 1992).

3. HISTORICAL PERSPECTIVE

Prior to the arrival of Europeans in the Hawaiian Islands in the late eighteenth century, there was no communication or trade with the outside world, and no need for deep harbours, because even the largest Hawaiian outrigger canoes could access most open shorelines and beaches throughout the archipelago. In the 81 years after European discovery of the Hawaiian Islands in 1778 more than 300 ships from foreign ports made landfall in Hawai'i. The maximum number of arrivals occurred in the 1840s, coinciding with the peak of whaling activity and the discovery of gold in California (Judd, 1929). Shipping traffic continued to increase as steam replaced sail and Hawai'i's shipping requirements expanded with urbanization and development of the plantation-based economy. Access to a deep harbour and port facilities was the major reason for relocating the capital to Honolulu, which became the major population center for the islands during the 1800s. However, Pearl Harbor, with its narrow entrance only 10 km to the west of Honolulu Harbor, remained relatively isolated and undeveloped until the early twentieth century when construction of the U.S. Navy Base began.

Important historical events that have drastically altered the physical setting and ecology of Pearl Harbor, can be grouped into four main periods. In the first period, prior to the twentieth century and the construction of the Pearl Harbor entrance channel, the harbour was more restricted from the open ocean than at present. A sand barrier at the entrance limited depth to about 5 m and prohibited access of all but the most shallow draft ocean-going vessels. With a more restricted ocean access, the harbour is likely to have been less saline and more estuarine but with a more pristine environment than occurred following European contact and development of the harbour. This more pristine condition continued into the nineteenth century when the harbour was intensively utilized by Hawaiians and supported the abundant fishponds and shellfish, including small pearl oysters from which the harbour derived its English name. Water quality was reportedly high and sedimentation and turbidity low until damaging land practices in the mid-nineteenth century began to increase the sediment load of land runoff reaching the harbour. This was also a time of the first introduction of non-native marine and brackish water species to the harbour. The first attempt in 1866 to introduce a non-native species, the commercial oyster *Crassostrea virginica*, (Kay, 1979) apparently was of only limited success, perhaps because of relatively low levels of phytoplankton productivity or detrital food in the water at that time. Later intentional introductions were mosquito fish (*Gambusia affinis*), sailfin mollys (*Poecilia latipinna*) and killifish (*Fundulus grandis*) in 1905, and the red mangrove (*Rhizophora mangle*) which probably began colonizing the harbour shores not long after it was introduced to Molokai in 1902 (Wagner et al., 1990).

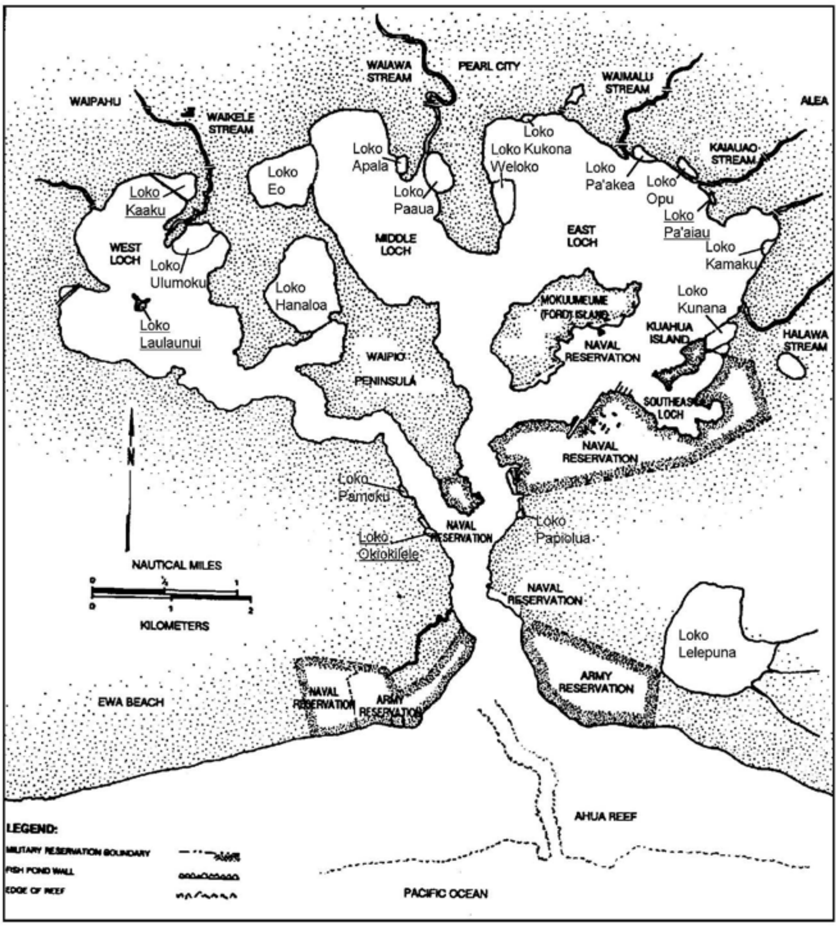

Figure 3. *Pearl Harbor in about 1920, showing many of the fishponds that still remained after the initial development of the naval base. Of these only the four underlined still existed in 1972 (adapted from Grovhoug, 1992 and based on an undated Oahu Fisheries chart).*

The second period was from about 1910 to 1940, beginning with the completion of deepening of the entrance channel and ending with the start of World War II. Following annexation of Hawai‘i to the U.S. in 1898 the U.S. Navy proceeded to develop Pearl Harbor as a coaling station and ship dry dock. As part of this effort, the harbour entrance channel depth was nearly doubled to 9 m and widened to approximately 60 m from 1901 to 1910. The channel was officially opened for navigation in 1911, and many habitats of the harbour were soon drastically altered as shorelines, especially in the Southeast Loch and Ford Island areas, were converted to docks and naval operations facilities. Formerly shallow areas in the lochs were deepened by dredging to accommodate ships, and fish ponds in the vicinity of the naval base were filled with dredged material. Urbanization of the East Loch area progressed as the Pearl City area was developed, and the Hawaiian Electric Company's Waiau power station began discharging heated effluent in 1938 at the head of the harbour's East Loch. Discharge of sewage waste, pollution by metals from shipyard activities and nutrient enrichment of waters receiving runoff from sugarcane cultivation can all be assumed to have increased greatly during the 1910-40 period. Further attempts to introduce non-native eastern oysters (*Crassostrea virginica*) as commercial species were made, which apparently established a breeding and thriving population, especially in West Loch. Introductions of Japanese clams (*Tapes japonica*) and Japanese oysters (*Crassostrea gigas*) were also attempted (Brock, 1960).

The Japanese attack on the Pearl Harbor Naval Base in December, 1941 marked the beginning of the third period when the harbour was most drastically altered and impacted by man-related activities. From 1940 to 1970, Pearl Harbor ship traffic and shipyard activities were at their peak and the environmental quality of the harbour reached its lowest point. Alteration of the shoreline and near-shore areas in the harbour continued, and all but four of the more than 30 fishponds that had still remained in 1920 were eliminated. The harbour became a receiving vessel for an estimated more than 100 treated or untreated sewage discharges and uncontrolled runoff from sugarcane plantations and mill wastes (Anonymous, 1969; Cox and Gordon, 1970). Non-point pollution sources from hillsides under urban development and naval shipyard activities further degraded water quality. Coliform bacterial counts at stream mouths in East Loch and near oyster beds in West Loch ranged from hundreds of thousands to billions of bacteria per 100 ml (Cox and Gordon, 1970). Possibly because of such a ready, albeit polluted, supply of particulate food, the oyster population soared, reaching an estimated 36 million oysters in West Loch in the 1960s. However, this was followed by a massive die-off of 99% of the oyster population in West Loch and a fish and invertebrate kill in Middle Loch in 1972 (Kawamoto and Sakuda, 1973).

Early in the final period from 1970 to the present, the first major surveys of Pearl Harbor water quality, sediment pollution load and biological communities were conducted by the U.S. Navy (Evans et al., 1974; Grovhoug, 1976, 1979) and by the Hawaiian Electric Co. (HECO) (McCain, 1974, 1975) for its thermal outfall in East Loch. The high organic load and polluted conditions that existed at that time were indicated by depressed bottom water oxygen concentrations, especially going toward the heads of West and Middle Lochs where sewage outfalls were still in operation

(Evans, et al., 1974). Extreme lows for bottom water fell to 0.1 ppm, with annual averages to as low as below 1.5 ppm at these sites, compared to surface values or bottom water in the channels that generally remained around 6 ppm.

These studies in the early 1970s preceded the removal of substantial pollution sources from the harbour. In 1975 the Navy instituted shipboard wastewater collection, holding and transfer tank systems to replace release of vessel wastewater effluents into the harbour. Between 1982 and 1984 sewage effluent discharge ended from all major sources (Grovhoug, 1992) except for one that discharged treated sewage into the main channel near the harbour entrance until January 2005, when the point of discharge was moved outside of the harbour entrance to a depth of 45 m. Urbanization of hillsides of the East and Middle Loch watersheds moderated as developments were completed, and better land management practices during construction helped to alleviate surface runoff-related sedimentation. Generally, Pearl Harbor water quality is indicated to have generally improved substantially since its low point in the 1970s. A 1990 study in the East and Southeast Lochs indicated that water quality parameters were within state water quality standards, that there was no substantial difference between surface and bottom water oxygen concentrations oxygen concentrations, and that metal concentrations in sediments were significantly less than 1972 values for most metals, although polychlorinated biphenyl (PCB) concentrations were substantially elevated in the Southeast Loch shipyard area (Grovhoug, 1992).

Two major petroleum hydrocarbon spills have occurred in Pearl Harbor, one of 100,000 gallons of aviation fuel at the head of Middle Loch in 1987 (AECOS, 1987) and one in 1996 of an estimated 39,000 gallons (982 barrels) of bunker fuel oil from the Chevron pipeline supplying the HECO station at the head of East Loch. The 1987 spill produced leaf yellowing, defoliation and some mortality in about 9.5 acres of mangroves (*Rhizophora mangle*) along the Middle Loch shoreline (AECOS, 1987). The 1996 spill resulted in intense oiling of the intertidal flats at the point of discharge near the HECO station intake at the head of East Loch, and deposition of oil and tar in the intertidal zone along the shores of Ford Island and Waipio Peninsula that were in the direct path of the oil spill. Although initial mortality to marine organisms was only four puffer fishes and two prawns, other organisms within the intertidal were directly exposed to oil and tar deposits remained after the initial spill. The long term consequences of this spill on the intertidal and other communities in Pearl Harbor have not been assessed.

An event which triggered substantial interest in species introductions into the harbour was the relocation of the floating dry dock *Machinist* from Subic Bay, Philippines in 1992. In correspondence and public affairs releases the Navy affirmed that the hull had been thoroughly cleaned and inspected before leaving the Philippines and that the dry dock had been deballasted at sea. Further assurances were made that water from ballast tanks had been microscopically inspected for pathogens, and that the hull had been inspected and additional cleaning performed on arrival. However, no initial investigations were made at the time whether marine introductions from the dry dock occurred or whether such introductions might have any measurable impact on the biota of Pearl Harbor or Hawai‘i . After about seven years at anchor at the head of Middle Loch, the dry dock was removed from Pearl Harbor and transported to Guam in 1999 (DeFelice, 1999; Paulay et al., 2002).

Another event related to potential introductions of introduced species to Pearl Harbor was the transport and mooring of the battleship *U.S.S. Missouri* in 1998 to Ford Island in East Loch where it was berthed adjacent to the *U.S.S. Arizona* memorial. The *Missouri* was the site of the signing of the armistice between the United States and Japan in 1945 and had last been used in the 1991 Gulf War. It was decommissioned and moored in Puget Sound in 1993. There was concern that fouling organisms accumulated during the five years at anchor were a potential source of more introduced or invasive species into Pearl Harbor, so the ship was moored enroute for nine days at the mouth of the Columbia River at Astoria Oregon where the salinity to the depth of the ship's bottom was 0.3 to 0.5 (Brock et al., 1989). The effectiveness of this attempt to kill off accumulated fouling was evaluated within hours of its arrival in Pearl Harbor (Brock et al.,1999; DeFelice and Godwin, 1999) and the fouling community of the ship's hull and nearby piers was monitored for the next year (Coles et al., 1999b).

4. MARINE BIOLOGICAL STUDIES IN PEARL HARBOR

There is limited information available on the marine and estuarine life of Pearl Harbor prior to 1900. Pearl oysters and other mollusks were reported from what was to become the modern Pearl Harbor as early as the eighteenth century. Early expeditions collected marine invertebrates, particularly mollusks, from Pearl Harbor, before 1850 (thus, for example, the type locality of the mussel *Brachidontes crebristriatus*, described by Conrad in 1837, is Pearl Harbor [Kay, 1979]). Newspaper accounts exist of the introduction into Pearl Harbor of the Atlantic oyster *Crassostrea virginica* from the 1860s to the 1890s (Kay, 1979). Indicative of the absence of formal surveys is that the first living organisms collected from Pearl Harbor that are in the Bishop Museum collections date only from 1902. Rathbun (1906) described many crabs that had been taken from Pearl Harbor, and the type specimen of the soft sea cucumber *Ophiodesoma spectabilis* was described by Fisher (1907) from a specimen that had been collected from Rainbow Bay at the head of the Harbor's East Loch. After 1910 collecting activity, especially of molluscs, increased in the harbour. Taxonomic descriptions and monographs on Hawaiian marine molluscs and other fouling organisms published through the 1930s included many type specimens collected from Pearl Harbor (Pilsbry, 1917, 1921, 1928; Bartsch, 1921; Miller, 1924; Dall et al., 1938). Pilsbry (1928) also published a report of the first barnacles collected from Pearl Harbor in 1913.

Earlier work on fouling organisms was later expanded on by C. H. Edmondson and W. M. Ingram who sampled fouling panels in Pearl Harbor in 1936. This study was the basis of series of papers on Hawaiian fouling organisms (Edmondson and Ingram, 1939; Edmondson, 1940, 1942, 1944) Other studies in Pearl Harbor during the 1930s produced reports of the first introductions of crustacean crabs into the harbour (Edmondson, 1931) and collections of the first caprellid amphipods (Edmondson and Mansfield,1948) and isopods (Miller, 1941). Access to the harbour was restricted during World War II, but some sampling by C. H.

Edmondson continued. The increased opportunity for species introductions during this period of high shipping activity was reflected in numerous reports of nonindigenous species introductions sampled in the late 1940s and early 1950s (Edmondson 1951, 1952, 1954; Doty, 1961). At least one species thought to be introduced by this means (Doty, 1961), the alga *Acanthophora spicifera*, has became widespread in nearshore areas throughout the main Hawaiian Islands (Smith et al., 2002). The late 1940s and early 1950s period also marked the first sampling and identification of Pearl Harbor sponges (de Laubenfels, 1950).

The only significant sampling activity in the vicinity of Pearl Harbor in the 1960s was done outside of the harbour entrance by the privately owned research vessel *Pele*. However, in the 1970s the most comprehensive sampling that had been done to that time in the harbour was conducted. Long (1969, 1970, 1972) conducted fouling studies inside and outside of the harbour on contract to the U. S. Navy. The Naval Undersea Center (NUC) conducted comprehensive biological studies of the East, Southeast, and Middle Loch areas of the harbour (Evans et al., 1972, 1974; Grovhoug, 1976). Further studies were made of the biological effects of the Navy's three small power stations in Southeast Loch (Grovhoug, 1979), and Hawaiian Electric conducted extensive marine environmental and biological studies in the vicinity of its Waiau Power Station in East Loch (McCain, 1974, 1975, 1977). Despite the considerably degraded conditions in the harbour at that time and the occurrence of the mass mortalities of oyster described above, the Navy survey (Evans et al., 1974) found 394 species or higher taxa, including 90 species of fish from 46 families, living in the harbour environment. Diverse benthic and fish communities were also found by the HECO surveys (McCain, 1974, 1975). The combined results of these studies more than doubled the total number of taxa that had been reported for Pearl Harbor during the previous six decades.

Environmental studies continued in Pearl Harbor during the 1980s and 1990s (Grovhoug and Rastetter, 1980; Grovhoug et al., 1989; Grovhoug, 1992; Seligman et al., 1989; Henderson, 1990, Coles et al., 1997), and at least 15 project related marine environmental studies were conducted by private consultants during this period. Marine monitoring continued in the vicinity of at the Waiau Power Station outfall (Brock, 1994, 1995). These activities somewhat increased the numbers of taxa reported, but not nearly to the extent that occurred in the 1970s.

5. INTRODUCTIONS-RELATED BIOLOGICAL STUDIES IN PEARL HARBOR

Researchers from the Bishop Museum have conducted three marine biological studies in Pearl Harbor focused on introduced marine species. The most comprehensive of these was in 1996 when sites at the 15 locations shown in Figure 2 were surveyed and all biota observed on site or identified from collected samples were recorded (Coles et al., 1997, 1999a). Many stations in the East, Southeast and Middle Lochs that had been previously sampled in the 1970s by the U.S. Navy Undersea Center were re-sampled in the 1996 study to detect specific changes in the biota that had occurred in the approximate 20 years between the surveys. The two other studies by Bishop Museum involved periodic inspections and collection of fouling biota from the hull of the *U.S.S. Missouri* for one year after it was moored at

Ford Island in June 1998 (Coles et al., 1999b), and seven inspection and collections of fouling organisms from the hull of the floating dry dock *U.S.S. Machinist,* moored in West Loch at Station 9 of Figure 2 (DeFelice, 1999).

In addition to the organisms identified from observations and collected at the 15 stations shown in Figure 2, information on species reported from Pearl Harbor along with dates, when available, were obtained from all available sources. These included published scientific papers, monographs and books, unpublished scientific and environmental reports for the U.S. Navy and private organizations, and specimens in the Bishop Museum collections that had been obtained from Pearl Harbor. From these sources and the 1996 survey a total of 1123 taxa were reported from Pearl Harbor since 1899, with 1091 of these reports dated and 844 identified to species level (Coles et al., 1999a). A full list of all taxa reported is available at http://hbs.bishopmuseum.org/pdf/PHReport.pdf. These reports were grouped by decade , and cumulative numbers are shown in Figure 4, along with the numbers of new taxa added by decade. The results indicated that increases in new reports throughout the century were not steady, but showed relative peaks in the 1930s, 1970s and 1990s, during times of markedly higher sampling activity in the harbour.

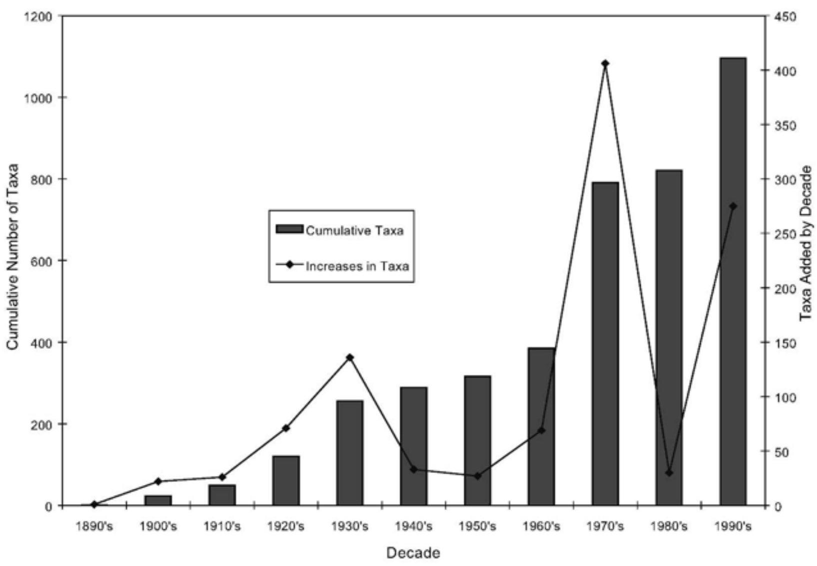

Figure 4. *Cumulative numbers of total taxa (bars) and taxa added by decade (line) for Pearl Harbor.*

The 1123 taxa reported in Pearl Harbor since first observations in 1899 included the 419 taxa observed or collected in the 1996 Bishop Museum surveys (Coles et al., 1999a), or more than 37% of all taxa reported for all sources. Three hundred fifty-five of the 1996 total taxa were identified to species, 173 were newly reported species for Pearl Harbor and 35 were new reports for Hawai‘i. The percentages of total species that were new reports for Pearl Harbor ranged 27-49% throughout the 15 stations, or 37% overall, indicating that, even for an area that has been sampled for nearly 100 years, a comprehensive collection and identification effort can yield a significant number of new reports.

An unexpected finding of the 1996 Pearl Harbor study was the first recorded presence of live reef corals in the harbour (Coles, 1999; Coles et al., 1999a). Four coral species were found, with one species occurring as far into the harbour as Stations 14 and 15 near the shoreline at East Loch (Figure 2). The greatest coral abundance was near Station 2, just inside West Loch channel, where coral colony sizes ranged from to *ca.* 15 cm, and an incipient new reef appeared to be forming. Live corals occurring in Pearl Harbor was quite unthinkable under the water quality conditions that existed 25 years earlier when Grovhoug (in Evans et al., 1974) noted that "stony corals were conspicuously absent from all biostations in Pearl Harbor" including a site near the location of the 1996 Station 2 (Figure 2). All coral colonies found in 1996 were small to medium in size, suggesting the conditions in the harbour had only become amenable to coral settlement and growth in recent years under conditions of improving water quality.

The organisms identified from the 1996 Pearl Harbor survey were evaluated in terms of their status as native, cryptogenic, or introduced. The assignment of individual taxa to a particular status was done on the basis of several criteria (Chapman, 1988, Chapman and Carlton, 1991). If the species is known only from Hawai'i or can be shown to occur in Hawai'i naturally (e.g., fossil evidence of prehistoric distribution including Hawai'i) it was categorized as *native*. If the species is known to occur elsewhere, a decision is made regarding its presence in Hawai'i. If no evidence exists regarding the natural occurrence of a species in Hawai'i, there is a possibility the species may be introduced. This evidence includes known introduction or appearance in local regions where not previously found, association with human mechanisms of dispersal (e.g., as fouling on ship bottoms), association with other introduced species, restriction to artificial or disturbed habitats (e.g., harbours), and widespread, disjunct geographic distributions. Species with these and other associated attributes were categorized as *nonindigenous* or *introduced*. Species that are not demonstrably introduced or native were considered to be *cryptogenic* (Carlton, 1996). Pending further study and additional evidence, a species may be moved from one status category to another.

Of the 419 taxa identified throughout Pearl Harbor in the 1996 survey 95 species, or 23% of the total, were considered introduced or cryptogenic. These included one introduced macroalga, 90 introduced or cryptogenic invertebrates and four introduced fishes (Coles et al., 1999a). The introduced or cryptogenic invertebrates were dominated by crustaceans (26 species, mostly amphipods) sponges and molluscs (17 species each), polychaetes and ascidians (10 species each), and bryozoans (eight species). The numbers of introduced and cryptogenic species reported at each station are shown in Figure 5 (bars) along with the percentage of the total taxa that were introduced or cryptogenic (line). Comparison of these results with the station locations (Figure 2) indicates that introduced and cryptogenic species as a proportion of total taxa generally decreased with proximity to the Pearl Harbor channel entrance. Lowest values of 19-28% occurred at Stations

1, 2 and 6 nearest the entrance, and highest values of 46-51% occurred at Stations 4 and 5 at the head of West Loch, Station 9 at the head of Middle Loch, and Station 13 at the northwest end of Ford Island.

The 95 introduced and cryptogenic species that were present in the 1996 Pearl Harbor surveys were grouped by the decade of their first appearance to determine if there was a pattern of numbers of introductions with time. The initial results suggested three peak decades for new introductions in the 1930s, 1970s and 1990s. However, further analysis indicated that these peak periods were effort-related, similar to the pattern shown in Figure 4. When the ratio of newly introduced or cryptogenic species to newly observed total taxa were plotted by decade (Figure 6) two peak decades were indicated, the highest corresponding to the 1940s during the time of World War II, and the second peak during with the 1910s, coinciding with the enlargement of the entrance channel in 1911, initial development of the Navy base, and time of World War I. Further analysis indicated that introductions, once

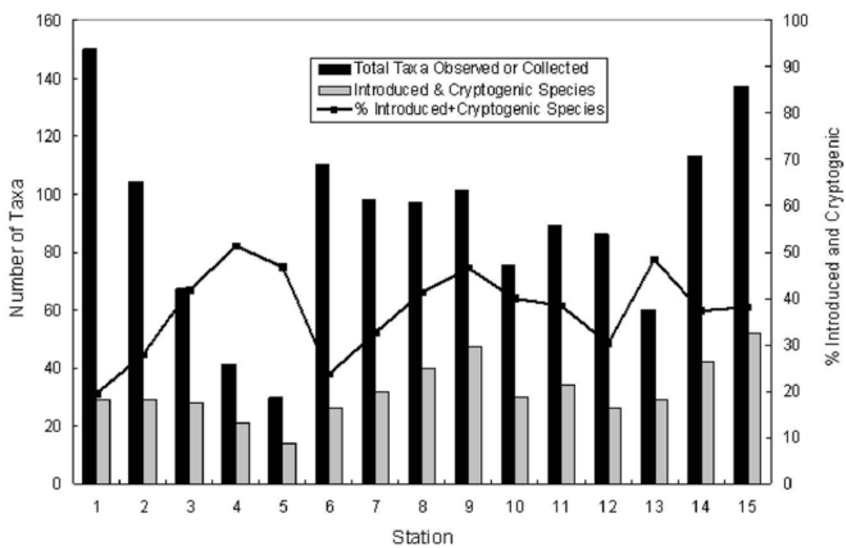

Figure 5. Numbers of total taxa, introduced and cryptogenic species, and % of total taxa that were introduced or cryptogenic species at 15 stations surveyed in Pearl Harbor in 1996.

reaching the harbour, were persistent. Of the 101 introduced species that had been reported in the harbour since 1899, 69 species, or 68%, were found in 1996. The possible origin of the 95 introduced or cryptogenic species that occurred in 1996 was estimated from information on distributions and type localities. Most of the species (36%) were of indeterminate origin, but for the remainder 32% were designated to be from the Indo-West Pacific, and 5% from the Eastern Pacific, while 24% were considered to originate from the Atlantic Ocean, including 11% from the Caribbean (Coles et al., 1999a).

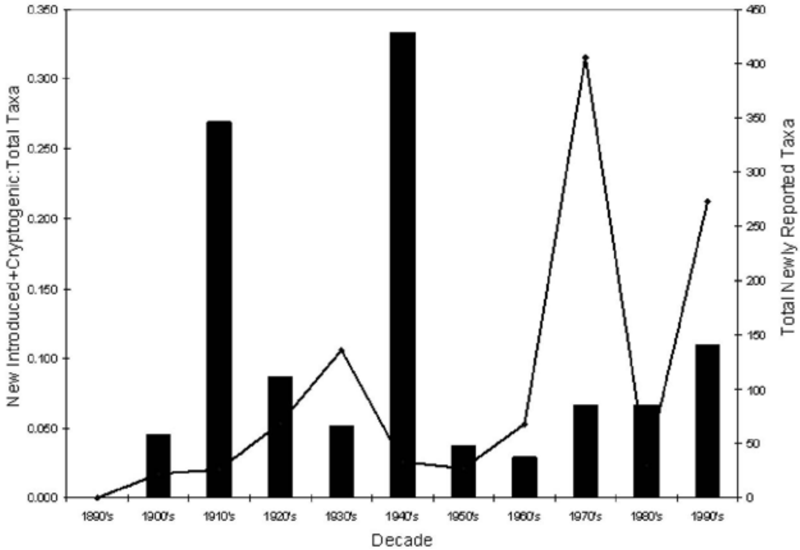

Figure 6. *Ratio of new reports by decade of introduced and cryptogenic species to total newly reported taxa (bars) compared to newly reported total taxa (line) in Pearl Harbor from 1890 to 1996.*

reaching the harbour, were persistent. Of the 101 introduced species that had been reported in the harbour since 1899, 69 species, or 68%, were found in 1996. The possible origin of the 95 introduced or cryptogenic species that occurred in 1996 was estimated from information on distributions and type localities. Most of the species (36%) were of indeterminate origin, but for the remainder 32% were designated to be from the Indo-West Pacific, and 5% from the Eastern Pacific, while 24% were considered to originate from the Atlantic Ocean, including 11% from the Caribbean (Coles et al., 1999a).

The distribution and timing of some of the introduced species identified in the 1996 study suggested that the 252 m long floating dry dock *U.S.S. Machinist* that had been brought to Pearl Harbor in May 1992 was a possible source of recent introductions. The dry dock, which was the site of the 1996 study's Station 9 (Figure 2) showed one of the highest percentages of introductions of any site (Figure 5). The heavily fouled hull of the dry dock was therefore further sampled six times in 1998-1999 before it was towed to Guam.

A total of 113 marine invertebrates were identified from these six samplings (DeFelice, 1999), 95 of them to species, which included 20 new records for Hawai‘i. Of the 95 species 75 were previously reported in the Hawaiian Islands and 49 (52%) were considered to be native species. The remaining 46 species were considered introduced or cryptogenic, for a total of about 48% of the total identified species, or 43% of the total identified taxa. This was nearly double the value determined for the 15 Pearl Harbor sites sampled in 1996. In terms of origin, 32% were indeterminate, 40% considered from the Indo-West Pacific, 3% from the East Pacific, and 15% from the Atlantic or Caribbean.

No attempt was made to clean the hull of the *Machinist* before it was towed from Pearl Harbor to Apra Harbor, Guam, where it was surveyed within a week of its arrival in July 1999 (Paulay et al., 2002). Over half of the 42 species that were identified from this initial survey were new records for Guam, i.e. introduced, and many were still alive and abundant on the hull. However, a re-survey one-year later indicated that many bivalve molluscs did not survive, probably because the six weeks in transit in oceanic waters between Hawai‘i and Guam and the generally lower organic content of Apra Harbor compared to Pearl Harbor exhausted the energy reserves of the molluscs. By contrast, most of the sponge species on the dry dock hull appeared to still be present in August 2000.

An opportunity to evaluate the effectiveness of attempting to control introductions from a highly fouled vessel occurred when the *U.S.S. Missouri* was brought to Pearl Harbor from Pudget Sound, Washington in June 1998 Freshwater exposure at the mouth of the Columbia River and the temperature difference between the Pacific Northwest and Hawai‘i resulted in a massive die-off in organisms on the hull during the approximate two-week transit to Pearl Harbor. The hull was surveyed independently by both Bishop Museum (Coles et al. 1999b; DeFelice and Godwin, 1999) and University of Hawai‘i (Brock et al., 1999) researchers within two hours of the ship's arrival in Pearl Harbor on June 22, 1998. Both groups found that most organisms still adhering to the hull were dead and/or decayed, but five species were found alive by DeFelice and Godwin (1999) and 11 were listed as alive by Brock et al. (1999). Most significant, a mussel later identified as *Mytilus galloprovincialis* was observed spawning, both in the field and later in specimens brought to the laboratory (DeFelice and Godwin 1999).

Observations 83 days after the arrival of the ship indicated none of the organisms that came on the ship's hull to have survived in Pearl Harbor (Brock et al., 1999). Monitoring of the fouling community on the ship's hull and nearby piers which began about four months after the ship’s arrival and continued through June 1999 (Coles et al., 1999b) also indicated no establishment or growth of any organisms that arrived with the ship. Rather, a fouling community remarkably similar to that already established on the *U.S.S. Machinist* dry dock quickly developed on the *Missouri* and continued through the eight month monitoring period, with 48% of the 91 identified species composed of introduced or cryptogenic species already known to occur in Pearl Harbor. Initial conclusions were therefore that no introductions had occurred from the *Missouri*. However, approximately three months after its arrival, small mussels verified by DNA analysis to be *Mytilus galloprovincialis* were collected from a ballast tank of a submarine in Southeast Loch, approximately 1 km from the *U.S.S. Missouri* (Apte et al., 2000). Based on observations of spawning from this species on the ship's arrival in Pearl Harbor and the size and estimated age of the submarine's juvenile mussels, it was concluded that they could have been introduced by the ship, despite the precautions taken to eliminate fouling and the stresses encountered by the parent population. This event may have provided an opportunity for successful recruitment of a new species into Pearl Harbor, but there have been no subsequent observations in the harbour to confirm whether any population has been established.

6. DISCUSSION AND CONCLUSIONS

Pearl Harbor has received both intentional and accidental species introductions since at least 1866 when the first attempts at culturing eastern oysters (*Crassostrea* sp.) in the harbour were made. Most introductions have probably occurred accidentally, and most introduced species are assumed to have been from vessel fouling in the last 100 years. Earlier undocumented introductions could have originated from the hulls of the first European ships to reach the islands or possibly from earlier Polynesian migrations to Hawai‘i. Some species may also have been released along with intentionally introduced species (e.g. oysters).

Some of the more vigorous vectors promoting species introductions in other world ports may operate at a relatively lower scale in Pearl Harbor. Ballast water is transported in large bulk cargo carriers that may arrive in commercial harbours with 25,000 or more metric tons of ballast water that is discharged when they pick up cargo. By contrast, virtually all vessel movement in and out of Pearl Harbor is military traffic, which generally carries far less ballast water than commercial transport vessels. Vessel fouling, especially of slow moving barges and dry docks, e.g. the *Machinist*, therefore has likely exceeded ballast water, as a primary vector for movement of introduced species in and out of the harbour. Vessel fouling also has been indicated as a major source of introduced species arrival and dispersal for Hawaiian commercial harbours (Godwin, 2003). However, there have been no actual studies on the amounts of ballast water released in Pearl Harbor, nor of its relative importance of ballast water compared to vessel fouling as vectors for marine species introduction.

Although few data are available to verify the general improvement of water quality in the harbour since comprehensive surveys were made in the early 1970s, the occurrence of reef coral well within the harbour, personal observations of improved water clarity, and the abatement of many sources of pollution to the harbour since the 1970s (Grovhoug, 1992) indicated that water conditions had improved considerably by the late 1990s. No recent observations or measurements are available to determine whether conditions have continued to improve or whether further establishment of introduced species has continued.

The 1996 Pearl Harbor survey was the first of many studies conducted in the last decade in Hawai‘i and elsewhere in the tropical Pacific to determine the presence and impact of introduced species in harbours, embayments, and open coastlines. Results from the available information are summarized in Table 1, which shows remarkable consistency among the results from harbours and semi-enclosed embayments throughout the main Hawaiian Islands. Surveys in five commercial and small boat harbours of Honolulu, Kāne‘ohe Bay, and eight harbours on the islands of Kaua‘i, Moloka‘i, Maui and Hawai‘i each found around 100 introduced and cryptogenic species, similar to what was found for Pearl Harbor. However, the greater total species richness determined in these studies resulted in somewhat lower values of 14-17% of the total taxa being composed of introduced or cryptogenic species.

Table 1. *Numbers of nonindigenous (N) and cryptogenic (C) species and total taxa determined from the Hawaiian Islands, Johnston Atoll, Guam and American Samoa.*

Location	N	C	Total N+C	Total Taxa	% N+C	Source
Oahu, Hawai'i						
Pearl Harbor	69	26	95	419	23.0	Coles, et al., 1997, 1999a
Honolulu Commercial and Small Boat Harbors	73	27	100	585	17.0	Coles, et al., 1999c
Kāne'ohe Bay	82	34	116	617	14.5	Coles, et al., 2002
Hawaiian Neighbor Island Harbors	72	32	104	694	14.9	Coles, et al., 2004
Guam						
Apra Harbor	27	29	46	682	6.7	Paulay et al., 2002
American. Samoa						
Pago Pago Harbor,	17	11	28	977	2.9	Coles, et al., 2003
North Queensland (Australia)						
Hay Point Port	9	3	12	506	2.4	Hewitt et al,. 1998
Mourilyan Harbour	2	2	4	401	1.0	Hoedt et al., 2000
Abbot Point Port	0	5	5	593	0.8	Hoedt et al., 2001
Lucinda Port	2	9	11	480	2.3	Hoedt et al., 2001

By contrast, the few studies completed elsewhere in the tropical Pacific indicate fewer introduced species to occur in these harbours that have been studied, and even lower values for their percent composition of the generally more diverse biota resident in these harbours. Paulay et al. (2002) found only 46 introduced or cryptogenic species out a total of 682 taxa for a percentage composition of 6.7% in Apra Harbor Guam, and Coles et al. (2003) found 28 introduced or cryptogenic species to comprise only 2.9% of the total taxa at six stations in Pago Pago Harbor, American Samoa. Even lower values occurred at four ports and harbours surveyed in North Queensland, Australia, where only 5 to 11 introduced or cryptogenic species were identified for total taxa ranging 401-593 and percentage compositions ranged only 0.8-2.4%. These low values were especially surprising, because the North Queensland ports service large bulk carriers transporting large quantities of raw materials from Australia and therefore are likely to have been exposed to massive volumes of ballast water discharge in the process of taking on cargo.

These comparisons indicate that marine introductions in Hawaiian harbours range well above values for most tropical Pacific ports and harbours for which information is available. The 69 nonindigenous species recorded in Pearl Harbor rank more closely with the generally higher numbers reported for harbours in the temperate Pacific such as San Francisco Bay (234, Cohen and Carlton, 1998) Waitemata Harbour, New Zealand (39, Hayward, 1997), four temperate ports in New South Wales and Western Australia (48, Hewitt, 2002), and Port Phillip Bay, Victoria, Australia (101, Hewitt and Campbell, 1999). The reasons for this are undoubtedly varied and complex, and possible explanations may include Hawaii's central location as a crossroad of ship traffic from throughout the Pacific (Carlton, 1987), Hawaii's temperature environment being intermediate between temperate and truly tropical and therefore allowing introductions from higher latitudes, and the more diverse biota of tropical areas offering fewer opportunities for introduced species to compete in their new habitats on arrival (Coles and Eldredge, 2002; Hutchings et al., 2002).

This review of biological studies and surveys of introduced marine species conducted in Pearl Harbor leads to the following preliminary considerations related to the potential for continued spreading and proliferation of introduced species throughout Asia-Pacific region. 1) The available evidence indicates that introduced species have occurred in the harbour for at least the last century and probably well before that. 2) Timing of introductions into the harbour appears to correlate with periods of high shipping and construction activity, i.e. during wartime in the case of Pearl Harbor. 3) For introduced species whose origin could be estimated, the Asia-Pacific region has been the greatest source. 4) When no precautions are taken to eliminate potential introductions, such as in the transport of the *Machinist* to Guam, many but not all potential introductions are likely to die enroute or on arrival because of lack of food resources. 5) Even when substantial effort is made to eliminate introductions, such as the transport of the *Missouri* from Pudget Sound, there is still a small probability that a reproducing population of new species can be established. 6) The model provided by Pearl Harbor for marine introductions may be of limited applicability to the Asia-Pacific region, since the Pearl Harbor is virtually restricted to military ships that do not on-load large volumes of ballast water, which is considered a primary vector for the transport and spread of introduced marine species for commercial shipping.

In contrast to the baseline of knowledge for introduced marine species that has been established for ports and harbours in Hawai'i, American Samoa, Guam and Australia, there is little to no information on this subject available for most of the Asia-Pacific region except for Hong Kong Harbour (Morton, 1989). Shipping activities from Asian ports in Japan, Korea, Taiwan, Singapore and especially China (Morton, 1989) have escalated dramatically in the past few decades as the economies of these countries have grown and focused on manufacturing goods which are sent all over the world. During the same period, marine introductions have surged in harbours, ports and other coastal ecosystems which have been studied elsewhere in the world (Carlton and Geller, 1993; Ruiz et al., 1997). Given the potential serious impacts, both environmentally and economically, of continued transport of introduced species on hulls or in ballast water of ships from Asian ports and harbours, lack of knowledge of the composition of the marine communities and potential invasive species from those areas is a major deficiency. Efforts should be directed toward developing sampling and analysis that would increase the information available for marine communities and introduced species in Asian ports to be comparable to that summarized in Table 1. This baseline of knowledge would be an essential step in developing and refining early warning systems for intercepting and controlling the further spread of introduced species throughout the Indo-Pacific and into other ocean regions.

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CHAPTER 15

PHYSICAL ENVIRONMENT IN THE GULF OF THAILAND WITH EMPHASIS ON THREE IMPORTANT PORTS

SUPHAT VONGVISESSOMJAI

1. INTRODUCTION

The Upper Gulf of Thailand (Figure 1) has a surface area of approximately 100 x 100 km and an average depth of about 15 m. There are four major rivers draining into the Upper Gulf, namely, the Maeklong River, the Thachin River, the Chao Phraya River, and the Bangpakong River. The features of bottom topography of the Upper Gulf can be described as follows: from the shallow northern coast, the bottom slopes gradually downward to a mean depth of 25 m at its mouth between Sattahip and Hua Hin; the eastern part, with rocky off-shore islands, is slightly deeper than the western portion. The average depth of the whole gulf over its total area of about 320,000 km^2 is 45 m. Maximum depths in the central parts are in the range of 70 to 80 m.

2.1. Tracks and Strength of Cyclone (Typhoons)

Cyclone disasters in Thailand (Figure 1) can be grouped into two, namely firstly disasters due to cyclonic winds, and secondly those due to river floods from cyclonic rainfall. This paper presents cyclone disasters due to strong winds and surges along shoreline. Detailed calculations of storm surge and wave are made at Ao Phai located in the Upper Gulf of Thailand which result in quite small values of extreme surges of 1.17 m and 0.70 m for the Probable Maximum (PMC) and 250 year cyclone respectively while their corresponding significant waves are 2.3 m, 5.9 s and 1.9 m, 5.1 s. However, when typhoons Harriet and Gay attacked southern shorelines which are open sea, the resulting surges and waves were much larger which caused much more damages and casualties.

2.2. Cyclone dynamics

Historically, disasters from cyclones have occurred frequently (see Table 1), e.g. in 1952 (typhoon Vae) and in 1962 (typhoon Harriet), in 1970 (typhoon Ruth). More

E. Wolanski (ed.), The Environment in Asia Pacific Harbours, 229–247.

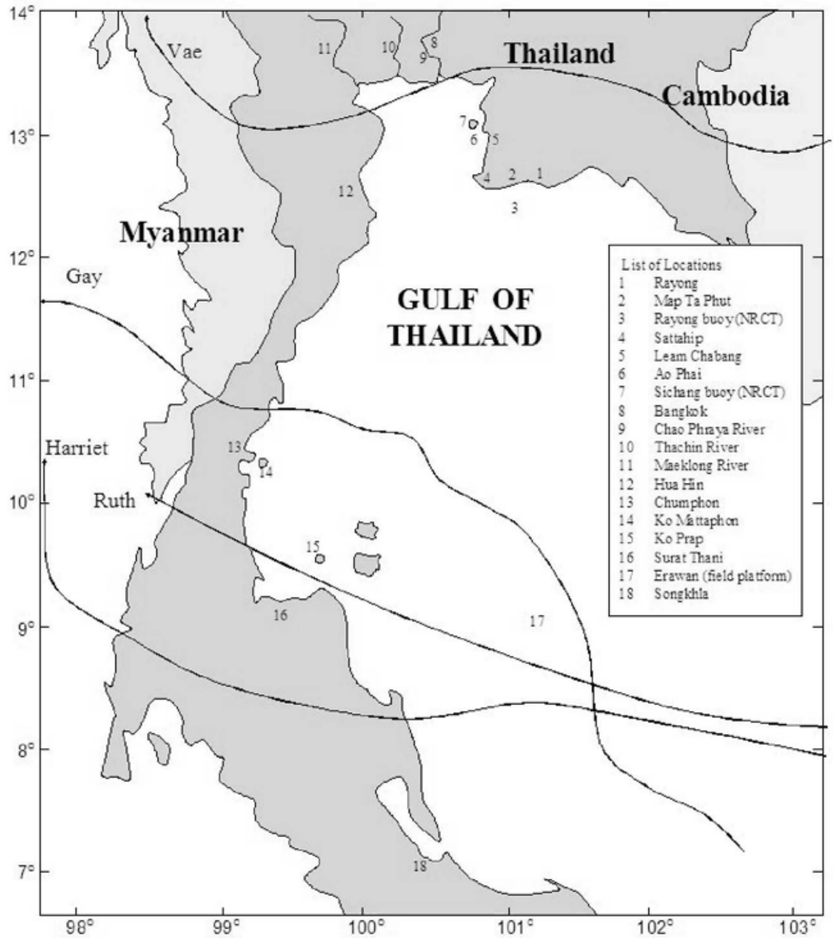

Figure 1. *Map of the Gulf of Thailand.*

recently, in 1989, typhoon Gay left in its path 580 dead, 620 boats sunk, 40,000 houses destroyed and caused about 11 billion Baht damage. The characteristics of these cyclones are described.

A tropical cyclone is characterized by four parameters, namely, its forward speed V_F, radius of the eye R, the minimum pressure at its center p_o and the maximum wind speed at the radius of the eye $U_{\max}$. The pressure p increases as a function of the radial distance r from its center from p_o to the atmospheric pressure p,

$$p - p_o = (p_n - p_o)\exp(-R/r), \text{ for } 0 < r < \infty \tag{1}$$

The radial wind V also varies as a function of r (Jelenianski , 1965),

$$V = V_o(r/R), \text{ for } r \leq R \tag{2}$$

$$V = V_o(R/r), \text{ for } r > R \tag{3}$$

where V_o is the maximum sustained wind in a cyclone.

Tracks of 4 disastrous cyclones over the South China Sea affecting Thailand are shown in Figure 1. The characteristics of the first three cyclones affecting Thingland are shown in Figure 2.

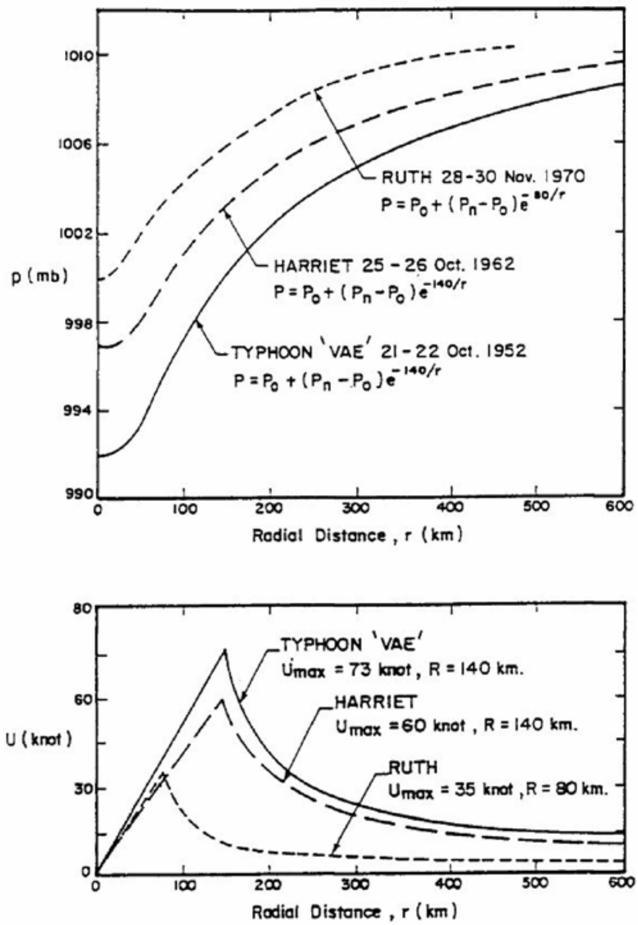

Figure 2. *Radial distribution of pressure and wind velocity of typhoons Vae, Harriet and Ruth.*

These cyclones can be seen clearly from weather satellite in form of huge clouds. The observed surges and tides at Ko Prap and Ko Mattaphon are shown in Figure 3.

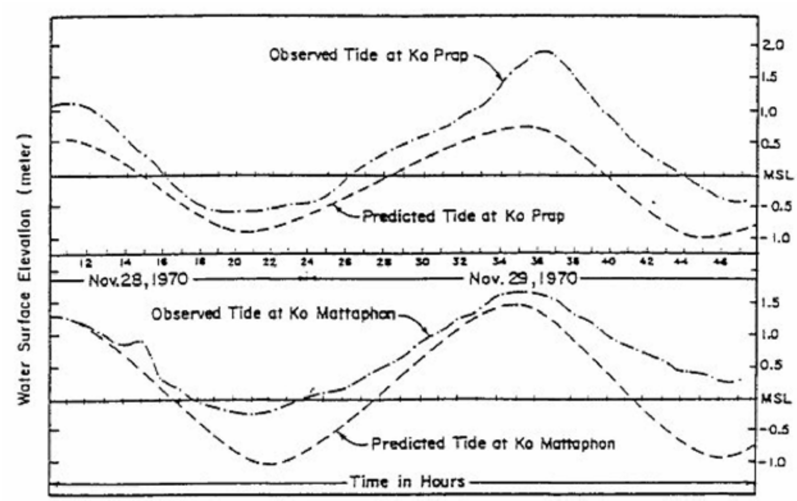

Figure 3. *Comparison of observed and predicted water surface elevation to show the surge height.*

The stress of surface winds produces a storm surge and waves. For a slowly moving cyclone, the following formulas given by CERC (1984) can be used to obtain a good estimate of the deep-water significant wave height and period:

$$H_o = 5.03 \exp\left[\frac{R\Delta p}{4700}\right]\left[1 + \frac{0.29\alpha V_F}{\sqrt{U_R}}\right] \tag{4}$$

and

$$T_S = 8.6 \exp\left[\frac{R\Delta p}{9400}\right]\left[1 + \frac{0.145\alpha V_F}{\sqrt{U_R}}\right] \tag{5}$$

where H_o = deepwater significant wave height (m), T_s = the corresponding significant wave period (s), R = radius of maximum wind (km), $\Delta p = p_n - p_o$, where p_n is the normal pressure (760 mm of mercury), and p_o is the central pressure (mm of mercury), V_F = the forward speed of the cyclone (knots), U_R = the maximum sustained wind speed (kn), calculated for 10 meters above the mean sea surface at radius (R) where

$$U_R = 0.865\, U_{\max}\text{ , (for stationary cyclone)} \tag{6}$$

$$U_R = 0.865\, U_{\max} + 0.5\, V_F\text{ , (for moving cyclone)} \tag{7}$$

$U_{\max}$ = maximum gradient wind speed (in kn)10 m above the water surface,

$$U_{\max} = 0.447\left[14.5(p_n - p_o)^{1/2} - R(0.31f)\right] \tag{8}$$

f = Coriolis parameter $(2w\ \sin\phi)$ in radians per hour where

Latitude (ϕ)	25°	30°	35°	40°
f	0.221	0.262	0.300	0.337

α = is a coefficient depending on V_F ,For slowly moving cyclone α = 1.0.

Once H_o is determined from Equation (4), it is possible to obtain the approximate deepwater significant wave height H_o by use of Figure 4 for other areas of the cyclone.

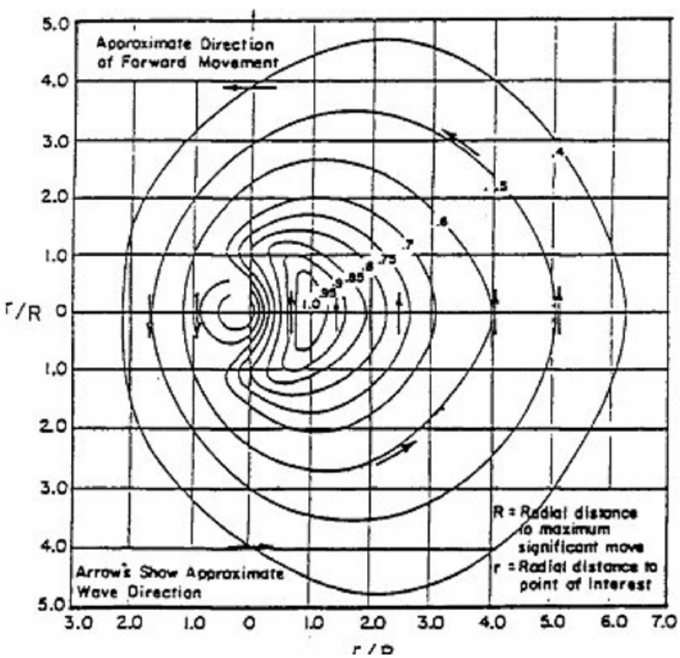

Figure 4. *Isolines of relative significant wave height for a slowly moving typhoon.*

It results

$$T_s = 12.1\sqrt{\frac{H_o}{g}} \qquad (9)$$

It is these large waves and that cause casualties and damages to the coast.

2.3. Cyclonic Surge and Waves at Ao Phai (see the location in Figure 1)

Planning and design of a power plant at Ao Phai require knowledge on probable maximum and minimum sea levels as well as surge and wave generated by cyclonic from the South China Sea.

2.3.1. Cyclonic Surge Analysis

The Probable Maximum Cyclone and the 250 year cyclone were determined by Vongvisessomjai et al. (1977), assuming that the Central Pressure Index (CPI) or the minimum central pressure (p_o) values of storms belong to a normally distribution population. This permits the extrapolation of the CPI to theoretically high recurrence interval so as to get inherent parameters adopted for PMC and the 250 year cyclone (see Table 1 and Figure 5).

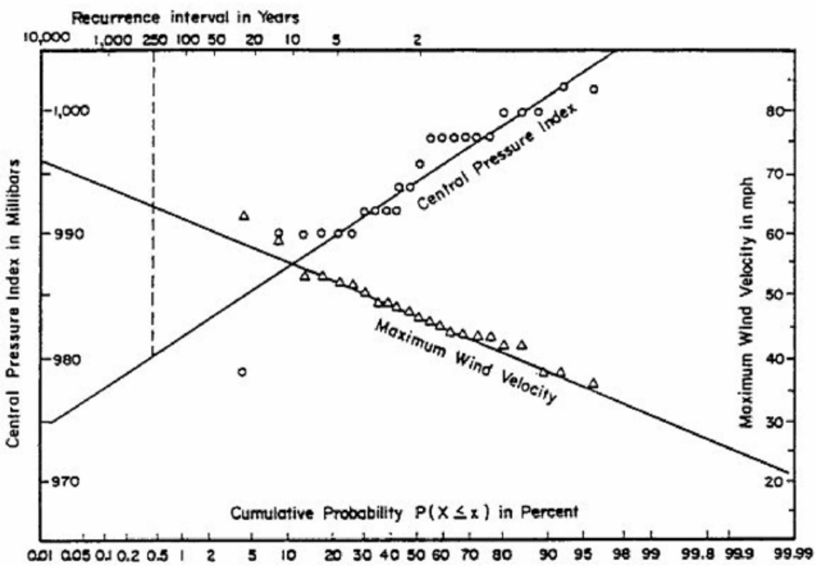

Figure 5. *Recurrence intervals of cyclone characteristics for the Gulf of Thailand.*

Table 1. *Historical typhoon (cyclone) characteristics.*

#	Year, Month, Date			Name	p_o (mb)	U_{max} (mph)	V_F (mph)	R (nmiles)
1	1952	Oct	21-22	VAE	992	84	14.97	76
2		Oct	24-25	TRIX	998	44	17.27	60
3	1960	Oct	03-04	-	992	52	3.45	8.4
4	1962	Oct	25-26	HARRIET	997	69	18.45	76
5	1966	Jun	17-18	-	-	-	12.67	-
6		Oct	25-26	-	990	48	9.21	49
7	1967	Jun	17-18	-	978	62	12.67	140
8		Oct	05-06	-	996	49	11.52	16
9		Oct	09-10	-	998	44	17.27	90
10		Nov	10-11	-	-	-	12.67	-
11	1968	Sep	05-06	BESS	992	44	2.30	115
12		Oct	21-22	HESTER	998	46	11.52	10
13	1967	Jun	24-25	-	998	46	6.91	7.5
14		Sep	20-21	-	992	51	12.67	57
15		Nov	02-03	-	1000	45	16.12	10
16	1970	Sep	20-21	-	994	52	13.82	20
17		Oct	25-26	KATE	1000	27	13.82	314
18		Nov	29-30	RUTH	1000	42	11.52	43
19	1972	Jun	03-04	NAMIE	990	48	6.91	123
20		Sep	06-07	-	990	49	4.61	16
21		Sep	18-19	-	-	-	-	-
22		Dec	04-05	SALLY	994	50	5.76	5
23	1973	Nov	12-13	-	1002	38	5.76	4
24		Nov	17-18	THELMA	998	45	5.76	39
25	1974	Oct	09-10	-	1002	34	6.91	15
26		Nov	05-06	-	998	45	13.82	60

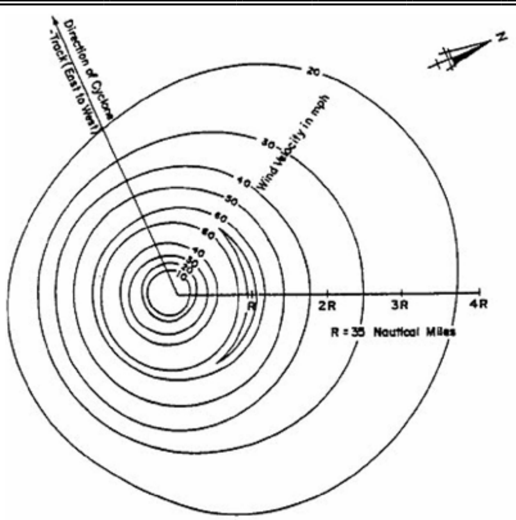

Figure 6. *Wind field of the probable maximum cyclone for the Gulf of Thailand.*

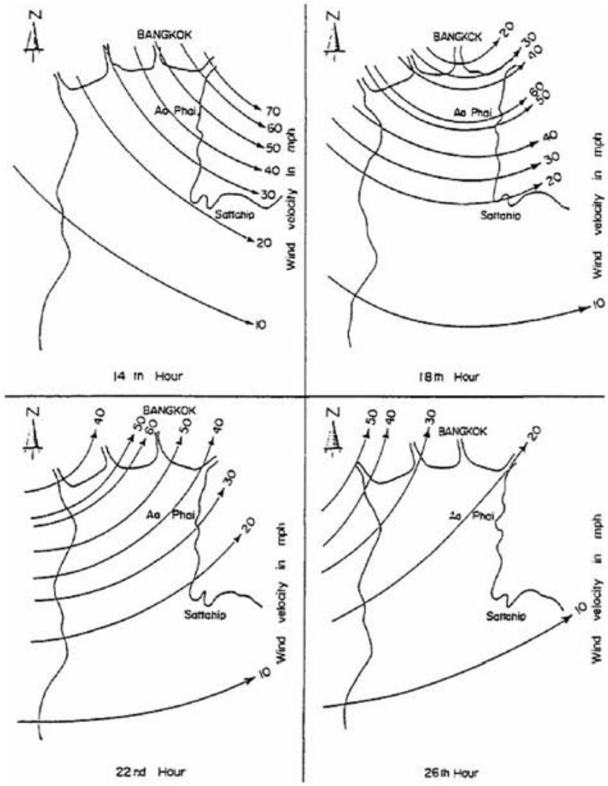

Figure 7. *Probable Maximum Cyclone (PMC) wind field over the Gulf of Thailand.*

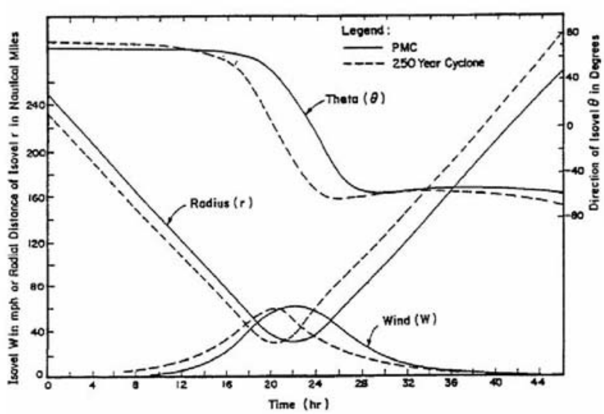

Figure 8. *Probable Maximum Cyclone (PMC) and 250-year cyclone characteristics as would be observed at Ao Phai.*

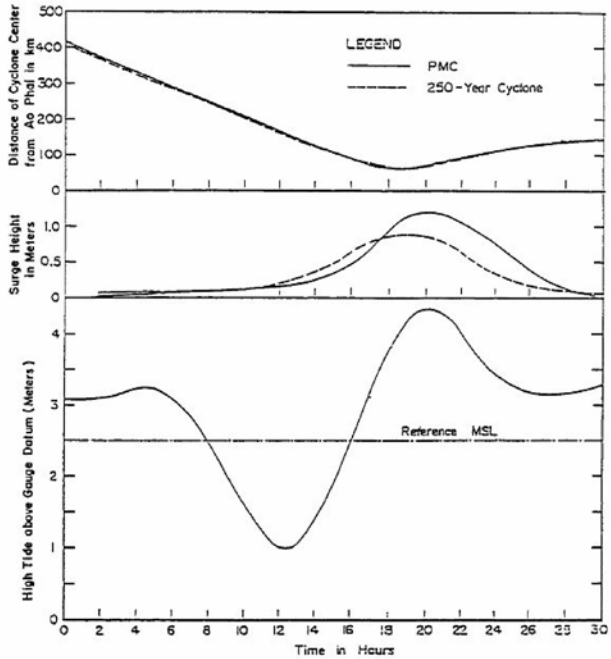

Figure 9. *Probable Maximum Cyclonic Surge (PMC and 250-year) and high tide at Ao Phai.*

2.3.2. Cyclonic Waves

The observed seasonal waves are found to be insignificant in height and period. However, short-duration waves generated by cyclones, known as "cyclonic waves" are important aspects of cyclones occurring in the nearshore zone and in the immediate neighborhood of coastal front structures. They can be prominently superposed on the surge levels. Quantitative estimates of the waves associated with cyclones are forecasted and included here.

A procedure for present computation of cyclonic waves is developed (Day, 1977). Two design cyclones, the PMC and the 250-year cyclone as used for the surge computation, are adopted. The results are shown in Figure 10. In this figure, the starting time of computation is taken to be that when the iso-velocity line of zero wind velocity touches the shoreline at the west side of the Gulf. The distance of the cyclone center from Ao Phai shown in this same figure indicated that the cyclone center is nearest to Ao PhaI at about the 18th hour. At this time, the cyclonic waves would be the most severe; this would be particularly so if the track of the cyclone is about 35 nautical miles north of Ao Phai. The severest cyclonic wave characteristics were found to be 2.3 m in height and 5.9 s in period at PMC condition. For the condition of the 250-year cyclone, the severest cyclonic wave characteristics were found to be 1.9 m and 5.1 s. No cyclonic waves have been actually recorded to permit comparisons. However, this cyclonic wave model has been successfully applied to tropical cyclones in Hong Kong and Taiwan (Day, 1977), and the computed order of magnitudes obtained above are within this expectation.

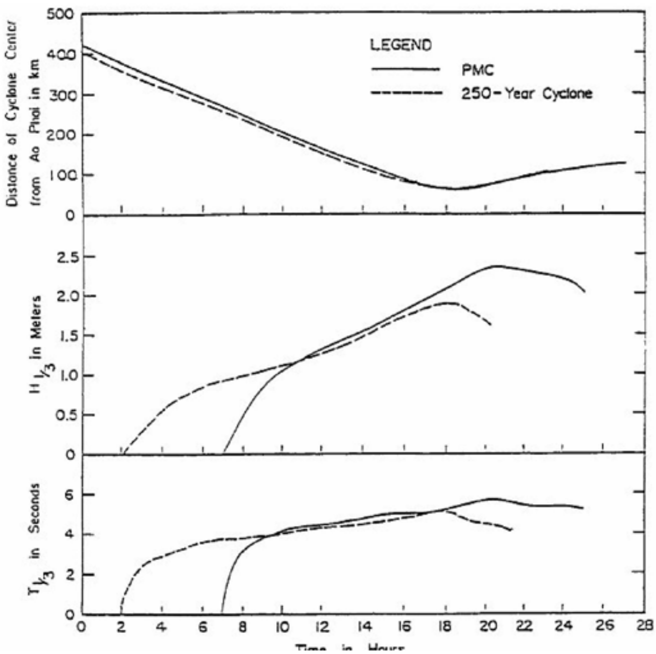

Figure 10. *Probable maximum cyclonic wave (PMC and 250-year) at Ao Phai.*

2.4. Conclusions

Severe cyclones (typhoons) rarely reach the Gulf. Cyclones result in quite small values of extreme surges of 1.17 m and 0.90 m for PMC and 250-year cyclones respectively. The corresponding significant waves are 2.3 m, 5.9 s and 1.9 m, 5.1 s at Ao Phai. When typhoons Harriet and Gay attacked southern shorelines which are open sea, the resulting surges and waves were much larger which caused much more damages and casualties.

These disasters can be alleviated from known characteristics of cyclones and through proper warning.

3. WINDS AND WAVES

Figure 11 shows monthly wind roses of Rayong, which divides the Gulf of Thailand into the Upper Gulf and the Lower Gulf. The prevailing winds can be summarized as follows:

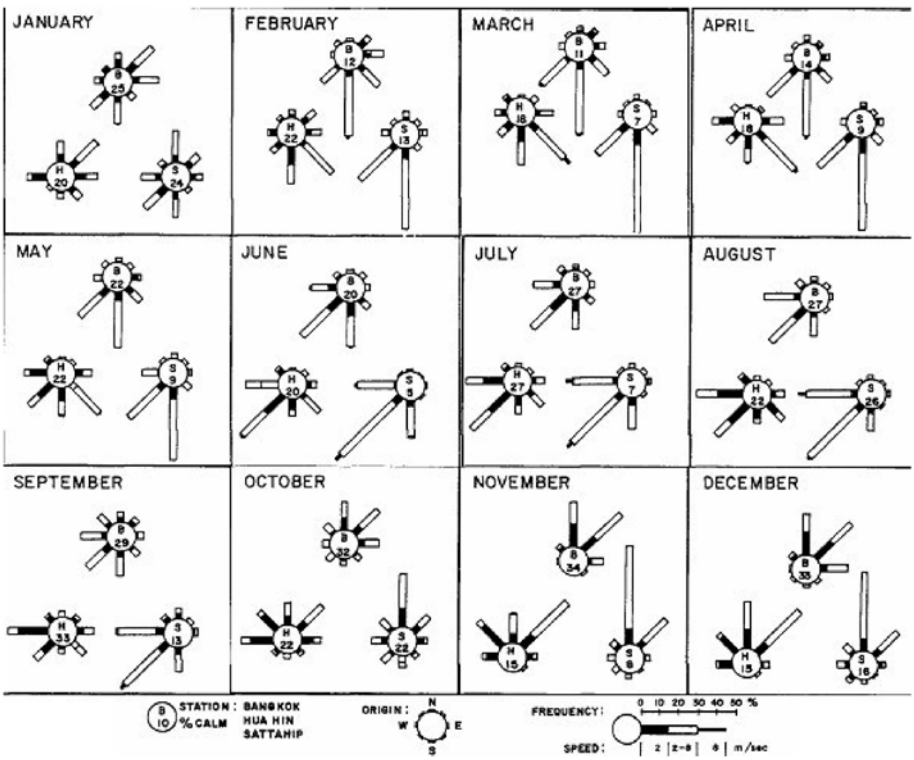

Figure 11. *Wind rose of Sattahip, Hua Hin, and Bangkok (NEDECO, 1963).*

3.1. Upper Gulf of Thailand

During the northeast monsoon, the prevailing winds are from the north and east during the early season, becoming quite variable and southerly. Wind speeds are between 4-8 knots. Winds in the spring transition are quite variable with about the same wind speeds as those in the late northeast monsoon season. In the southwest monsoon season, the prevailing winds are more persistent from the south and west.

3.2. Lower Gulf of Thailand

The winds are generally in the east during the northeast monsoon season, with stronger winds during the end of the season. During summer, wind directions are quite similar to those in winter season, but speeds decrease. In the southwest monsoon season, along the northern portion of this region, the winds become stronger while those of the areas from Songkhla southward are comparatively weaker. In the fall transition, winds are weak and variable. These wind speeds increase as the northeast monsoon progresses. Surat Thani experiences weak winds (2-3 knots) throughout the year while Songkhla has stronger winds, mainly of 11-12 knots during the late northeast monsoon season.

Waves are generated by these winds. Basically the monsoon winds generate seasonal waves which control the long-term sedimentation, and the tropical cyclones generate storm waves, which are used as design waves for coastal and harbour structures. Outside of cyclones, the waves are generally moderate in all areas, and even the most exposed gulf areas have, over half the time, no waves over 0.50 m (Silvester and Vongvisessomjai, 1970).

NEDECO (1963) measured waves at Bangkok Bar from April to October 1962. From a correlation of these waves with wind data at Bangkok, hindsight wave characteristics were shown for the period 1956 to 1960 (Table 2).

Table 2. *Wave characteristics at Bangkok bar. H is wave height and f is the frequency.*

Month	H (m)	f (%)	H×f (m)	Month	H (m)	f (%)	H×f (m)
January	0.30	49	0.15	July	0.30	64	0.20
February	0.55	79	0.45	August	0.40	63	0.25
March	0.50	82	0.40	September	0.40	53	0.25
April	0.55	75	0.40	October	0.30	29	0.10
May	0.45	71	0.30	November	0.35	12	0.05
June	0.35	73	0.25	December	0.30	15	0.05

Similar studies at Sattapith Port predicted a design wave of 2.93 meters with a period of 8.5 seconds (Lyon Associates, King, and Gavaris, 1966).

NEDECO (1972), in the study of the deep-sea port of Laem Chabang estimated the wave exceedance and return period (Table 3).

Table 3. *Design Wave of Leam Chabang Port.*

Wave Height (m)	Percentage Exceeding (%)	Return Period (year)
2.1	0.01	5
2.3	0.001	50
2.5	0.0001	500

JICA (1983) estimated similar parameters for Map Ta Phut on the eastern seaboard (Table 4).

Recent advances in marine science and technology have been made by UNOCAL Thailand in recording 15 months of wind and wave data at the Erawan field platform. The 100 year return period waves was estimated to be 6.4 m, the significant wave height was less than 1 m 90 percent of the time.

The above wave characteristics in terms of wave height and wave angle for each fraction can be used to compute the longshore transport rate from the longshore component of energy flux (Coastal Engineering Research Center, 1984). The variation of the longshore transport rate is then used to determine the rate of erosion or accretion of the shoreline.

Table 4. *Design wave of Map Ta Phut Port.*

Wave Height (m)	Return Period (year)
3.12	10
3.67	50
3.85	100

Recently, the National Research Council of Thailand deployed nine oceanographic buoys in the Gulf of Thailand (Figure 1). Examples of wind and wave roses at Rayong are shown in Figures 11 and 12 respectively.

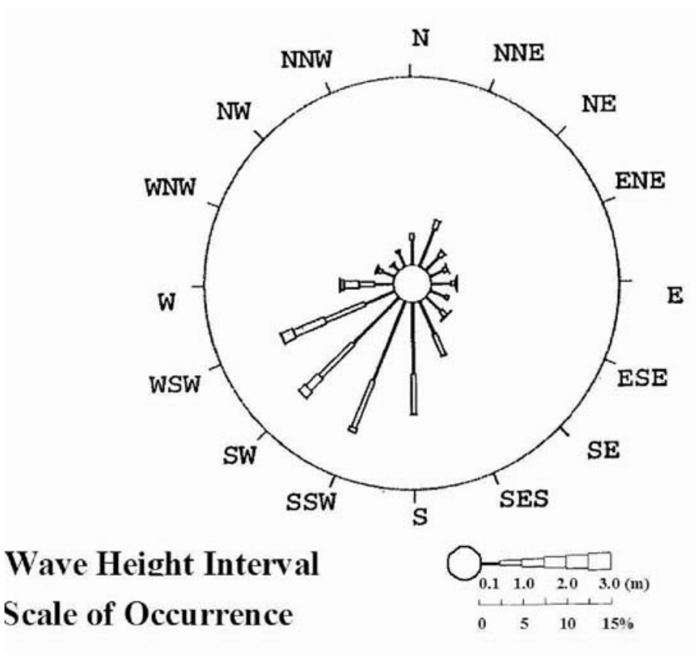

Figure 12. *Wave rose at Rayong.*

4. TIDES AND TIDAL CURRENTS

One of the components of the current system in the Gulf of Thailand is derived from the river flows, especially near the river mouths. The current in the Upper Gulf at some distance from the four major rivers, are dominantly tidal. Since the tidal current here is controlled by that in the whole Gulf of Thailand, the tide and tidal current in the whole Gulf of Thailand will be presented first.

The tidal charts were calculated by Yin (1977). The types of tide can be determined by the ratio, R, between the amplitudes of the major diurnal and semi-diurnal components:

$$R=\left(K_1+O_1\right)/\left(M_2+S_2\right)$$

There is no occurrence of tides having this ratio less than 0.25. An amphidromic point of the diurnal tide exists (Figure 13).

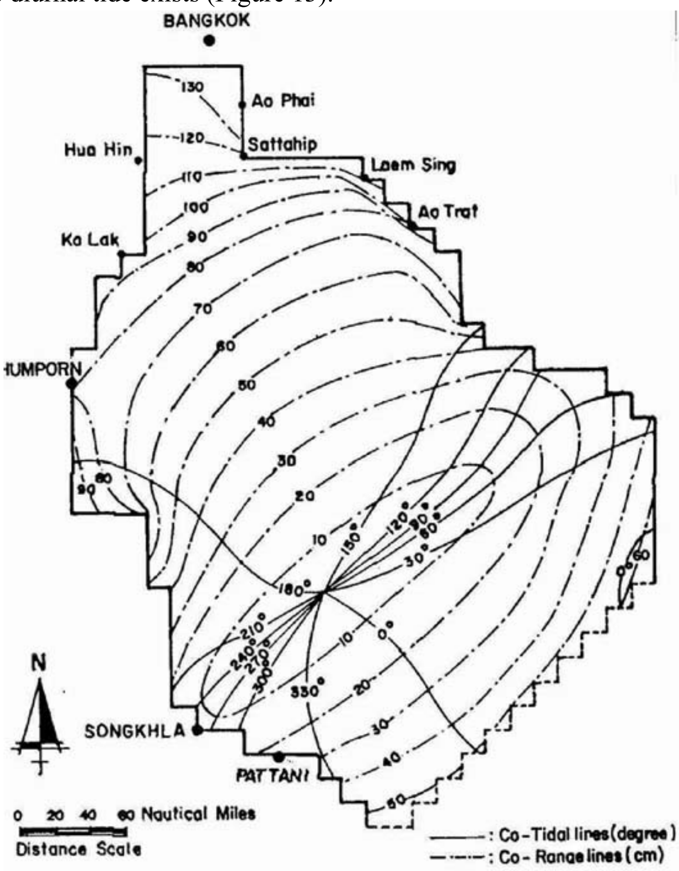

Figure 13. *Tidal chart of the K_1 component.*

For the semi-diurnal tide M_2, shown in Figure 14, the amphidromic point is located near Chumphon; as a result a strong diurnal tide occurs, whilst in the Upper Gulf a mixed tide results as shown in Figure 15.

The surface current and direction at Sichang buoy station in 14-day tide cycle is presented together with the tide in Figure 16; this shows complex patterns of mixed tidal currents.

The flushing time is defined as the time necessary for removal of the accumulated volume of freshwater by the inflowing river discharge. The freshwater flows to the Upper Gulf of Thailand are dominated by four rivers at the north-end especially the Chao Phraya river; its flow is approximately three times (or more)

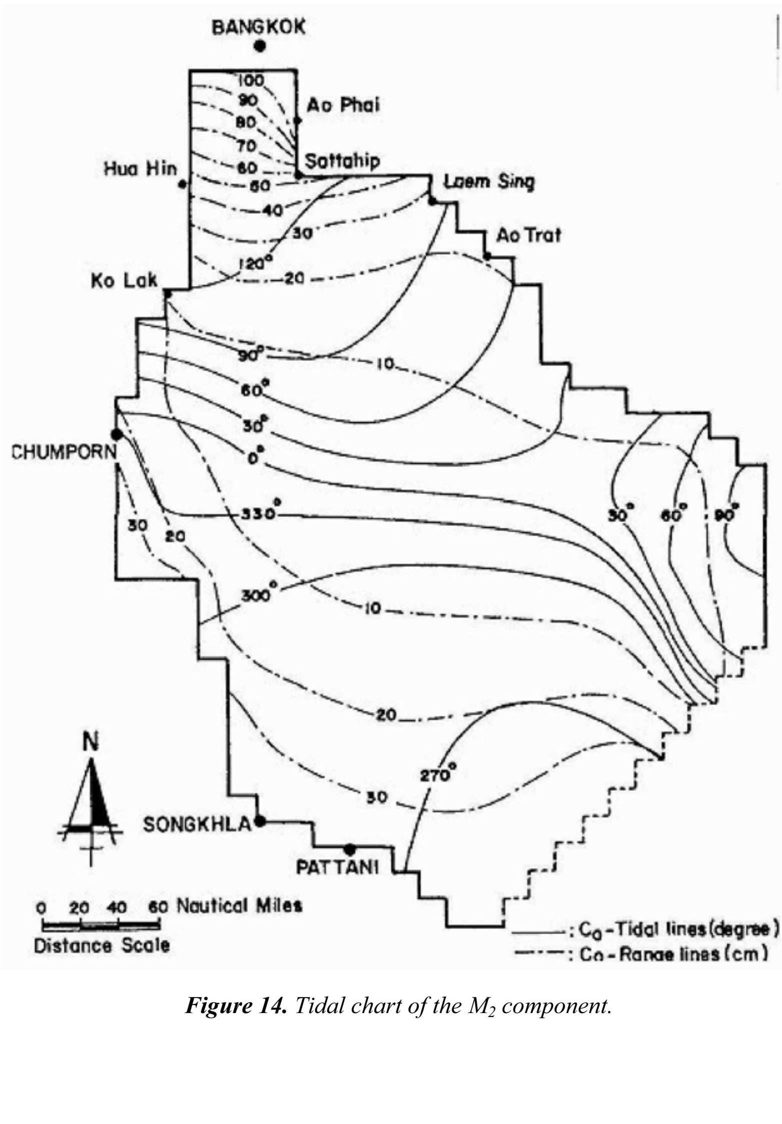

Figure 14. *Tidal chart of the M_2 component.*

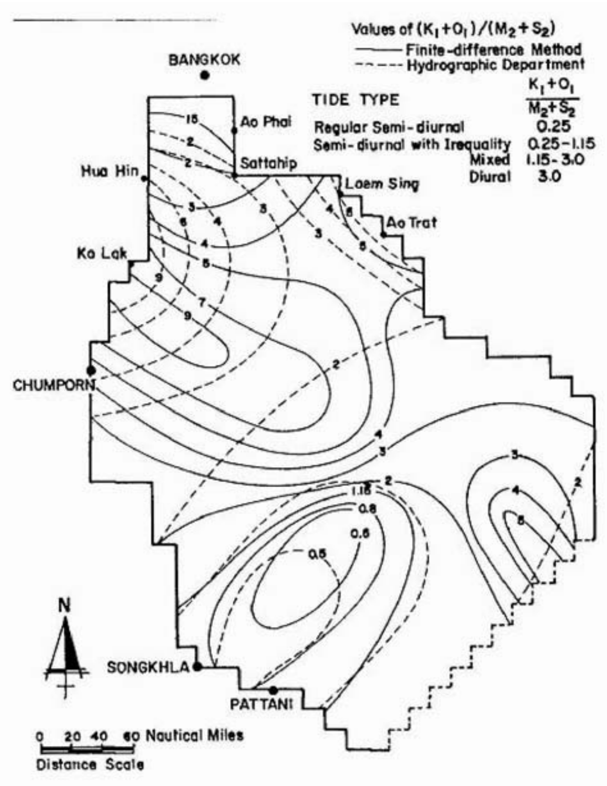

Figure 15. *Types of tides in the Gulf of Thailand.*

greater than the others. The salinity as treated as non-degradable pollutants, the estimated flushing time in the Upper Gulf of Thailand were monthly derived based on data in 1962 (McGarry et al., 1972). During April-May as known as dry season, pollutants discharged into the Upper Gulf of Thailand will be accumulated within the area because of the longest flushing time of over 300 days. The situation becomes better with the flushing time decrease to an average of 70 days, starting from August-November due to the flooding period of the river. Although the river discharge has a great importance but the seawater circulation plays the most importance role for the whole year round.

5. THREE IMPORTANT PORTS OF THAILAND

Three important ports of Thailand are Bangkok, Laem Chabang and Map Ta Phut. Bangkok Port, which is the major existing port for both export and import, and Laem Chabang Port, which is quite close to Bangkok (Figure 1), were planned to alleviate the congestion of Bangkok Port, especially in the export of agricultural products such as rice, tapioca and sugar. Construction of the port was completed in 1991. The discovery of natural gas in the Gulf of Thailand, and the onshore pipeline terminal at Map Ta Phut, led to the creation of the Eastern Seaboard Development Project for heavy industries based on natural gas resources such as petrochemical, fertilizer soda ash.

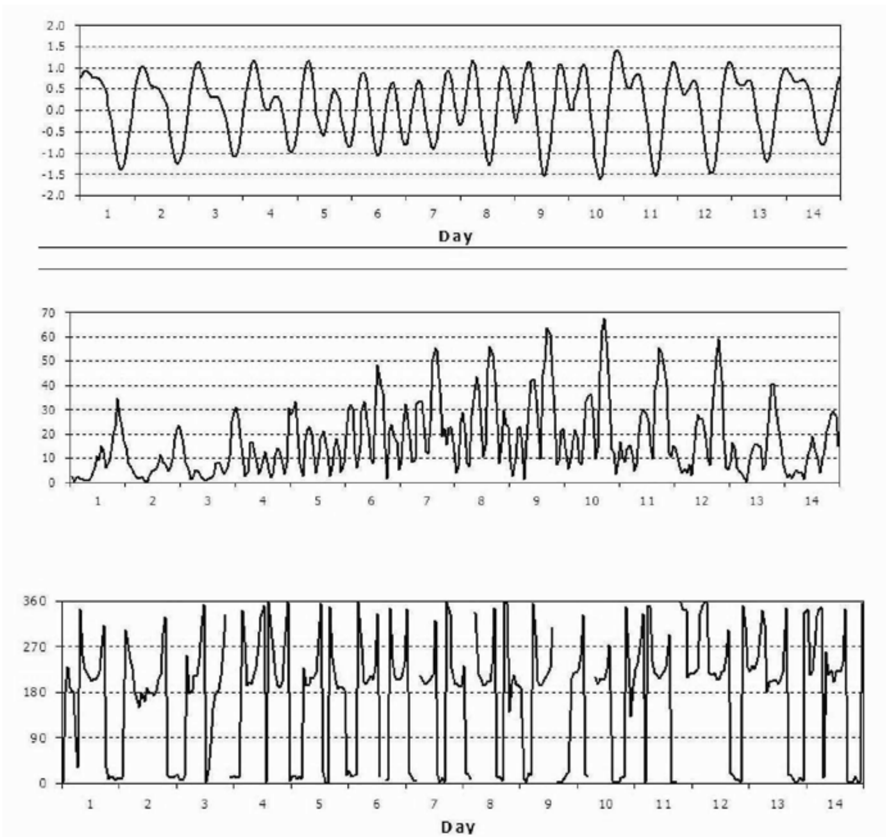

Figure 16. *Tides(top; in m), surface currents (middle; in cm s^{-1}) and direction (bottom; in ^{o}) at Sichang Buoy station during a 14-day tide cycle.*

5.1. Bangkok Port

In 1936 a consulting firm, Christini and Neilson, was assigned to design Bangkok Port and construction began two years later in 1938. The World Bank provided loans for improvement and expansion in 1951 and a law established the Port Authority of Thailand as a public utility state enterprise under the general supervision of the Ministry of Transport and Communications.

Bangkok Port is located in the Chao, Phraya River which flows through the city of Bangkok and drains its water and sediment into the Upper Gulf of Thailand through a dredged channel shown in Figure 17. The numbers of vessels calling at Bangkok Port were about 1800 each in 1985, 1986, 1987 and increased to 2100 and 2200 in 1988 and 1989 respectively.

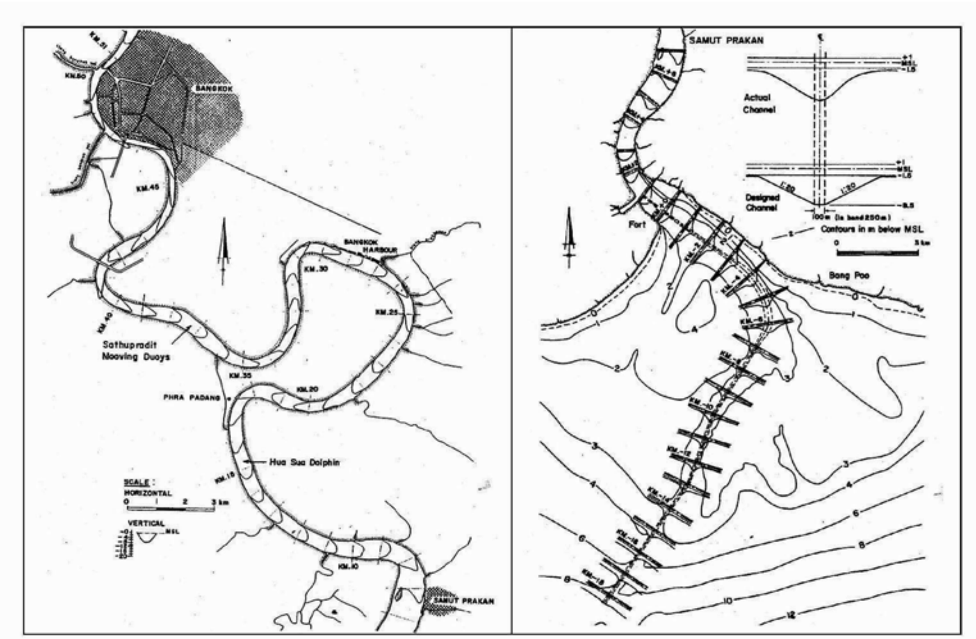

__Figure 17.__ Bathymetry of (left) the river part of the Seaway, and (right) the Bangkok Bar channel (NEDECO, 1965).

5.2. Laem Chabang Port

Since 1961, the Royal Thai Government has been making strenuous efforts to construct a new deep-sea port to accommodate larger with draughts of over 8.2 meters that cannot enter the Bangkok Port. In 1972, NEDECO recommended that the new port be constructed at Laem Chabang, Chonburi, due to its suitable location and the possibility of future expansion. A loan from the Overseas Economic Cooperation Fund (OECF) of Japan was granted. In 1984, a joint-venture of 4 consulting firms known as PAAS CONSORTIUM was assigned to undertake the detailed engineering design for the first stage development of the port. The design work was finished in April 1986. The construction contract was signed in 1987, and was completed in October 1991.

The Laem Chabang Commercial Port is designed to accommodate large container ships and bulk carriers which cannot be accommodated at the Bangkok

Port. It is also an infrastructure for the Eastern Seaboard Development Project. A 14 m deep, 2.5 km long channel is maintained by dredging. The port is protected by a breakwater 1.3 km long.

5.3. Map Ta Phut Port

Construction of the Map Ta Phut Port and its Industrial Estate started in October 1989, and were completed in April 1992.

Due to larger design waves, the breakwater at Map Ta Phut is much stronger and more massive than that at Laem Chabang.

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CHAPTER 16

ENVIRONMENTAL ISSUES IN THE GULF OF THAILAND

GULLAYA WATTAYAKORN

1. INTRODUCTION

The Gulf of Thailand is a semi-enclosed tropical sea located in the South China Sea (Pacific Ocean), surrounded by the countries Malaysia, Thailand, Cambodia and Vietnam. The Gulf covers roughly 320,000 km^2. The boundary of the Gulf is defined by the line from Cape Camau in southern Vietnam (just south of the mouth of the Mekong River) to the coastal city of Kota Bharu on the east coast of Peninsular Malaysia. It is relatively shallow; the mean depth is 45 m, and the maximum depth only 80 m. The general shape of the Gulf's bottom topography can be considered elliptic parabolic. It is separated from the South China Sea by two ridges that limit water exchanges with the open South China Sea. The first extends southeast from Cape Camou for about 60 nautical miles with an average sill depth of less than 25 m. The second ridge, which extends off Kota Bharu for approximately 90 nautical miles, has an average sill depth of 50 m. There is a narrow, deeper channel between the two ridges with the observed depth of 67 m (Emery and Niino, 1963). The Gulf may be divided into two portions, Upper Gulf and Lower Gulf. The Upper Gulf at the innermost area has an inverted U-shape. The Upper Gulf is the catchment basin of four large rivers on the northern side and two on the western coast. Numerous rivers discharge freshwater and sediment into the Gulf. Among them, the Chao Phraya River has the biggest volume transport next to the Mekong River. The average runoff per year of the Chao Phraya is 13.22×10^3 km^3 and that of the Mekong is 326×10^3 km^3. It is estimated that a considerable amount of nutrients is also discharged from these rivers, promoting primary productivity in the Gulf (Piyakarnchana et al., 1990).

Seasonal circulation in the Gulf of Thailand, deduced from oceanographic data measured in 1993-1994, suggested that circulation the Gulf is generally weak and variable. The mean circulation in the Gulf is forced by the South China Sea and not by the local wind as previously suspected, a phenomenon particularly marked during the Northeast monsoon when Mekong River water enters the lower Gulf (Wattayakorn et al., 1998). During January to February, currents throughout the Gulf are at their weakest, with little mixing of Upper and Lower Gulf water masses.

E. Wolanski (ed), The Environment in Asia Pacific Harbours, 249–259.

From March to August, anticyclonic (clockwise) circulation predominates in the Lower Gulf, and penetrates into the Upper Gulf. In September, the circulation direction reverses to cyclonic in the Lower Gulf, initially producing cyclonic current in the Upper Gulf; by November the flow in the Upper Gulf becomes anticyclonic (Figure 2). On the whole, the Gulf of Thailand is poorly flushed. In the Upper Gulf, little mixing occurs between coastal and offshore waters. As a consequence of these comparatively static conditions, contaminants discharged into the Upper Gulf may accumulate. Variability in current directions may also result in the return of contaminants that were initially flushed from an area.

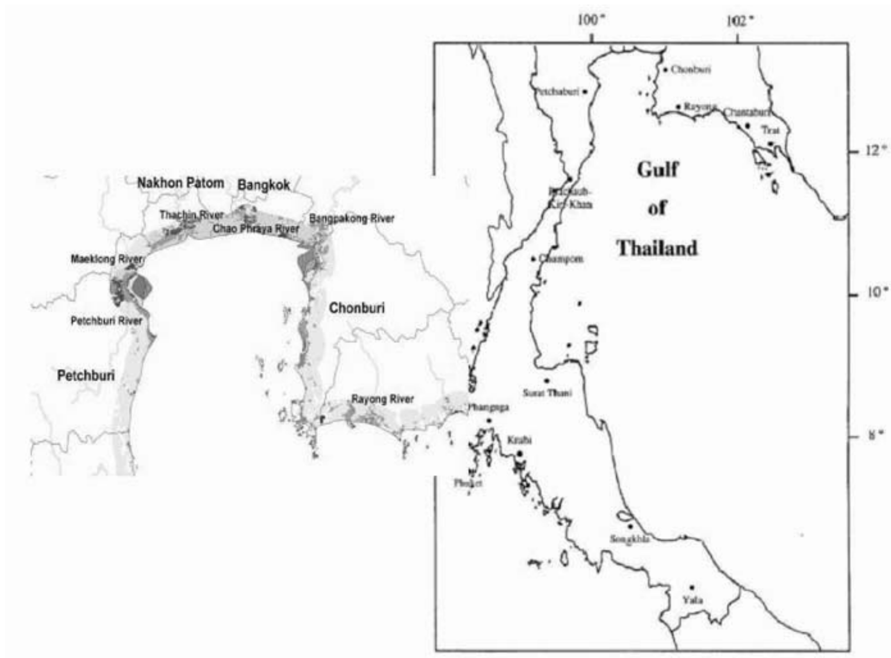

Figure 1. *Map of the Gulf of Thailand, showing major rivers discharging into the Inner Gulf.*

The innermost part of the Gulf, in the vicinity of Bangkok, extends from Ban Pak Thale, Phetchburi Province, in the west, north and east past the mouths of the Maeklong, Thachin, Chao Phraya, and Bangpakong Rivers, to the town of Chonburi in the southeast (Figure 1), is a large area of intertidal mudflats around the shores of a huge, shallow sea bay forming the estuary of the four major rivers. The area formerly supported extensive mangroves. While the largest areas have now been cleared for aquaculture and salt pans, much secondary mangrove still remains and is usually found as a narrow (10-l00 m) fringe along the seaward margins. Extensive areas of low scrub are found in the brackish marshes along the landward edge. In places, the open shrimp ponds and salt pans extend two to three km inland and, together with the offshore mudflats, provide an important feeding and roosting area for many thousands of shorebirds (Erftemeijer and Jugmongkol, 1999). They are also important fish breeding and nursery grounds, where many species reproduce. The barriers on most major rivers, such as dams, weirs, and hydropower structures, also have a major impact on migratory species that swim upriver to spawn.

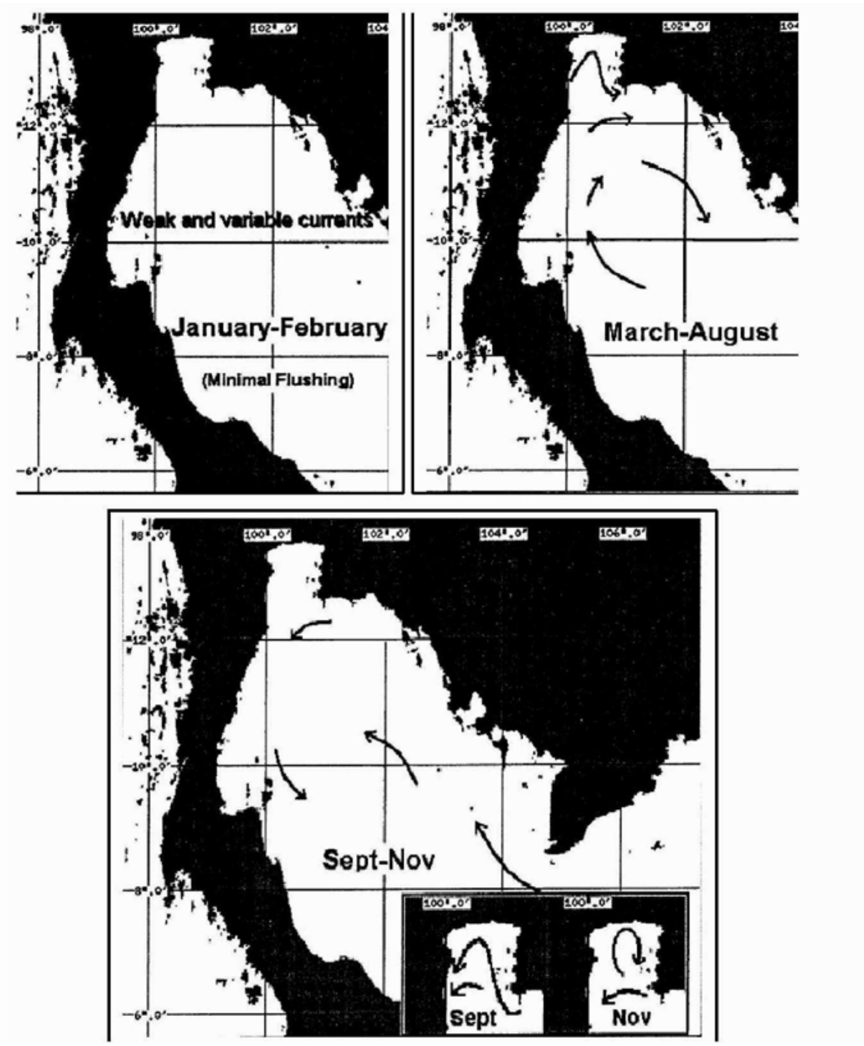

Figure 2. Sketch of the water circulation in the Gulf of Thailand, deduced from the oceanographic data during 1993–1994. (Adapted from Wattayakorn et al., 1998).

The Gulf of Thailand is an important resource to the national economy of Thailand that receives income from the Gulf fishing industry, tourism and port operations. Bangkok is the economic center of Thailand. The human population density is extremely high, and there is an increasing amount of heavy industry, especially extending eastwards from Bangkok along the lower reaches of the Chao Phraya River. The Port of Bangkok, at the mouth of the Chao Phraya River, is the transportation life-blood of the country. It serves as the main egress point for agricultural export products from the country's interior as well as access for raw materials such as fertilizer, grain, steel and oil products. In addition, several deep sea ports within the Gulf have been constructed to facilitate the huge increase in sea transport of import and export cargoes. The deep sea port at Laem Chabang provides relief to the congestion in Bangkok, while the ports at Songkhla and Map Ta Phut offer ocean shipping alternatives to industries and resources within the Gulf of Thailand.

Thailand's population increased from 23 million in 1961 to 65 million by 2004. The rapid increasing population with associated industrialization and economic development in the coastal areas in the past two decades has resulted in construction and planning of many coastal development projects. Many of these developments have altered local coastal and estuarine processes, which have caused changes in the shoreline and sediment processes. In addition, the Gulf of Thailand has been a receptacle for agriculture; lumbering; ports and shipping; fishing and aquaculture; human settlement; recreation and tourism for a long time. The marine waters and coastal areas of Thailand, therefore, have been threatened by both the forces of nature as well as by human activities. Foremost among these threats are siltation; pollution from domestic and industrial wastes; heavy metals; agrochemicals, particularly pesticides; pollution resulting from oil spills from tankers and from onshore and offshore drilling for oil and minerals. In addition, the vast ecosystems of mangrove forests, coral reefs, and intertidal flora and fauna of the country are threatened by the conversion and/or reclamation of land for multipurpose uses and by recreational activities. Therefore, the problems of the marine environment in the Gulf of Thailand may be broadly divided into those of overexploitation of fisheries, loss of habitats, and pollution. This paper will briefly deal with specific items under these three headings.

2. OVER-EXPLOITATION OF FISHERIES RESOURCES

The Gulf of Thailand is among the productive areas of the South China Sea in which Thai people are dependent on marine fisheries. The rising demand for fish from the increasing population, for food and export, has led to the rapid increase in marine fishery production in the country, especially in the seventies and eighties. In the Gulf of Thailand, most of the pelagic fishery except Indian mackerel is fully exploited. Almost all demersal stocks are also overfished including fish, shrimp, squid, cuttle fish, and others (FAO, 1995). Overfishing of the inshore and coastal waters has also been reported in many technical publications (Chullasorn and Chotiyaputta, 1997; Jirapanpitat, 1992).

The marine capture fisheries in the Gulf are characterized by the use of multi-gear by a large number of small-scale fishermen, to exploit a large number of fish and other aquatic organisms. The Gulf trawl fisheries suffer from overcapacity, due to their uncontrolled growth in the late 1960s - early 1970s. Statistics from the Department of Fisheries have shown that the coastal and inshore fishery resources have been fully exploited since the early seventies (Supongpan, 1996). The massive increase in trawling effort that occurred from the early 1960s on, and which resulted, in the early 1980s, in a strong decline in catch per effort (CPUE), from about 290 kg h^{-1} in 1963 to about 50 kg h^{-1} in the 1980s and 20-30 kg h^{-1} in the 1990s. Night-time CPUE has also declined drastically to less than half of its earlier value from 57 kg h^{-1} in 1976 to about 21 kg h^{-1} in 1995 (Eiamsa-Ard and Amornchairojkul, 1997). Captured fisheries statistics (1986-1995) of Suratthani indicate that fishery production of Bandon Bay area was greatly declined during this period. Most of fish production was mainly trash fish resulting from excessive fishing effort, capture of undersize fish as well as the destructive nature of the push net which was used as the main fishing gear in the area (Wattayakorn et al., 1999). Changes in the species and size composition of the catch are other clear evidence of the onset of overfishing. Chullasorn and Chotiyaputta (1997) reported that the catch from trawl surveys shows that 30% to 40% consists of trash fish. Of this trash fish, 30% comprised of juveniles of commercially important species. Due to overfishing and the resulting decreased availability of fish, subsistence and artisanal fishermen are often forced into destructive fishing techniques such as blast fishing and poisons. The use of such fishing techniques can result in lasting deleterious impact to the marine environment, especially to coral reefs.

3. DEGRADATION OF COASTAL ECOSYSTEMS

3.1. Mangroves

Approximately 60% of the current population in Thailand (roughly 39 million inhabitants) lives in coastal towns and villages. This leads to rapid coastal development for industries and housing, and extensive coastal habitat destruction and loss. Mangrove destruction is the most obvious and has probably had the greatest loss. With Thailand's developing economy, since the 1960s the mangrove forests along the Gulf of Thailand have been reduced by 50-80% (Table 1). Mainly shrimp farms, hotels, growing cities and other coastal developments, have replaced them. Estimates of the amount of mangrove conversion due to shrimp farming vary, but recent studies suggest that up to 50-65% of Thailand's mangroves have been lost to shrimp farm conversion since 1975 (Aksornkoae and Tokrisna, 2004; Charuppat and Charuppat, 1997; Dierberg and Kiattisimkul, 1996). Mangroves are still being cut down for agriculture, aquaculture, and coastal construction but at a much lower rate. For example, mangrove forest in the Bandon Bay area was depleted by 30% during the period of 1993-1998 as compared to 87% during 1961-1986, most of which has been converted to shrimp ponds (Wattayakorn et al., 1999).

The high profits of shrimp farming resulted in the increase in the number of shrimp farms and culture areas from about 6,000 farms in 1987, taking up an area of 45,000 hectares, to 28,000 farms covering 79,000 hectares in 1999 (DOF, 2001).

Rapid expansion of intensive shrimp farming along the coast, some of which involved clear-cutting of mangrove forests, has caused many environmental problems such as poor coastal water quality, deteriorating of marine resources, and saltwater intrusion into nearby agricultural areas. The catastrophic collapses of several shrimp farms in the Upper Gulf of Thailand in 1989 were attributed to poor water quality originated from industrialization within the watersheds and to pollution from the farms (Dierberg and Kiattisimkul, 1996). Many shrimp ponds are abandoned after they become unprofitable, leaving vast areas unsuitable for agriculture or other aquaculture activities.

Table 1. *Comparison of the mangrove areas in Thailand in 1975 and in 1996. (Adapted from Charuppat and Charuppat 1997).*

Mangrove area	1975 (rai)*	1996 (rai)*	Decrease (rai)*
Gulf of Thailand	756,250	216,741	539,509
Eastern Region: Trat, Chantaburi, Rayong, Chonburi, Chachoengsao	306,250	79,113	227,137
Central Region: Bangkok, Samut Prakarn, Samut Sakhon, Samut Songkhram, Petchburi, Prachuab Khirikhan	228,875	34,057	194,068
Southern Region: Chumphon, Suratthani, Nakhon Si Thammarat, Phattalung, Songkhla, Pattani	221,125	103,571	118,304

Note: 1 rai = 0.16 hectare

3.2. Coral reefs

Coral reefs are under stress in many areas in the Gulf, especially those near shallow shelves and dense populations. Storms and monsoon waves are the major natural causes of coral reef damage. Typhoon Gay hit southern Thailand in 1989 and caused major damage to some reefs. Extreme low tides and coral bleaching are other natural phenomena causing severe damage. The erosion that has resulted from logging has killed coral reefs by increasing the turbidity of coastal waters, such that the coral polyps no longer have enough light to photosynthesize metabolites and may even be buried by increased sedimentation. Over 60% of all major reef groups in the Gulf of Thailand have less than 50% live coral cover and there is increased algal growth because of nutrient pollution from the land, including near the major tourist resorts of Pattaya Bay and Samui Island (Chansang and Phongsuwan, 1993). Other anthropogenic disturbances on localized coral reefs in the Gulf are boat grounding and destructive fishing methods such as the use of dynamite and bottom-trawlers (Sudara and Patimanukasem, 1991). Loss of coral reefs has long-term implications because of the time that they take to recover.

3.3 Seagrass beds

Seagrass beds are the least studied marine habitats compared to coral reefs and mangroves. Seagrass beds in Thailand are more abundant in the Andaman Sea than in the Gulf of Thailand. However, no comprehensive evaluation on the seagrass cover in the country has been undertaken to date. Species of seagrass such as *Enhalus acoroides, Halodule pinifolia,* and *Halophila ovalis* were reported in the Gulf of Thailand. A survey of seagrass around Samui Island found degraded seagrass beds in those areas where there were considerable industrial construction, shrimp farming and land development (Poovachiranon et al., 1994). It is apparent from the site surveys that economic activities are the main factor affecting seagrass depletion (Nateekanjanalarp and Sudara, 1992).

4. COASTAL AND MARINE WATER POLLUTION

Pollution has considerably degraded the coastal and marine environment, including estuaries, of Thailand over the past three decades. Coastal and marine water pollution in Thailand is mainly due to direct discharges from rivers, surface runoff and drainage from port areas, domestic and industrial effluent discharges through outfalls and various contaminants from ships. Urban centers in Thailand are often located on coasts and estuaries and much of the domestic wastes and garbage is dumped directly into the shallow coastal environment. Hence, rivers are generally heavily contaminated with municipal sewage, industrial effluent and sediments.

The primary sources of marine-based pollution are offshore oil and gas operations, wastes from maritime transportation, shipping and oil spills. For land-based pollution, the primary sources are domestic sources, industrial development and tourism areas, especially beach resorts and agriculture and aquaculture activities (Piyakarnchana et al., 1990). Land-based sources contribute some 70% of the pollutants, mostly from domestic sources. An estimated volume of more than 200,000 tonnes of waste quantified as BOD is discharged into the Gulf each year (Chongprasit and Srinetre, 1998). A smaller quantity of industrial waste consisting of more toxic chemical substances is also released into the Gulf. About 50% of the land-based contaminants are delivered into the Gulf by the four large rivers at the head of the Upper Gulf. In general, water quality is lower than acceptable standards in the Inner Gulf region, especially at the mouths of the four major rivers, the popular tourist spots along the coast, and near certain islands. Water quality is deteriorating due to increasing inputs of nutrients from the increased use of fertilizers in agriculture, the mariculture industry and from household sewage. This chapter will address some current problems i.e. eutrophication (red tides) and oil pollution in the Gulf of Thailand.

4.1. Red Tides

Primary production prevailing in the Gulf of Thailand is known to be relatively high, with a recent boost by increased nutrients from rivers and shrimp farms, which in turn leads to increasing occurrences of phytoplankton blooms (or “red tides”), oxygen depletion events, food poisonings and other pollution effects, particularly in the Inner Gulf (Eiamsa-Ard and Amornchairojkul, 1997; Longhurst, 1998; Piyakarnchana, 1999).

The occurrence of red tides in Thailand was first reported in 1957. In the past, red tides were regarded as natural phenomena and there was no serious impact of red tides on the marine environment or organisms. However, its frequency has been increasing in recent years. *Noctiluca scintillans* and *Trichodesmiun erythraeum* were the two species of plankton that frequently bloom in the Inner Gulf (Suvapeepan, 1995). Blooming of *Noctiluca* sp. usually changes the apparent color of water into dark green. The bloom of *Trichodesmium* sp. changes the apparent color of seawater into yellow green color and then to red brown. The algal blooms caused by both species of phytoplankton have no direct harmful effects on fish and shellfish. However the heavy blooms can result in sudden reduction of dissolved oxygen and a high amount of ammonia concentration in the water, which in turn sometimes lead to fish kills. A large bloom of *Noctiluca sp.* caused a mass mortality of fish at Sriracha Bay in August, 1991 and along the Pattaya Bay in August, 1992 (Sukasem, 1992). Blooming of diatoms i.e. *Rhizosolenia* sp. and *Chetoceros* sp. was also reported to occur as a result of eutrophication in the Inner Gulf (Suvapeepan, 1995).

According to the survey by the Aquatic Resources Research Institute, Chulalongkorn University (2003), 97 incidents of red tides were recorded in the Gulf of Thailand from 1957 to 2001. To date, there is only one incident of paralytic shellfish poisoning (PSP) recorded in Thailand, in May 1983 after consuming the contaminated green mussels (*Perna viridis)* in the red-tide area of the Pranburi River mouth. Sixty-three people were ill and one died because of this incident (Tamiyavanish, 1984). The causative organism of that incident could not be established. Since then, all the phytoplankton blooms recorded in Thailand are harmless to humans.

4.2. Oil pollution

Oil pollution in the Gulf of Thailand has risen with industrial development in coastal regions. The shipping of oil coupled with increasing emphasis on offshore oil exploration makes the Gulf of Thailand extremely vulnerable to oil pollution. Wattayakorn (1986, 1987 and 1991) has reported chronic petroleum hydrocarbon contamination in coastal waters. Pollution was believed to originate primarily from the discharge of oil from small coastal boats, via urban, industrial, refinery and sewage effluent. Additional oil contamination could also originate from maritime transportation of crude and refined oil through the region, as a result of the discharge of ballast water from tankers. Increased pollution in the form of tarballs and oil slicks has been observed in the past years (Wattayakorn et al., 1998). The deleterious effects on the marine environment and living resources as a result of the growing frequency of oil spills (due to both constant deballasting activities and accidents such as collision in shallow waters) have caused public concern and gained widespread attention in environmental protection in Thailand.

The accidental oil spills have been frequently reported along oil transport routes, at points of discharge and loading of oil carriers. There have been over 50 oil spill accidents reported to occur in the Gulf during 1973-2002. Frequent spills were found at the mouth of the main entrance to the Bangkok Port (Chao Phraya River mouth) and Laemchabang Port (Chonburi Province). These oil spills represent the greatest source of petroleum related pollution in the Gulf. These incidents are expected to continue because of insufficient understanding of navigational routes and inadequate contingency plans. A list of large oil spill accidents that have occurred recently in the Gulf can be found in the Table 2.

Table 2. *Recent major oil spill accidents in the Gulf of Thailand.*

Date	Oil type	Volume (tonnes)	Location	Cause
6 Mar 1994	Diesel	400	Chonburi Province	Collision of tanker and container vessel
30 Oct 1996	Crude Oil	160	Rayong Province	Leaking during loading
15 Jan 2002	Diesel oil	230	Chonburi Province	Grounding
17 Dec 2002	Bunker Oil	230	Chonburi Province	Collision of tanker and container vessel

Under normal operations, most cargo and oil/gas ports are not major sources of pollution. Only in fishing ports, where regulations on pollution control are difficult to implement on small boats, is oil pollution from fuel/lubrication oil dumping and bilge water discharge seen. Fishing ports exist in every coastal province and they are usually near to major urban areas, thus making it difficult to separate the contribution from the two sources. There are probably over 40,000 fishing boats of various sizes registered and operating in the Gulf of Thailand. The used lubricating oil from these fishing boats is believed to be illegally dumped into the sea. In addition, leaks and spills of fuel (diesel) oil during filling and transfer also occur.

There is insufficient institutional and administrative capacity to ensure environmentally responsible maritime practices. There is also a prevalence of inadequate skills to detect, control and enhance areas of spills.

5. MANAGEMENT APPROACHES

Despite the growing awareness and concern of the public, coastal and other aquatic ecosystems continue to be degraded by pollution and unsound forms of utilization. Scientists have been conducting research and monitoring activities on the Gulf for decades. These activities were typically geared for a specific need, limited in coverage, and failed to provide an overall picture of the Gulf's condition. There is still a poor understanding of the biological, physical, chemical and socio-economic In addition, politicians and the general public are not aware of the potential value of marine science to society.

However, several measures have been taken for protection and conservation of marine resources in the Gulf. Since 1972, trawlers are prohibited from an area within 3 km from the shoreline, and within a perimeter of 400 m of any stationary fishing gear. In 1984, an area of about 26,400 km^2 in the Gulf was declared as a conservation area, prohibiting fishing by all gear types during spawning season from

15 February to 15 May (Phasuk, 1994). To provide protection of endangered and threatened species, the Department of Fisheries issued regulations prohibiting the catching of sea turtles, the collection of their eggs and the export of sea turtle shells. Catching of dugongs and collection of corals are also prohibited. The Department of Fisheries also emphasizes the rehabilitation of fishing grounds, and promotes artificial reef projects to create protected habitats for marine life. However, enforcement of these regulations is far from effective (Phasuk, 1994). Royal Forestry Department (RFD) has also established and managed several marine national parks, and prohibited some fishing activities in certain areas.

Economic measures have been introduced to support the prevention of environmental problems. Incentives such as import tax reduction for environmentally friendly equipment and machines, recognition for companies that operate in an environmentally friendly manner, and use of the polluter pays principle (PPP) to encourage voluntary action on pollution prevention in private companies. An environmental fund was set up and managed effectively as a financial support for environmental liability and cleanup. The budget derived from the energy conservation fund was also used to conduct the energy saving campaign.

Other government activities undertaken include education and the dissemination of environmental knowledge through the media. The government also encourages more involvement from the private sector and non-government organization (NGOs). Many obsolete environmental laws and regulations were amended to suit the current situation. Moreover, the government attempted to issue several significant laws such as the Community Forest Act and Water Resource Act. The Thai government has shown its commitment to international environmental problems by finalizing its ratification of the Convention on Biodiversity while implementing the other ratified conventions. These activities will all help to alleviate environmental problems in the future.

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CHAPTER 17

THE ENVIRONMENT IN HO CHI MINH CITY HARBOURS

NGUYEN HUU NHAN

1. INTRODUCTION

Ho Chi Minh City (HCMC) is located in the southern region of Vietnam (Figure 1). HCMC, including the Thi Vai–Vung Tau (TV-VT) area (Figure 2), had, has, and will have for the foreseeable future the biggest harbour network in Vietnam. The shipping routes in HCMC, including the Thi Vai–Vung Tau (TV-VT) area, meander through a large network of rivers, canals, and shallow bays (Figure 2). Almost every river and canal is a waterway for ships and boats.

The shipping routes have played a very important role in Vietnam's social and economic development, especially in the Southern Focus Economical Zone (SFEZ). Harbours here play a role as the junction of import and export of goods in the South of Vietnam. They have served also as entrance corridor to Cambodia and other countries. Actually, this was and is the busiest navigation hub of Vietnam.

All these shipping routes operate in very sensitive regions such as seasonal wetlands, mangroves, urban centers with high population density, industrial zones, and aquaculture zones. Therefore, the environment faces serious challenges as:

- Water and bottom mud polluted by sewage from domestic, industrial, aquaculture, and shipping.
- The degradation and reclamation of mangrove ecosystem, the erosion of river banks and marine shorelines, and the depletion of biological resources by urbanization, non-planned infrastructure development, shrimp ponds, dykes, roads, harbours, and shipping routes.
- The increase occurrence of environmental accidents, especially oil spills because tidal currents are strong, the waterways are narrow, twisted and shallow, and the infrastructure to prevent such accidents is deficient and poor.

This manuscript describes the environmental status of the harbours of HCMC and TV-VT area, and details the problems and proposes strategies for their solution.

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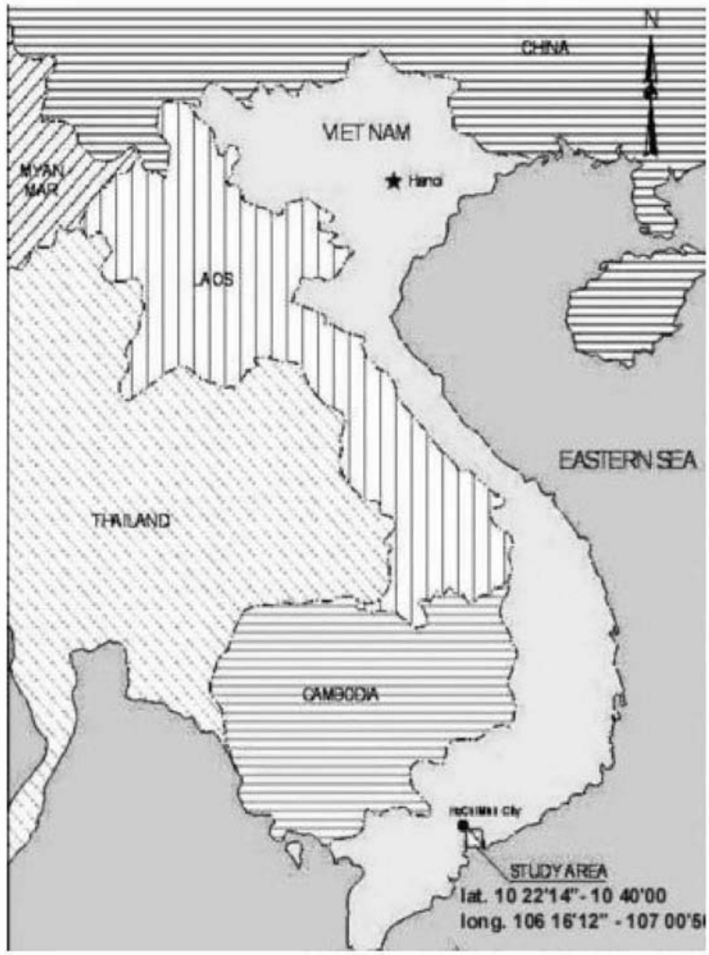


Figure 1. *A general location map.*

2. GENERAL OVERVIEW

HCMC, a city in southern Vietnam, formerly known as Saigon (Figure 1), is located just northeast of the Mekong River Delta. It is located 1,730 km from Hanoi and 50 km from the South China Sea. The city is crisscrossed by many rivers, arroyos and canals, including the Saigon, Dong Nai, Nha Be, Soi Rap and Long Tau rivers (Figure 2). HCMC is Vietnam's major harbour and an important commercial and industrial center. Its area is 2,095 km^2 with a population about 6 million in 2004. The main ethnic groups are the Vietnamese (80%) and the Chinese (18%).

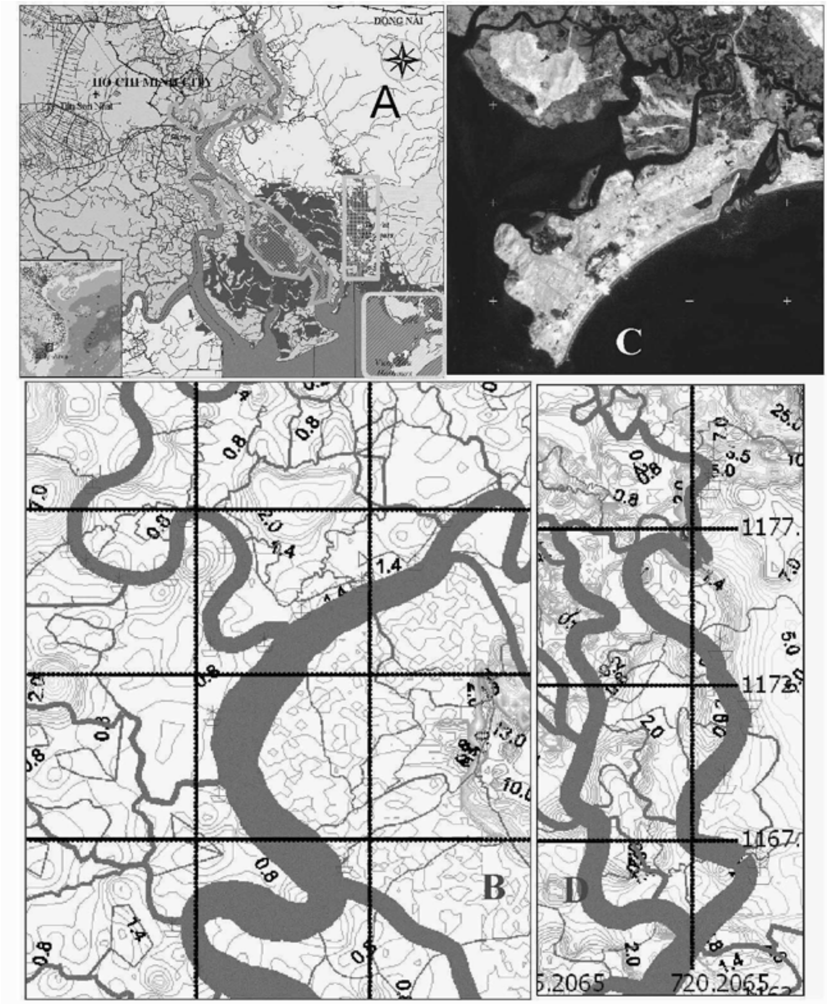


***Figure 2**. **A:** The location of study. The shipping routes in HCMC and TV-VT area are operating in very sensitive regions as: wetland, mangrove, and domestic centres with high population density, industrial zones. B: The harbour network in HCMC located in Sai Gon, Dong Nai, Nha Be, Soi Rap rivers. C: The harbour network in Vung Tau City's coastal zone and the Dinh River. D: The harbour network in Thi Vai River located in mangroves.*

Many centuries ago, Saigon was already a busy commercial center. Merchants from China, Japan and several European countries would sail upstream the Saigon River to reach the islet of Pho, a trading center. In 1874, Cho Lon merged with Saigon, forming the largest city in the Indochina. It had been many times celebrated as the *Pearl of the Far East.* After the reunification of the Vietnam, the 6th Vietnamese Assembly in its meeting of the 2nd of July, 1976, has officially renamed Saigon as HCMC. The history of the city relates closely with the struggle for the independence and freedom of Vietnam.

HCMC is the main junction for trains, roads, water, and air transportation systems for domestic trips and foreign destination. The Southern Focus Economic Zone (SFEZ), lead by HCMC, has an area of 2,804.3 km^2, and a population of 12,607,500 person in December 2003. A large volume of agricultural, forestry and industrial products, including processed foods from the Mekong River delta and the Northeast, Southern and Central highlands is transported through the harbours in HCMC and TV-VT area.

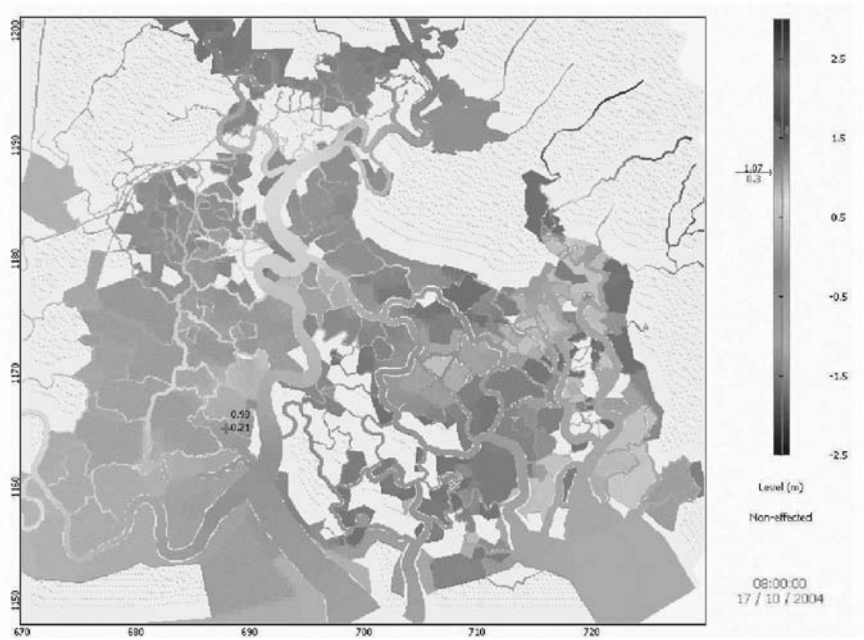
During the past few years, the SFEZ has been the economic growth area of Vietnam. The total GDP of the SFEZ in 2000 was VND 85,862 billions (at constant 1994 price) and this accounted for 31.2% of Vietnam's GDP. The SFEZ with its high concentration of industry plays a vital role in Vietnam economy. Since the early 1990s, it has also been successful to attract foreign and local investors by establishing industrial parks and export processing zones.

The climate is generally hot and humid. There are two distinctive seasons: the rainy season, from May to November, and the dry season, from December to April. The annual average temperature is 27° C. The hottest month is April and the coolest month is December. The insolation is 5-9 hours per day of sunshine. The annual average rainfall is about 1,800 mm in HCMC and 1,400 mm in Vung Tau, with most of the rainfall concentrated in June and September. The rainy season begins in late May and ends in late October. The number of rainy days per year is about 160. The potential evaporation in the area amounts 120.4 mm $month^{-1}$ and the highest in June (173.2 mm). The mean annual relative humidity is 80%. The wind direction in the dry season is from East to Southeast and in the rainy season from West to Southwest. The average wind speed at about 2.5-3.5 m s^{-1}. Typhoons are rare. Calm periods are uncommon. The sea breeze effect is quite large; hence easterly winds are strongest from 1400 to 2000 h.

There are 7-10 km^2 of streams and channels traversing the area. The tidal regime is semi-diurnal with amplitude of water level oscillation is 2 m at neap tide and 4 m at spring tide; this generates strong currents and rapid sea level variations that complicate navigation (Animation 1). Since the average elevation is lower than 1.5 m, two thirds of the area is submerged under high tide, creating a saline estuary wetland.

Alluvial soils are the principal soil type. These soils are generally slightly acidic (pH values of 4.5-6.5). Soils are predominantly saline sulphatic clay or mud with large quantities of sulfites which become oxidized to sulfates and hence acid when exposed to air; there are also marine sandy soils and sand dunes, which all were developed from new deposits of marine and riverine alluvia in the Quaternary period (UNESCAP, 1995).

An important natural ecosystem is protected in the Can Gio Mangrove Biosphere Reserve (CGMBR; 10° 22' N and 106° 46' E; Figure 3). It is located in the downstream reaches of the Dong Nai – Saigon river network that flows southeastward from HCMC, and is thus impacted thus by wastes from HCMC. Actually, all shipping Routes in HCMC and TV-VT area are located around



Animation 1. *The flood and inundation in the study area in the rainy season as predicted by the integrated tool "HydroGis". The tidal influence is dominant. The flow changes direction four times per day. The tidal regime is semi-diurnal with the amplitude of water level oscillation of 2 m at neap tide and 4 m at spring tide. Since the average ground elevation is lower than 1.5 m, two thirds of the area is submerged under high tide, creating an estuarine saline wetland.*

CGMBR; thus human activities threaten this ecosystem. The total area of CGMBR is 75,740 ha, of which the core zone comprises 4,721 ha, the buffer zone 41,139 ha, and the transition zone 29,880 ha. In 2001, UNESCO officially recognized Can Gio as a world's biosphere reserve. In mid-2003, the World Tourist Organization (WTO) recognized the Can Gio Mangrove Ecological Tourist Zone as one of the world's 65 most sustainable ecological tourist zones. Since 2003, the birds' nesting ground and Bat Swamp are considered by the People Committee of HCMC for a conservation zone in order to preserve the ecology of the wetland forests, conserve the CGMBR biosphere values, and create a place for sightseeing, studies, and education.

3. SHIPPING ROUTES

Shipping activities in HCMC and the TV-VT area (HAECON, 2001) are intensive in the complex river network and in Ganh Rai Bay. There are a number of large harbours and access channels (Figure 2). The present harbour network in the SFEZ may be divided into two main groups; one group is centered in HCMC and the other group in the TV-VT area. There are three access channels; firstly the shipping lane

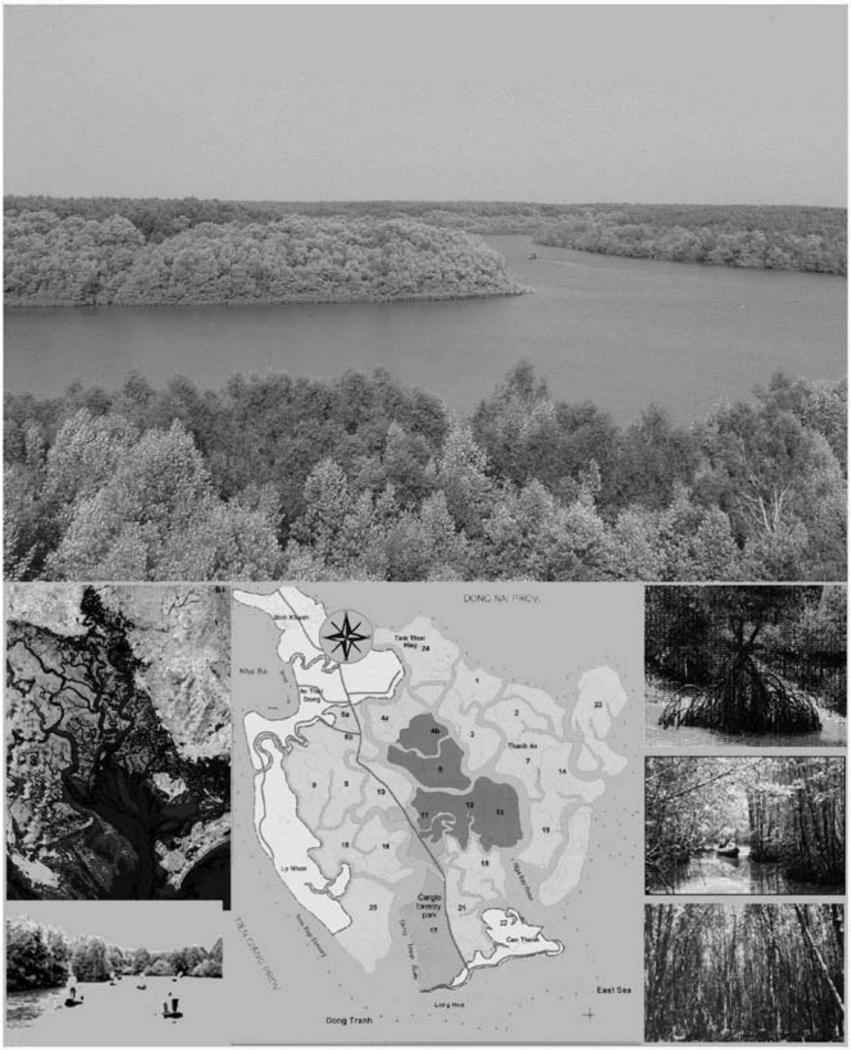


Figure 3. Maps and photograph of the Can Gio Mangrove Biosphere Reserve.

from the open sea to Ganh Rai Bay, secondly from Ganh Rai Bay to harbours in HCMC along Long Tau-Nha be-Sai Gon rivers, and thirdly from Ganh Rai Bay to harbours in the TV-VT area along Thi Vai river.

All of them are operating in very sensitive areas including wetlands, mangroves, and urban centers with high population density and industrial zones (Figure 4). Marine resources are heavily used and pollution is obvious (Figure 5).

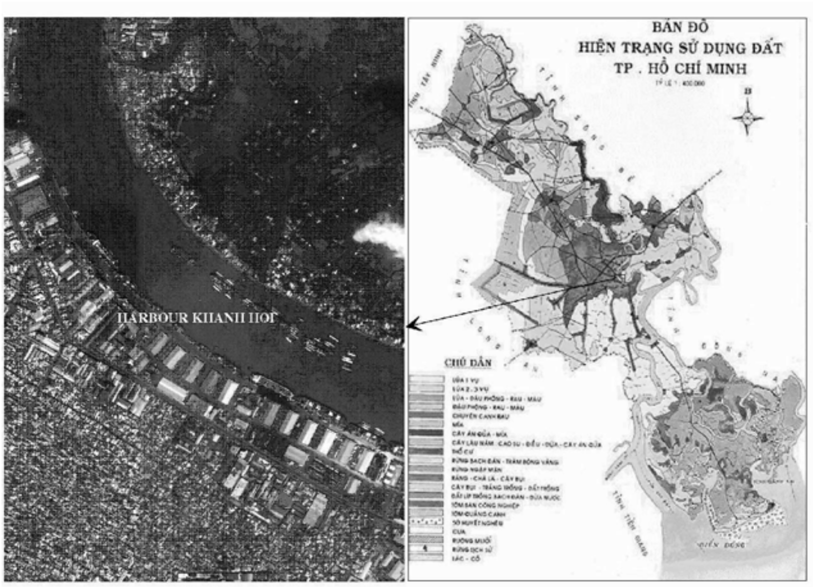


Figure 4. *The present harbours in HCMC located inside the high density urban areas face huge pollution, therefore their environment is alarmingly degraded.*

4. HARBOURS IN HCMC

At present, Harbours in HCMC consist of about various 27 harbours and shipyards and can be divided into the following 5 sub-groups: a group in the Sai Gon River; a group in the Dong Nai River (Harbour Cat Lai); a group in the Nha Be and Long Tau rivers; a group in the Soi Rap River; and a group in the Thieng Lieng - Nga Bay River.

Sai Gon Harbour (Figures 4 and 5) was established in 1860. It has been the most important harbour of the country with the largest annual volume of cargo. The harbour consists of four terminals. It has 16 marginal berths (2,793 m in total length) and 25 buoy berths with draft from 3.3 m to 13.5 m. These berths can accommodate a maximum vessel size of 32,000 DWT along wharf and 60,000 DWT at the buoys. Its warehouse and yard systems are quite complete with 68,344 m^2 in 29 sheds and 135,609 m^2 in total area, respectively. Its design handling capacity is 8.0 million tons y^{-1}. Sai Gon Harbour is a general harbour belonging to VINALINE but it can handle other cargoes such as containers and bulk, including rice, fertilizer, cement, and steel. In addition, the Nha Rong terminal has been usually used to receive foreign passenger liners. The cargo throughput volume of the harbour in 1995, 1997, and 1998 in turn was 7.2, 6.8, and 8.34 million tons, respectively. In

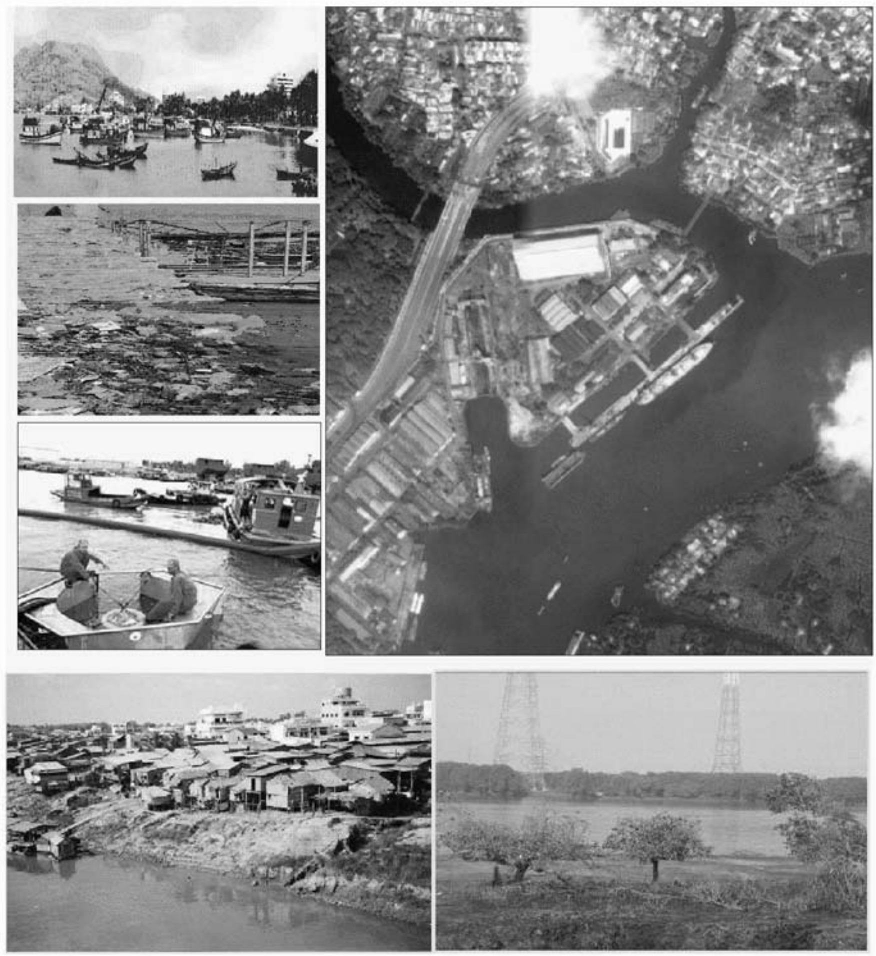


Figure 5. *Examples of human activities and environmental degradation.*

2000, this figure increased to 9.701 million tons (i.e. an increase of 16% from 1999), including containers of 230,000 TEUs (i.e., 38% higher than in 1999). It is forecasted to reach at 11.5 million tons by 2010, so it will be necessary to increase the capacity of terminals. However, there are some disadvantages for this increase such as its support area is very narrow (200 m width in average) and the access channel is not deep enough for big vessels of over 30,000 DWT.

Ben Nghe Harbour was established in 1987. It is under the authority of the Department of Transport and Public Works of HCMC. The harbour is located near Sai Gon Harbour. Its developed area is 22 ha within a total area of 32 ha. At present, this harbour includes 4 marginal berths (816 m in total length) and 7 buoy berths. With draft from 8 to 11 m, it can accommodate vessels up to 30,000 DWT. Its warehouse and yard area is 8,640 m^2 and 51,000 m^2, respectively. This is a general harbour but it can handle containers. The cargo throughput volume was 1.8 million tons in 1995. In 2000, this figure reached 2.77 million tons (a decrease of 3% from 1999), including 111,000 TEUs of containers (an increase of 18% from 1999). Warehouses and yards in the harbour area are being expanded to meet the higher future demand of container throughput.

Tan Cang Harbour was established in 1963. It has been used for military purpose since 1968. It is located at the most upstream side of the harbour group in the Sai Gon River. After year 1975, it was put under control of the Ministry of Defense. However, from 1993 the harbour has been further expanded and used for import and export of commercial commodities. Its design handling capacity is 7 million tons y^{-1}. The harbour has 4 marginal berths with a total length of 1,260 m. Maximum water depth at these wharves is 9.5 m so they are only able to accommodate vessels up to 12,000 DWT. Total land area of harbour is 38.3 ha, of which a warehouse system of 18,786 m^2 and a yard area of 179,000 m^2. Tan Cang Harbour is equipped with specialized facilities for handling container such as transfer cranes and shore cranes. As a result, it is the biggest container handling harbour in Vietnam. The cargo throughput of the harbour in 1995 and 1997 was 3.2 and 4.3 million tons, respectively. In 2000 this figure dropped to 4.08 million tons, including 386,000 TEUs.

VICT (Vietnam International Container Terminal) is located downstream of Ben Nghe Harbour and is under management of the First Logistic Development Co. VICT started operation in November 1998. At that time one jetty berth of 305 m in length and 10 m in draft, and a buoy berth was completed. The harbour can accommodate container vessels up to 20,000 DWT or 1,500 TEU. Two rail-mounted shore gantry container cranes were installed on wharf. In additional, VICT has 4340 m^2 of warehouse space and 80,000 m^2 of container yard area. Its design handling capacity is 1.2 million tons or 150,000 TEUs y^{-1} and maximum container storage capacity is 600,000 TEUs y^{-1}. VICT is designed to meet the requirements of a modern container port. Its layout and operation processes are customized to facilitate a fast handling of containers and a smooth traffic flow. It is also the only terminal in Vietnam so far to utilize a sophisticated computer system to control the harbour operation. Container throughput in 2000 was 129,852 TEUs that comprised 63,500 for import, 56,000 for export and 9,852 TEUs for domestic use. In 1999 the throughput was only 45,000 TEUs in total. VICT has advantage in a good location near Ben Nghe Port, however its development and expansion potentials are limited due to the planned berth length of 580 m and a total land area of 20 ha only.

Tan Thuan Dong Harbour was put into operation in 1994. It is managed and operated by VIETSOVLIGHTER JV Enterprise. It has a 142 m long marginal berth with draft of 9.5 m can receive vessels up to 10,000 DWT. Its design handling capacity is 300,000 tons y^{-1}. The total land area of harbour is about 3 ha, of which the warehouse space is 8,000 m^2, and the yard area is 15,000 m^2. Its expansion potential is very limited because it is squeezed between Ben Nghe Harbour and the Tan Thuan terminal of Sai Gon Port.

The Vegetable Harbour is managed by VEGETRANSCO and has only a marginal berth completed in 1991 with a length of 222 m and a draft of 11.5m. It can accommodate vessels up to 10,000 DWT. In addition, there are three buoy berths for vessels up to 15,000 DWT. The total land area of this harbour is about 6 ha, of which the storage space is about 11,600 m^2, and the yard area is 11,800m^2. The design handling capacity of harbour is 500,000 tons y^{-1}. The cargo throughput of the harbour is increasing annually from 308,000 tons in 1995 to 617,188 tons in 2000. The main cargoes include fertilizer, foodstuff, cement, steel, and other products.

The Lotus (Bong Sen) Harbour started operating in 1994. It is a general harbour and managed and operated by LOTUS JV Enterprise. It has a 150 m long marginal berth with draft of 9.5 m and it can receive 10,000 DWT vessels. Its design handling capacity is 600,000 tons y^{-1}. The total land area of the harbour is about 6 ha, of which the warehouse space is 3,600 m^2. In 1999 the cargo throughput was 302,000 tons.

The Morning Star Harbour was established in 1997. This is a specialized harbour for cement import and export under management and operation of the PHU DONG JV Co. The harbour composes of a 160 m long jetty berth outside and a pier berth for barges inside. That jetty berth can accommodate vessels up to 20,000 DWT. The design handling capacity is 300,000 tons y^{-1}. However the cargo throughput of the port was 41,000 tons and 68,000 tons in 1998 and 1999, respectively.

The Sai Gon Petro Harbour specializes in petroleum import and export and is managed and operated by the SAI GON PETRO Co. It includes 2 berths with a total length of over 300m. The harbour can accommodate tankers up to 25,000 DWT. The cargo throughput was 789,800 tons in 1995 and 1,152,300 tons in 1998. This figure dropped to 927,579 tons in 2000.

The Tan Cang Cat Lai Harbour developed in 1998 a jetty berth in an area between the PETEC Harbour and the Cat Lai Ferry Terminal on the right bank of the Cat Lai River. This jetty berth is 152 m long and 9.5 m in draft. An additional 150 m long berth is under construction. Both berths are designed for handling general cargo. Total land area of new Terminal is about 15.3 ha, of which the warehouse space is 6,000 m^2 and a yard area of 41,000 m^2. Its planned handling capacity is 2.2 million tons y^{-1}. Cargo throughput of this harbour was 156,579 tons in 1999.

The PETEC Cat Lai Oil Harbour was constructed in 1981 for export and import of petroleum products. It is under management and operation of PETEC Company. It had two oil berths with a total length of 115 m and a yard area of 13 ha. It can accommodate tankers up to 25,000 DWT. The recorded cargo throughput was about 900,000 tons in 1995 and 1,100,000 tons in 1998. This figure decreased to 1,019,650 tons in 2000.

The VITAICO Harbour specializes in the import and export of wood chip, and it started operation in 1992. It belongs to VITAICO JV Company. This port has a 175 m long berth and it can accommodate vessels up to 20,000 DWT. Its yard area is 40,000 m^2. Its conveyer's handling capacity is 2,000 tons d^{-1}. The cargo throughput was 110,634 tons in 1995. This figure dropped to 46,319 tons and 55,904 tons in 1998 and 2000, respectively.

The Nha Be Petroleum Harbour is located on the right bank of the Nha Be River. After 1975, it is put under management and operation of the Second Zone PETROLIMEX Co. Its total length of riverbank is about 3 km. There are 5 oil berths with a total length of 740 m, of which two are specialized berths that can serve tankers up to 28,000 DWT, while the other berths are for tankers from 1,000 – 5,000 DWT. Its 140,000 m^2 yard can store 500,000 tons of petroleum at the any time. The operation capacity is 10-12 million tons y^{-1}.

The NAVIOIL Harbour was constructed in 1975. It is located on the right bank and specializes in export and import of vegetable oil. It is managed and operated by Vietnam Vegetables Oil Co. It has a 210 m long berth and two anchor piers. It can serve vessels up to 5,000 DWT. The total land area of harbour is near 5 ha, of which the warehouse space is 11,200 m^2 and the general yard area is about 2 ha. The cargo throughput was 90,508 tons in 1999.

The Phu Dong JV Harbour was constructed in 1998 and is managed by PHU DONG JV Co. It is located on the left bank and specializes in the export and import of wood chip. This port has a 146 m long specialized berth with a conveyer system, which can handle 291 tons of wood chip $hour^{-1}$, and it serves vessels up to 25,000 DWT. Its total land area is 4.8 ha.The design handling capacity is 300,000 tons y^{-1}. The cargo throughput was 68,000 tons in 1999.

The VIKO-WOCHIMEX Wood Chip Harbour was constructed in 1996, belonging to VIKO-WOCHIMEX factory. It has a 180m long berth and two anchor piers can accommodate vessels up to 15,000 DWT. The conveyer's handling capacity is 80 tons h^{-1}. Total land area is about 6 ha, of which general cargo yard is 1.06 ha. The cargo throughput was 8,449 tons in 1996 and reached to 29,125 tons in 1999.

The Dong Nai Harbour is located near the Dong Nai bridge. It is a general harbour under the supervision of the Dong Nai province. This is the biggest harbour in the Bien Hoa area. The total land area is 12.8 ha. Its 60 m long berth can accommodate ships up to 2,000 DWT. It has 1,488 m^2 of warehouse and 51,000 m^2 of general cargo yard. Additionally, it has a buoy berth. The cargo throughput in 1999 and 2000 was 357,754 tons and 404,745 tons, respectively. Of this, almost all (94%) was domestic cargo. The two other harbours are STC GAS-VN and VT GAS, which are smaller in scope and specialized for handling gas.

There are other harbours. In the Nha Be and Long Tau rivers, there are also a number of shipyards such as the Sai Gon South and the Ship Marine (LACOM) shipyards. There are also a number of small harbours and shipyards such as Elf Gas, Bien Dong harbours and Bason, and Sai Gon shipyards. These facilities are located along the right bank of the Sai Gon River and are located between the bigger ports described above.

The Hiep Phuoc Power Plant Harbour was constructed recently to import oil for the Hiep Phuoc thermal power Plant. It has an oil berth with a total length of 300 m and can accommodate tankers up to 40,000 DWT. The design capacity is 530,000 tons y^{-1}.

The Hiep Phuoc Cement Harbour belongs to the Nghi Son Cement Company and is located downstream of the Hiep Phuoc Power Plant Harbour. It is used for import of bulk cement and export of bagged cement. This can accommodate vessels up to 20,000 DWT. The total yard area of harbour is 7.9 ha. Its design capacity is about 900,000 tons y^{-1}.

The Thieng Lieng anchorage Area is located in the Nga Bay River and used to handle dangerous cargo.

5. HARBOURS IN THE TV-VT AREA

The VEDAN Harbour is a specialized harbour to import and export raw materials and products of the VEDAN Company. The port's total land area is 15,500 m^2. It was put into operation in 1994, including two pier berths of 174 m and 186 m in length and 8.5 m – 12 m in draft, respectively. Additionally, it has a buoy berth. They can accommodate the 10,000 DWT dry cargo ships and 12,000 DWT liquid cargo ships. The design handling capacity is 950,000 tons y^{-1} but in 1999 the cargo throughput was 1,500,000 tons.

The Go Dau Harbour, constructed in 1995, is a general cargo harbour. Its main pier berth is 30 m long; it can receive vessels up to 2,000 DWT. It has a lighter pier 20 m long and a buoy berth for vessels up to 10,000 DWT. The total land area is 16.4 ha, including 720 m^2 of warehouse and 7,100 m^2 of open yard. The cargo throughput was 83,787 tons in 1999.

The Go Dau B Harbour comprises two terminals, including Go Dau B1 (72.5 m in length, 8.5 m in depth) and Go Dau B2 (120 m in length, 9.0 m depth). They can accommodate vessels up to 15,000 DWT and are used for handling general cargoes. These terminals are located just downstream of Long Thanh Super Phosphate Harbour and physically limited by the PVC (Unique) Gas Port. Go Dau B Harbour has a 1,052 m^2 warehouse and 24,900 m^2 of open storage.

The Long Thanh Super Phosphate Harbour imports and exports the raw materials and products of the Super Phosphate Factory. It includes two jetty berths with 71 m in total length for ships up to 3,000 DWT. Additionally, it has two buoy berths. The total land area is 1.25 ha, of which the warehouse space is 1,170 m^2. The cargo handling capacity is 100 tons h^{-1} and the cargo throughput in 1999 was 63,141 tons.

The Phu My Power Plant Harbour is a specialized harbour to supply oil and under operation of Phu My Thermal Power Plant. It was completed in 1997, including one oil berth with the length of 175 m and can accommodate tankers up to 10,000 DWT. The oil throughput of this harbour was about 170,000 tons in 1999.

The Baria Serece Harbour is a general harbour under control of the Baria Serece JV Co. There are 8 wharves (804 m in total length) and a buoy berth that can accommodate vessels up to 30,000 DWT. The harbour land area is 12.8 ha, of which about 26,000 m^2 is a general and container yard, and 13,000m^2 is for warehouses. The design handling capacity is 1,6 million tons y^{-1}. The cargo throughput was 830,000 tons in 1999 and 888,000 tons in 2000. The main cargoes are fertilizer, minerals, and agricultural products.

The Cai Mep LPG Harbour is for import and export of LPG belonging to PetroVietnam. It includes two special berths with total length of 400 m and it can accommodate LPG tankers up to 30,000 DWT. The total land area is 40 ha on which a LPG storage facility and a conduit system were constructed. A 25 km-long gas pipeline was constructed linking this harbour with the Dinh Co Gas Processing Plant. It receives and processes gas from fields offshore Vung Tau. The gas throughput of this harbour was 249,000 tons in 2000.

Besides above ports, several deep-water harbour projects have been planned in this area such as Cai Mep general and container cargo harbours and the harbours of Sai Gon Petro, PETEC, TOTAL, HOAN NGUYEN Steel Co. The PVC harbour and VINAFOOD II harbours are presently under construction

The Oil K2 Harbour is a specialized harbour to import oil and petrol for the Baria– Vung Tau Petroleum Co. It includes one 82 m oil berth and can receive tankers up to 5,000 DWT. It can store 9,000 m^3 of oil product; the design handling capacity is 150,000 tons y^{-1}. The cargo throughput was 52,000 tons in 1999.

The VIETSOVPETRO Harbour is a specialized harbour for oil and gas services operated by Vietsovpetro JV Co. It now had 10 berths with total length of 1,377 m and a draft of 5-8.5 m. It can accommodate ships up to 10,000 DWT. The land area is 53 ha and its warehouse and open storage areas are 20,000 m^2 and 74,300 m^2, respectively. The cargo throughput of harbour was 534,000 tons in 1999

The PTSC Harbour has three terminals belonging to PTSC of PetroVietnam. The upstream terminal is specialized for oil and gas. It is operational since 1992 and under control of the Harbour Enterprise of PTSC. This terminal includes a 120 m long berth with a draft of 6 m, and it can receive vessels less than 10,000 DWT. Its land area is 21.5 ha, of which warehouse space is 7,600 m^2. The downstream terminal is also specialized for oil and gas. It is operational since 1997. This terminal includes 7 berths with a total length of 460 m and a draft of 6 – 9 m, and it can accommodate vessels up to 10,000 DWT. Its land area is 53 ha, of which the warehouse space is 12,000 m^2 and the general yard is 140,000 m^2. The handling capacity is 800 tons d^{-1}. The PTSC oil terminal is specialized for import and export of oil. It was put into operation in 1993 under the control of the Fuel, Material and Equipment Service Enterprise of PTSC. It includes a pier berth of 156 m in length and 9 m in draft, and it can accommodate tankers up to 10,000 DWT.

Apart from above ports, along the Dinh River and the Vung Tau Peninsula area, there are some other small harbours such as Truong Sa Seafood Co. of the Ministry of Defense and the Trade and Cat Lo harbours belonging to Vieco Co., etc. In Vung Tau offshore, there are also some non-berth oil exporting stations such as at Bach Ho (White Tiger) and Dai Hung (Greater Bear) Oil Fields. These stations are under supervision of PetroVietnam.

6. ACCESS CHANNELS TO HARBOURS

There are two main channels used by ships to sail from Ganh Rai Bay to the harbours in HCMC and TV-VT area; these are the Long Tau - Sai Gon River Channel, and the Vung Tau – Thi Vai River Channel.

Apart from these channels, small ships can also use the Soi Rap River channel; over one hundred years ago, this route was used as the main channel from the sea to Sai Gon port. This channel has some advantages such as a wide area and a large curvature of meanders. However, there are a number of shoals in the estuary. The average depth of the estuary is 5 m. Currently, since no survey has been carried out for a long time and without any maintenance of aids to navigation, the Soai Rap River channel cannot be used by large ships. As a result, the Long Tau River channel became the official route for vessels sailing from the Sea to harbour in HCMC.

6.1. The Long Tau – Sai Gon River Channel

Vessels calling in HCMC harbour from the sea have to pass the Vung Tau Cape, Ganh Rai Bay and the river sections of Long Tau, Nga Bay, Nha Be and Sai Gon. The length of this river channel is 82 km. This channel is narrow but rather deep and stable. Navigation aids including lighthouses, light buoys and beacons were used since 1920, but very simply. Since 1975, the Vietnam National Maritime Bureau has endeavoured to improve the channel condition and navigation aids. Channel maintenance works have been carried out since 1992; the channel depth was increased from 8 m to 8.5 m. Navigation aids were added more adequately. Vessels up to 25,000 DWT can readily sail in both directions.

6.2. The Vung Tau – Thi Vai River Channel

This channel includes the Vung Tau Access Channel (11.5 km long, 200 m wide; minimum depth of 8.1 m) and the Thi Vai Access Channel (42 km long, 100-150m wide, minimum depth of 7 to 10.5 m).

7. DISCUSSIONS

Most of the harbours in the Sai Gon and Nha Be rivers were constructed a long time ago and in curved river sections. Therefore, the water depth and the turning radius restrict the use of large ships. In additional, their back-up areas inshore are limited; the handling equipment is old and under-capacity. As a result these harbours do not meet the growing cargo throughput demands, especially for container traffic. Many harbours have to run at full capacity and several have expanded in the past years. Their capacities still do not meet the demand forecast.

The TV-VT area has a very big potential to develop deep-water ports. However industry there is not yet fully developed, so the throughput is still low. Almost all the new harbours in the TV-VT area are for specialised business activities of specific owners.

7.1 The quality of water and bottom mud

The wastes of almost harbours are directly discharged to rivers and the sea without, or with minimal treatment. The Access channels to harbours in HCMC and TV-VT area are narrow, winding, and shallow. This fact, together with the rapid changes of tidal levels and currents, is one of main reasons of accidents and the resulting environmental degradation.

The quality of water and bottom sediment in the open sea is quite good; in coastal waters and in estuaries there are serious problems. The measured data used in this paper were published by local government offices and a handful of scientific reports (HCMC's Department of Science, Technology and Environment (DOSTE), 2000; Ba Ria-Vung Tau Province DOSTE, 2000; Sy, 2005; Hens, 2005; Nhan, 2003). The information is sketchy because environmental research is still minimal, but it describes an alarming picture.

7.2 The pollutant sources:

The completed studies (HCMC's and Ba Ria-Vung Tau province DOSTE, 2000; Nhan et al., 2003) show that the water quality in the HCMC and TV-VT areas is critical low. The statistical data is impressive:

- Domestic sewage: The sewage discharge in HCMC is 388,932 m^3 d^{-1}. The pollutant loading is 240,687 kg d^{-1} of BOD5, 423,049 kg d^{-1} of COD5, and 522,590 kg d^{-1} of suspended solids. The waste discharge in the TV-VT area equals 8-10% that of HCMC. All water waste directly flows into the Sai Gon and ThiVai rivers.
- Industrial sewage: The industrial sewage of HCMC is 75,000 m^3 d^{-1} and TV-VT area is 30,000 m^3 d^{-1}. The pollutant loading is: 14,385 kg d^{-1} of BOD5, 33,495 kg d^{-1} of COD5, and 23,310 kg d^{-1} of suspended solids. Almost all this waste water is discharged to the Sai Gon Dong Nai, Nha Be, Soi Rap, and Thi Vai rivers
- Agricultural sewage: Insecticide used in HCMC and TV-VT area amounts to 190,000 tons y^{-1}.
- Oil spills: from 1992 to 2004, there were 12 oil spills with volume more 100 tons, and many more smaller oil spills. One of these spills in Cat Lai Harbour was catastrophic: 1,700 tons of oil was spilled in Sai Gon –Dong Nai River. Almost oil spills happened in HCMC and TV-VT area. Also, the sewage from harbour network and access channels (including oil liquid and solid) is increasing daily. Furthermore, the "sneaky" discharge of oil from numerous small vessels and from oil storage facilities, workshops, and petrol stations is leading to a chronic pollution problem:
- Aquaculture activities: Discharge from shrimp and fishponds and their processing areas spread bacteria, virus, organics, and eutrophication substances. There are no data on these pollutant sources.
- Land fills and engineering constructions: Buildings and hydrotechnical constructions, land fills, waste dumping, and canalizing occur at large scale. This results in inhibiting, or even blocking, water flows, changing inundation regime, and spreading pollutants. This impacts on ecosystems.

The policies needed to manage these pollutant sources include:

- Establishing treatment facilities for water and solid wastes from urban and industrial areas, shipping routes, and aquaculture activities.
- Building water supply schemes for all urban areas.
- Create a long-term program of environmental protection.
- Monitor water, mud and soil quality.
- Complete an environmental impact assessment for all developments.

7.3 Pollution in the Sai Gon Harbour group

The water and sediment in Sai Gon Harbour group are critically polluted, mainly from domestic sewage and industrial wastewater because almost all are discharged into the river network without treatment. The water and sediment quality in main river branches Dong Nai, Nha Be, Long Tau is less serious, except the Sai Gon River, where the Sai Gon Harbour group is located.

According to JICA (1999), the classification of water pollution in the Sai Gon River is as follows:

- Serious pollution by suspended solids (20-100 mg l^{-1}), colour (black, turbid), odour (stinking), organic (BOD5: 10-35 mg l^{-1}, DO: 3-5 mg l^{-1}), bacteria (fecal coliforms: 10-95 MPN/100 ml), oil (mineral hydrocarbons: 0.01-0.1mg l^{-1}), and eutrophication (Total N: 0.3-3 mg l^{-1}, Total P: 0.03-1.8 mg l^{-1}).
- Furthermore, the status of oil pollution from waterway transportation has become a major problem from the “sneaky” discharge of oil from vessels and boats, and from oil spills from storage facilities and petrol stations.
- Moderate pollution by heavy metal (Pb: 0.002-0.009 mg l^{-1}; Cd: <0.002 mg l^{-1}; Hg: 0.002-0.009 mg l^{-1}; Cu: 0.001-0.009 mg l^{-1})
- Light pollution by toxic chemicals and agriculture pesticides.

The harbours located along the Dong Nai River are less polluted: BOD5: 4-10 mg l^{-1}; DO: 5-7mg l^{-1}; suspended solids: 2-15 mg l^{-1}; Total N: 0.3-0.7 mg l^{-1}; Total P: 0.01-0.07 mg l^{-1}; Cu < 0.002 mg l^{-1}; Cd < 0.001 mg l^{-1}; Pb < 0.002 mg l^{-1}, Hg <0.002 mg l^{-1}; Coliform: 15-90 MPN/100 ml.

The harbours located along the Nha Be River are moderately polluted: BOD5: 4-15 mg l^{-1}; DO: 7-8mg l^{-1}; suspended solids: 2-80 mg l^{-1}; Total N: 0.4-0.9 mg l^{-1}; Total P: 0.02-0.3 mg l^{-1}; Cu < 0.005 mg l^{-1}; Cd < 0.002 mg l^{-1}; Pb < 0.006 mg l^{-1}, Hg < 0.005 mg l^{-1}; Coliform: 5-15 MPN/100 ml.

The water in harbours located along the canals of Tan Hoa-Lo Gom, Tau Hu–Doi-Te-Ben Nghe, Nhieu Loc-Thi Nghe are polluted to a maximal level: black in colour; very stinky odour (since high concentration of CH_4 and H_2S); 0-l mg l^{-1}; 50-1000 mg l^{-1}; Coliforms: 1,5 x 10^2- 10^8 MPN/100 ml; suspended solids: 20-300 mg l^{-1}; Pb: 0.02-0.3 mg l^{-1}. The waters here look like as black condensate. This is ultimate limit of possible pollution.

According to study of UNDP (1998), the quality of surficial sediment in the Sai Gon Harbour group is very poor: pH: 7.8; BOD5: 4.5 %; COD: 5.1%; Total N: 0.23%; Total P: 0.04%; Oil: 0.01%; Cu: 0.04 mg l^{-1}; Zn: 1.54 mg l^{-1}; Cd <0.04 mg l^{-1}; Hg: 0.64 mg l^{-1}; Pb: 0.04 mg l^{-1}.

7.4 Pollution in harbours in the Vung Tau area

The study by the Baria-Vung Tau Province shows that all shipping wastes and domestic wastes are discharged directly in harbour waters without treatment. At locations with a high density of fishing boats, the water is polluted by oil, organics, and coliforms. The data describing the water quality in the Vung Tau harbour group can be summarized as follows:

- The Organic pollution: at Ben Dinh: DO: 2.6 mg l^{-1}, COD: 37 mg l^{-1}; at Phuoc Tinh: DO: 3 mg l^{-1}, COD: 63 mg l^{-1}; at Binh Chau DO: 3.7 mg l^{-1}, COD: 22 mg l^{-1}; at Cat Lo: DO < 4 mg l^{-1}, COD: 10 mg l^{-1}; at Ba Ria – Sereces, at Phuoc Hoa: DO > 6 mg l^{-1}; COD < 5 mg l^{-1}. The high level of organics here is due to domestic shipping routes and wastes from fish markets.
- The eutrophication pollution (Total N, mg l^{-1}): at Ben Dinh> 4.8; at Binh Chau > 2.9; at Phuoc Tinh > 4.2; at Cat Lo >2.8; at Ba Ria-Sereces < 0.4. The high level of eutrophication is due to waste from fishing boats and fish markets.
- The Oil pollution (mg l^{-1}): at Ben Dinh: 0.7; at Vietsovpetro: 0.4, at Hai San company: 0.4, at Phuoc Tinh: 0.12, at Binh Chau: 0.15. The high level of oil pollution is due to wastes from ships, small vessels, and harbours.
- The coliforms in the harbours of Cat Lo, Ben Dinh, Binh Chau, Phuoc Tinh, and Sea Food Truongsa are 8,500 – 38,000 MPN/100 ml, much higher than acceptable standards. This high level of coliform is due to ships and wastes from ships markets.

7.5 Pollution in harbours in Thi Vai River

According to report of HCMC Environmental Protection Agency (HEPA), the quality of water and sediment in harbours in the Thi Vai River is extremely poor.

Water quality parameters are as follows:

- Salinity: in the rainy season: 19- 25; in the dry season: 25- 30.
- Colour: The upstream part of Thi Vai River began to change in 1993 from blue to black. At present, the waters all along the Thi Vai River are black, except the river part connected with Ganh Rai Bay.
- Odour is stinking.
- pH is in range 6.1 – 8.1, less than pH of standard marine water, especially in Go Dau and Vedan harbours.

- Suspended solids concentration: 20 – 163 mg l^{-1}.
- Total P: 0.032 – 0.111 mg l^{-1}.
- Organic pollution: DO in Go Dau and Vedan harbours is near 0 and in Ganh Bay is 1 – 6.1 mg l^{-1}. DO in 2005 year is less than in 2003. In Ganh Rai Bay, BOD5 is very high (8.6 – 26.4 mg l^{-1} in Ganh Rai Bay) and extremely high (30-40 mg l^{-1}) in the Go Dau and Vedan harbours.

Surficial sediment quality data are as follows:

- Cyanide (CN) < 0.01 mg kg^{-1}.
- Heavy metal: Cd: 0.9 – 9.5 mg kg^{-1} (very high) in Go Dau and Vedan harbours; Pb, Cr, As have concentration less than Viet Nam allowable standards; Hg is in range of 0.3–1.3 mg kg^{-1} (very high).

The black colour, the stinking odour, the low pH, and the alarming organic pollution of water in Thi Vai river has many causes, including waste from the Vedan sodium glutamate plant and another plants in the industrial zones of Go Dau and Phu My. The polluted water here is poorly flushed because in the dry season the Thi Vai River has almost zero freshwater flow and because tidal flushing is ineffective. The velocity changes direction 4 times daily and the pollutants are largely transported back and forth and slowly diffused. Pollution from the Thi Vai River also degrades water quality in Ganh Bay and CGMBR, and this influence is increasing daily.

The large scale pollution by heavy metals in the sediments of the Thi Vai harbour group, Thi Vai basin, and Ganh Ray Bay is apparently caused by wastes from the industrial zones on left bank of the Thi Vai River, especially from Go Dau and Phu My zones. The concentration of Cd and Hg is much higher than the Vietnam standards. In particular, the Cd and Hg levels in the sediments of Go Dau and Vedan harbours are alarming.

7.6 Recommendation:

There is an urgent need for

- monitoring the environmental status of water and sediments in the Sai Gon, Dong Nai, Nha be, and Thi Vai rivers, CGMBR and Ganh Rai Bay.
- finding solutions and enforcing regulations for the management of wastes from domestic users, industrial zones, shipping routes, and aquaculture activities.
- quickly improving the infrastructure for collecting and treating wastes from domestic use, industrial zones, shipping routes and aquaculture activities.
- environmental management plans for all river basins, Ganh Rai Bay, the coastal zone and harbours in SFEZ.
- creating and enforcing regulations for responding to environmental accidents.

8. ENVIRONMENT OF CGMBR, CHALLENGES AND POLICY

8. 1 The forest and socio-economic quality of CGMBR

Previously, CGMBR was a biodiversity-rich environment. From 1964-1970, the Can Gio mangroves were heavily sprayed with herbicides, including 665,666 gallons of Agent Orange, 343,385 gallons of Agent White, and 49,200 gallons of Agent Blue. As a result, 57% of the mangrove forest in this district was destroyed (Ross, 1975). In some areas the herbicide spraying killed large trees of Rhizophora, Sonneratia, and Bruguiera and in many areas the vegetation was completely destroyed. Only *Avicennia* and nipa palms were able to survive and regenerate after the application of herbicides. New species, such as *Phoenix paludosa* and

Acrostichum aureum, a fern that presently dominates elevated land, have expanded. Some trees of *Avicennia officinalis* and *Excoecaria agallocha* are now found only as shrubs. After many years of chemical spraying, the degraded land still has only scattered small trees of *Avicennia, Ceriops, Lumnitzera, Thesphesia, Pluchea,* or *Sesuvium portulacastrum* and *Paspalum vaginatum*. Since 1978, a vast program of reforestation has been undertaken by HCMC with the main species being *Rhizophora apiculata*. Up to now, the reforestation effort has brought vast ecological improvements to the environment. Wild animals such as monkeys, otters, pythons, wild boars, crocodiles, and various kinds of birds have returned to the artificially regenerated mangrove forests. Since 1991, the Can Gio mangrove forest has been declared an "Environmental Protection Forest" of Vietnam. It is today one of the largest most extensive rehabilitated mangroves in the world.

This biosphere reserve includes both salt water and brackish water species. The CGMBR have a high biodiversity with more than 200 species of fauna and 52 species of flora. Can Gio forest is home to: 157 plant species; 130 kinds of algae; 137 species of fish; 9 species of amphibians; 145 species of birds; more than 700 monkeys; 80 crocodiles; 51 species of waterfowl; 4 species of mammals; mangroves with saline water species such as *Sonneratia alba*, *Avicennia alba*, mixed communities of *Rhizophora apiculata* - *Sonneratia alba*, as well as *Xylocarpus granatum*, *Kandelia candel*, *Rhizophora mucronata* etc., and with brackish water species such as *Sonneratia caseolaris*, communities of *Cryptocoryne ciliate, Acanthus ebrateatus*, *Nypa fruticans*, and *Acrostichum aureum*; seagrass beds dominated by *Halophyla* sp., *Halodule* sp. and *Thalassia* sp.; agricultural crop land with rice, taro/yam, beans and coconut; plantations; urban areas.

One of the main advantages of Can Gio is that it provides the opportunity to work on environmental protection on a continuum of habitats, ranging from the sea to the boundary of HCMC. The mangrove forest is regarded as the “green lungs” of the city. There are some 58,000 people living within the biosphere reserve boundaries, 54,000 of which live in the transition area. The local people are of different origins and there is a mixture of cultures and ways of life. The main economic activities are agriculture, fisheries, aquaculture, and salt production. Some families have been allocated with forests for protection for 30 years and they use a small portion of the land for aquaculture and salt production. Other families, engaged in miscellaneous occupations, have no land, and must earn their living by catching crabs and mollusces and collecting firewood.

Can Gio is the poorest district of HCMC. It is expected that the CGMBR could be a site where sustainable development, conservation, and cultural socio-economic activities in sylvo-forestry and fishery management systems can be tested, refined, demonstrated and implemented.

The oldest mangrove plantation in Can Gio is 14 years old. The mean annual increments of diameter at breast height and height over the period are 0.61 cm and 0.81 m respectively. These vary with soil types and their degree of inundation by tides. *Rhizophora apiculata* in Can Gio has multi-stems, the biggest stem is considered as the main stem and the remainder as sub-stems. A final harvest volume at 20 years age of about 100-110 m^3 ha^{-1} is expected. The majority of this will be used for poles and construction timber and the remainder as fuel wood.

Together with the products from the three thinning operations reported later, amounting to about 3346 m^3, the total wood production can be estimated at approximately 140-150 m^3 ha^{-1} or about 7 m^3 ha^{-1} annually on average. A longer rotation may be considered for some stands in order to obtain timber of bigger size and to increase the average annual production. Forestry managers evaluated the total economic value of CGMBR as US$ 212 million (in 2000 year price).

Traditionally, fishery activities have been a main source of income for the people of Can Gio, but due to the limited resources it is difficult to sustain these unless better ways of exploitation and a more rational use are adopted. Every year, about 11,500 tons of fish and shrimp were captured in Can Gio, of which 986 tons were shrimps. The local people often capture fish and crabs with simple tools, e.g. hooks, fishing poles, and cast nets; hence there is a low catch per person. Generally, the fishermen of Can Gio District are poor and have a low standard of living.

There are 20,418 ha open water area, of which 7,000 - 8,000 ha has potential for shrimp breeding. The traditional shrimp culture, using trapping ponds, covers about 3,640 ha and is practiced by local people and state farms. It involves creating ponds by damming natural waterways within the forest, allowing shrimp juveniles to come into the ponds during high tide, and later harvesting them once or twice a month. One "pond" will usually cover 5-10 ha of mangrove area. Production is low (about 50-150 kg ha^{-1} y^{-1}). Although this traditional method is the least damaging to the mangrove forest, it may threaten the surrounding mangroves, because it often involves illegal logging of *Rhizophora* wood for use as poles and piling foundations in the ponds. Most private ponds are semi-extensive with dikes established around the pond and sluices made of a light material such as wooden planks or poles. Additional post-larvae are supplied as well as supplemental feed. The average yield was 150-250 kg ha^{-1} y^{-1} in 1992.

Semi-intensive shrimp ponds occupied an area of 360 ha in 1991, and this increased by 100 ha y^{-1}. These ponds are generally smaller, about 0.5-1 ha, with mechanical pumping of water, nursery- raised larvae, feeding, and require high investments. An area of 120 ha is operating with a production of 42 tons shrimp; i.e., an average yield of 300 kg ha^{-1} y^{-1}. The ponds were established either on *Phoenix paludosa* land, which is elevated land that could not be planted with *Rhizophora*, or on abandoned salt fields with stunted *Avicennia lanata* growing on it.

For state-owned ponds, the total water area reserved for shrimp breeding (for 25 state farms and production units) is 2,586 ha, of which only 120 ha are in use for semi-intensive and intensive farming (Can Gio People's Committee, 1992) with a yield of 300-450 kg ha^{-1} y^{-1}. The remaining area consists of semi-extensive ponds with dikes established around them. The initial yield was encouraging (150 -250 kg ha^{-1} y^{-1}) but has declined to only about 50 kg ha^{-1} y^{-1} at present (Dinh et al., 1990). There are two causes for this decline, namely insufficient inputs of (natural) feed and larvae and poor water quality.

As indicated above there is an urgent need for systems that integrate shrimp farming with tree growing in order to achieve a more sustainable land use. For these systems, as well as for the mangrove area as a whole, a ratio of 80% forest to 20% shrimp farms and other land uses is recommended (Quynh and The, 1992). Such systems have already been practiced for some time in Indonesia and have also been introduced in Minh Hai province, south of Can Gio with initial promising results (Liem, 1992). There are three principally different systems applied in Minh Hai:

1. Mangrove plantation forest (on 20 year rotation) where fish and shrimp are caught within natural canals. No ponds or dikes are constructed.
2. Each household manages 8 ha of forest area on a 12 year rotation surrounding 2 ha of permanent shrimp ponds. The farmer gets all shrimp products and shares forestry products with the government enterprise.
3. Forest established within dikes where shrimp can be raised for about 5 years until canopy closure. Hereafter a new area must be used for shrimp breeding.

The choice of the most suitable system will thus depend on the specific local situation in terms of topography, tidal regime, and socio-economic conditions. These precautions should be incorporated into the overall mangrove management plan.

Apart from shrimp culture, crab culture in ponds and floating cages is also becoming more common among people in Can Gio. In 1991, 140 households raised soft shell crabs and 78 households raised red pancreas and other crabs in about 180 floating cages. Nine households raised spiny lobsters in 15 cages. As indicated by the initial results, floating cage culture can bring high economic return if a number of problems can be solved, including planning of breeding, culture techniques, environmental protection, marketing and financial support for investments. One advantage of the cage culture is that it does not require the conversion of mangrove forest as pond culture does. It may thus serve as a more environment-friendly production system, which can act either as an alternative or a supplement to shrimp farming.

The central and local governments already undertake a number of activities for the environmental protection of this reserve because the importance of a mangrove forest is much wider than just being a site for wood production. The mangrove resources are numerous and of great importance to the socio-economy of the Can Gio District, providing both forestry and marine products. In addition, they also have an important role in coastline protection and accretion of land, improvement of the microclimate, as well as being an important nursery ground and habitat for many marine prawns, fish, and other animals. With its new status as an environmental protection forest, more recognition is given to the more indirect beneficial roles of the mangrove forest. This change of emphasis on objectives should have a considerable impact on the management of the mangrove forests. Following a study to determine the feasibility of converting a production forest into an environmental protection forest the Council of Ministers approved the conversion in Decree No. 173 CT/HDBT, dated May 29, 1991. At the total cost of 500 million Dong (from the HCMC budget) 1,465 ha of *Rhizophora apiculata*, 101 ha of *Nypa fruticans* and 94 ha of *Eucalyptus camaldulensis* were planted and manpower and building support provided. About 86 households were allocated a total of 7,173 ha of land and forest land to maintain, protect, and afforest. They were allowed to plant trees on their land and gather all products. To establish the forest, a contract period of 30 years was agreed between volunteer households and the Can Gio Forest Enterprise. The forest produce is shared between the Forest Enterprise (35%) and the households (65%). People can also make shrimp ponds within the forest, under the control of, and with technical support from, the Forest Enterprise and with the approval of the HCMC Forest Department. In order to preserve the CGMBR, the HCMC Forest Department is now controlling all the *Avicennia alba* grown on mud flats. The riverbanks must be retained as a buffer area, 5-10 m wide. People can still collect dry wood from within these forests for their own use.

Some laws and Regulations for CGMBR were promulgated as following:

- Regulations for first and second thinning operations were also issued in 1984 and 1989 respectively, while the regulation for a third thinning is still being compiled;
- The hunting of wild animals and birds in the mangrove area is now prohibited, according to official rules No. 1222/NN-LN dated 15 November 1991, issued by the HCMC Agriculture Service;
- To manage shrimp ponds within the forest, the HCMC Agricultural Service issued a communication on management and establishment of shrimp ponds within the CGMBR. In the past, many state farms, forest enterprises, and individuals in Can Gio cut trees, and build dams and levees indiscriminately for shrimp ponds, thus affecting the growth and development of *Rhizophora*. To avoid this, the establishment of shrimp ponds is now strictly controlled by forest rangers and the Forest Enterprise;
- A temporary regulation (No. 178/LN-QD) dated 7 March 1992 of the HCMC Agriculture Service concerns contracts for hiring of households and management units for individual plots for forest protection. Based on these contracts, the money for paying forest guards and households will be made available from the HCMC budget every year.
- In the future, further regulations and guidelines for the use and protection of these forests can be expected to enable ecologically sustainable socio-economic growth.

8.2 Pollution challenges in CGMBR

In recent decades, the mangrove forests in study area have been affected by many detrimental changes in extent, composition, and actual forest quality. These changes are due to various causes but the principal ones are those resulting from human activities and climate-related impacts. Scientists have warned of growing reduction in biological diversity in the CGMBR, calling on relevant agencies to take urgent measures to improve the situation. The situation is serious:

- In 2004, 25.3 ha out of 30,079 ha of the Can Gio mangrove forest were lost, and 60% of the planted mangroves perished (reported by the HCMC Forestry Development Sub-department at a recent seminar on the management, use and sustainable development of the Can Gio forest, 2005) by urbanization, building industrial zones and harbours in HCMC and TV-VT area.
- The construction of a road has blocked tidal waters from flowing in and out of parts of the Can Gio forest, causing the loss of many trees.

- Shrimp farms, which cover nearly 3,000 ha of the forest, have destroyed parts of this ecosystem.
- The construction dykes and embankments in mangrove forests for shrimp ponds and agriculture inhibits tidal flushing.
- The building of dams for reservoirs and hydrological plants reduces the flow of river water resulting in increased salt water intrusion. Natural changes, accompanying incursions of harmful insects and depleted soil, access channel dredge are other reasons for the decrease of biodiversity in this world-class ecological site.
- The construction of ports, industrial zones, and transport infrastructure in Thi Vai river basin as well as aquaculture have caused adverse impacts on mangrove ecosystems, which were already severely degraded during the war and not yet fully rehabilitated. The environmental impacts include: (1) Loss in mangrove forest area; (2) decrease tidal flushing, decrease water depth and time of tidal inundation, degrading soil properties; (3) acid soils will increase; (4) organic and solid waste pollution is increasing.
- The global climate change may also impact the area. Sea level may rise by up to 1.5 mm y^{-1} and this may accelerate the speed of coastal erosion that destroys mangroves. This erosion also destroys the shelter of a great many tidal and forest animals as well as the spawning grounds of some fish and shrimp species. This sea level rise promotes the invasion of mangroves into farming areas. It may also prevent soil accumulation so that tidal flats become more deeply flooded; this may hinder the development of pioneer mangrove communities like *Avicennia alba* and *Sonneratia alba* near river mouths because their pneumatophores can be drowned.

8.3 Policy of the protection and use of CGMBR

In order to further improve the interactions between all sectors and to enable the development of more successful strategies, the following policy and development issues should be addressed:

- Promoting Public Awareness: The protection and use of the mangrove ecosystem requires an intimate understanding of the forest and the environment that people live in. In the past, extension programs, training activities, seminars, workshops and exhibitions on the mangrove ecosystem have received low priorities. The lack of awareness of the importance of the mangrove forests coupled with the poor state of the economy has led to damage of the forest. To win public acceptance and support in Vietnam at the local level for forestry programs, social and community forestry projects should be carried out to promote awareness of conservation issues and mobilize the local people to participate in mangrove forest protection. Public campaigns and educational activities should be undertaken to make people more aware of the direct and indirect importance of mangrove forests, and of laws and regulations for the forest, and to provide guidance on forestry and fishery techniques, including aquaculture. This awareness campaign should also be directed towards decision makers outside the mangrove area.

- Land Allocation: In the past, land and forest land was allocated only to state organizations, not to local people or individuals. Hence, conflicts of interest arose frequently between state organizations and authorities such as state farms, Forest Enterprises, and the local people. Mangrove forests were damaged, illegal cutting was common, and the state farms were not able to protect the forest. The local people, who lived in or nearby the forests, had no legal benefits from the forest, not even firewood for cooking. The HCMC Forest Department has initiated a change in policy by allocating land and forest land to local people for forest protection, maintenance, and afforestation. Besides allocating land, other support is also provided such as financial support for house construction, to purchase boats and rice during the first 6 months, as well as tree seeds and assistance and supervision of their forest-related activities. This has resulted in better protection of the forest. The forestry activities provide jobs for people through forest protection, thinning, and afforestation. If these people plant trees on the allocated land, they become the legal owners of these trees, and hence they behave more responsibly. For various reasons, such as limited economic resources to support households in the initial period after the land has been allocated to them, land allocation is presently carried out very slowly.
- Land Use Planning: In order to ensure the overall sustainability of the mangrove area, including permanent and productive forest lands, agricultural land, shrimp ponds, fish farms, rivers etc. a land use plan specifying zones for each land use category should be drawn up. This will aid in promoting cooperation between diverse economic interest groups as well as ensuring environmental stability. This plan should form the basis of the land allocation programme. In order to ensure a sustainable system the government should naturally include privately owned lands as well.
- Improving Standard of Living: Most people who live in the CGMBR are poor and live under harsh conditions such as lack of fresh water, education, and communication, few job opportunities, and the risk of malaria. As a means of raising the standard of living in the mangrove forest areas, the government should give a high priority to the provision of infrastructure like roads, hospitals, schools, and electricity. In addition, the government should try to provide opportunities for job-generating activities, for instance by offering low interest loans for investments in charcoal kilns, aquaculture, livestock breeding, agriculture, and tree planting. In particular, there appears to be ample room for a sustainable expansion of the local charcoal industry.
- Shrimp Breeding: A thorough evaluation should be made of all semi-intensive and intensive shrimp ponds already present in Can Gio. This is considered necessary in order to draw up policies and guidelines. Clear cutting of mangrove forests for conversion into shrimp ponds is now

strictly forbidden and this rule may have to be amended. Although semi-intensive farming appears to be the best model, many improvements in management and techniques still have to be introduced to prevent the collapse of the mangrove forest ecosystem.

- Energy Policies: Although they live in the mangrove area, people often lack firewood for domestic use. This is because most forests already have owners or because the area has been designated an environmental protection forest, hence with restricted access. People are only permitted to collect dry wood and, because this amount is limited, this often leads to illegal cutting. This calls for a solution to the energy problems facing the people. This can be done through the promotion of afforestation, by planting trees in homesteads, around schools, along roads and riverbanks. This may increase the local supply and, together with the promotion of fuelwood-saving cooking stoves, would reduce illegal cutting.
- It is necessary to legislate the requirement for environmental impact assessments for all development projects such ports, industrial zones, and roads. Inter-sectoral collaboration should be promoted in planning land-use management.
- Unfortunately, research on mangrove forests has until now been given low priority by the authorities. Even though some research has been carried out during the past 10 years, e.g. in the fields of species survey, thinning operations, charcoal processing and silvofishery, this has often been done in isolation and with little sharing of the results among all concerned. Furthermore, many issues remain, such as: species choice for replanting on elevated land and on abandoned salt fields; planting trials with measurement of erosion along riverbanks and the coastline; relations between mangrove forest and fishery; primary and secondary productivity; surveying wild fauna to set up a wildlife conservation scheme; possible environmental impacts of ecotourism; and economic prospect of alternative mangrove products. There is therefore a need for collaboration between the concerned research agencies, to ensure that these activities will collectively produce the optimum benefits and avoid wasting of scarce resources.

9. EROSION AND DEPOSITION

There are many erosion sites in HCMC, and these increase yearly. The massive erosion of river beaches in the Districts of Binh Thanh and Nha Be along the Sai Gon and Muong Chuoi rivers threaten HCMC. There are many seasons for this erosion, all appear due to human activities through land filling, shipping, and building of dykes, shrimp ponds and other infrastructures. There are some temporal measures in place to combat erosion. There are almost no detailed studies to find needed long-term measures. Presumably the severity of this problem will grow in time. This is also suggested by the estimates of erosion/deposition patterns in Ganh Rai Bay due to shipping activities as shown in Figure 6 (HAECON, 2001).

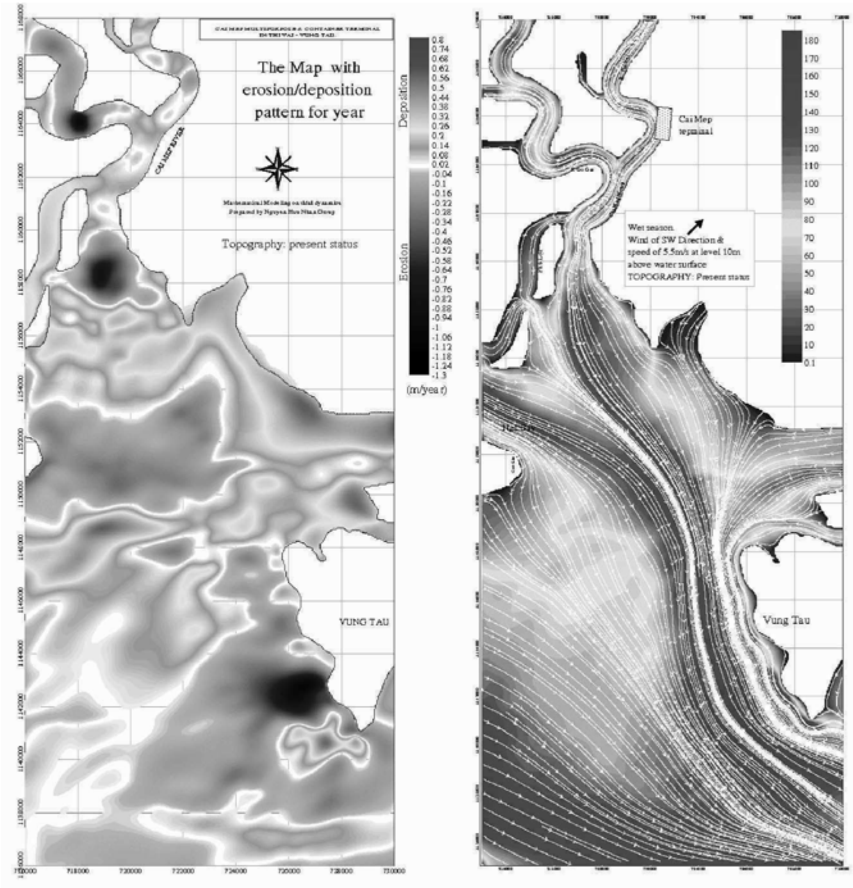


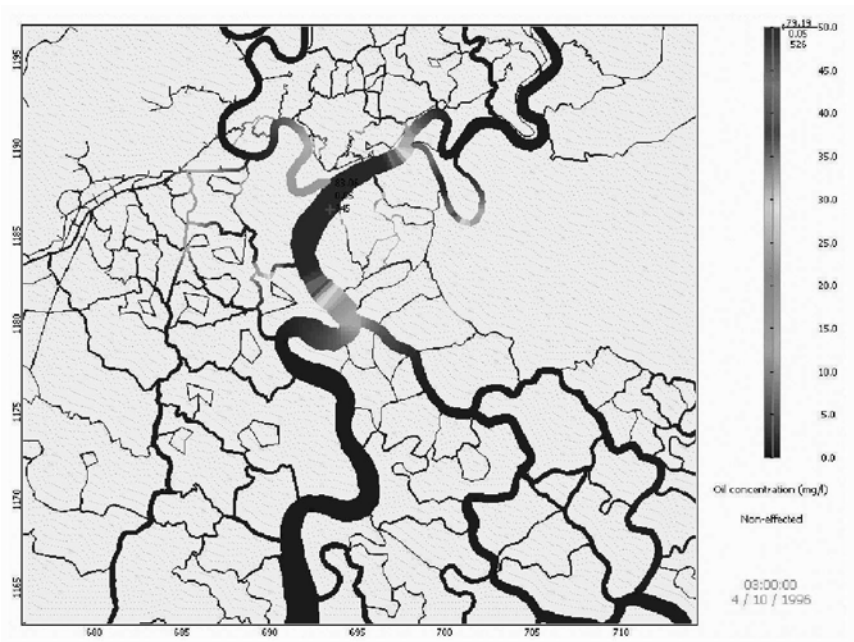
Figure 6. *Model predictions of erosion/deposition and tidal flow pattern in Ganh Rai Bay related to shipping routes in HCMC and TV-VT area. The sediment dynamics here is quite stable and suitable for harbour development because the access channel requires little dredging. However, almost all the bottom surface is mud.*

10. ENVIRONMENTAL ACCCIDENTS

Oil spills often occur. From 1990 to 2005 years, there were

- 9 large oil spills induced by technical accidents at Petromine “Bach Ho” and Dai Hung. The total volume of oil spilled on open sea was 1100-1200 tons, of which almost all was fresh oil. There were no responses and cleans of spilled oil.
- 4 large oil spills in coastal seas off Vung Tau City. The total volume of oil spilled was 400-500 tons, of which almost was industrial oil (DO, FO). There were no responses and just very minor cleanups of spilled oil on the shoreline.

- 6 large oil spills in HCMC harbour network (Cat Lai: 3, VICT: 1, Nha Be: 2, Can Gio Mangrove: 1). Among them, the spilled oil at Cat Lai Harbour on Jan. 01 of 1996 was 1700 tons. The oil slick was tidally trapped for several days in this estuary (see Animation 2) and caused very serious environmental damages. The total volume of oil spilled in the river basin of HCMC was 2600-2800 tons, of which almost all was industrial oil (DO, FO). There were some responses and just very some minor cleanups of spilled oil. A recent oil spill happened at Cat Lai harbour on January 21, 2005, with a volume of spilled oil of 350 tons.



Animation 2. *The spread and weathering of 1,700 tons of oil spilled at Cat Lai port in 1996 in the study area as predicted by "HydroGis".*

In fact, there were many more non-registered, small-scale, oil spills in HCMC and Ba Ria –Vung Tau Area. An analysis of the recorded accidents showed that:

1. The causes and the high frequency of oil spills are due to:
   -the tidal regime in river basin: a large tidal oscillation (3-4 m), 4 daily changes of current direction, and strong tidal currents forming tidal jets (see Figure 6).
   -the waterways are narrow, meandering, and shallow.
   -the infrastructure to combat oil spill is inadequate.
2. Large-scale and long-term impacts are inevitable due to the strong tidal currents and poor tidal flushing especially in the dry season.

3. The complex topography makes oil spills hard to control.
4. Wetlands are always the first in the list of victims.
5. The management measures are ineffective and the results of cleanups are poor.
6. There are insufficient security, warning, and rescue equipments on the vessels.
7. There are not enough chemicals to attack the spills, or booms to contain the spills. The International MARPOL 73/78 Convention was not implemented in Vietnam. Especially there is no environmental protection equipment on fishing boats. Environmental management is only focusing on accidents. There is no treatment equipment except at Sai Gon Harbour.

A plan to prevent and respond to environmental accidents in HCMC and the TV-VT area is urgently needed. This plan must:

- rely on a map of sensitivity to environmental accidents
- aim for an optimal localization of harbours, fuel and toxic stores, and pipe-lines.
- setup an optimal plan to an effective response to environment accidents in HCMC and the TV-VT area.
- provide the needed tools to combat environmental accidents; these include equipments such as boats, booms, dispersants and chemicals, pumps, absorbent materials, and communication tools. These tools must also include computer assisted modeling tools, GIS databases, and guides, to optimize the use of these resources to combat spills.
- training.

11. AN INTEGRATED TOOL FOR MANAGEMENT AND PREDICTION

The problems of managing and controlling environment in harbours of HCMC and the TV-VT area there are already severe. The problems are exacerbated by new complicated hydrodynamics features introduced by engineering developments and urbanization in the HCMC and the TV-VT area, including dykes, channels, sluices, roads, harbour, shrimp ponds, and other infrastructures.

This requires the development of a methodology, database, and integrated tools for management and prediction of water resources, water quality, foods, salinity intrusion, environmental impacts of infrastructure developments and oil spills. This is carried out by an integrated model "HydroGis" and "WQMA" (Nhan, 2003).

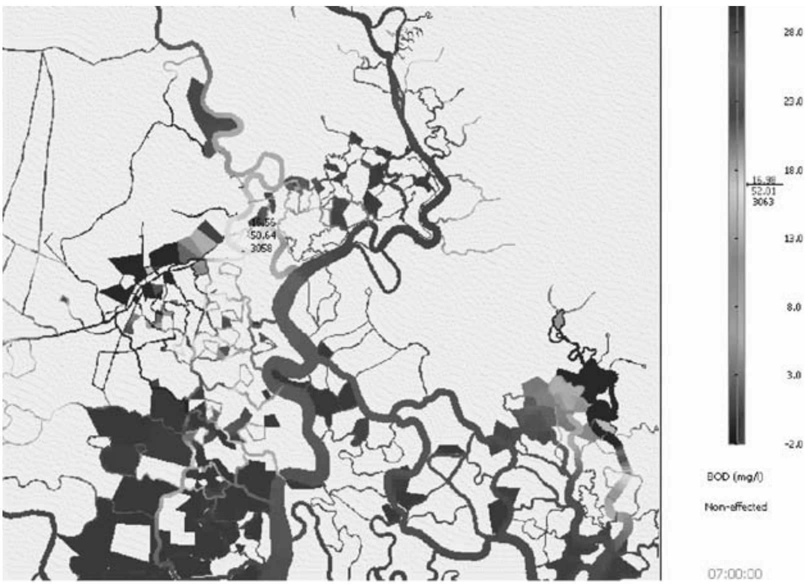
The input databases for these numerical experiments included:

- Geometrical and topographical databases of the river network (1200 cross-sections) and flood plan cell network (with 2054 cells).
- Hydrological and meteorological database on rainfall, wind, temperature, evaporation, infiltration, underground water flow, and river discharges.
- The water level at the downstream boundaries was predicted by 67 tidal harmonics.
- The database on pollutant sources from domestic areas, industrial zones, and harbours.
- The database on the oil spill at Cat Lai harbour in 1996 with related data of

weather and bio-physico-chemical properties of spilled oil DO.

The output databases included:

- Database in GIS of all parameters (water level, discharge, depth of inundation at any cell and cross-section). Animation 1 shows the space-time variations of water surface during the wet season (October).
- Database in GIS of all parameters modeled including organic pollutant (BOD5). The results are shown in Animation 3 for the wet season (October).
- Database in GIS of the spreading and weathering of 1,700 tons of spilled oil DO from Cat Lai harbour on January 27, 1996, as shown in Animation 2.



Animation 3. *The distribution of organic pollutant (BOD5) discharged from domestic areas, industrial zones, and harbours in the study area as predicted by "HydroGis".*

The studies reveal that the fate of pollutants in all harbours of HCMC and the TV-VT area is controlled by the tides. The pollutant is rapidly spread over a large area, and is slowly flushed to the sea. As a result, most branches of the Sai Gon and Thi Vai rivers are severely polluted by organic waste. The complex bathymetry ensures that spilled oil rapidly contaminates the river banks. Therefore, pollution prevention is a necessity.

12. CONCLUSIONS

Ho Chi Minh City (HCMC) and the Thi Vai–Vung Tau (TV-VT) area are the economic locomotive of Vietnam. Their shipping routes and their extensive network of harbours are crucial in this. Present studies shown that the environment of these water bodies is severely degraded. Urgent remediation measures are needed and include: firstly, the management and treatment of waste from urban and industrial areas, shipping routes, and aquaculture; secondly, there is an urgent need to remedy the serious status of exceedingly poor water and sediment quality, especially inside HCMC and the harbours of Go Dau and Thi Vai; thirdly, a solution must be found to arrest and reverse the present decline of environmental health and biodiversity in the CGMBR caused by human activities and shipping routes; finally, there is a pressing need to address the risk and high frequency of oil spills in HCMC and the TV-VT area. To be effective, these initiatives should be supported by multi-disciplinary research efforts and science-based tools, such as "HydroGIS".

13. ACKNOWLEDGMENTS

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CHAPTER 18

BIOPHYSICAL ENVIRONMENT OF MANILA BAY – THEN AND NOW

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MARIA LOURDES SAN DIEGO-MCGLONE,
CESAR L. VILLANOY, AND FERNANDO B. SIRINGAN

1. INTRODUCTION

Manila Bay is one of the most important bodies of water in the Philippines because of its historical, cultural and economic value. The bay has both local and international ports; it has been the seat of socio-economic development since pre-Hispanic times. It is also endowed with abundant natural resources, which have been the primary source of livelihood for residents in the coastal areas surrounding the bay. However with the continuous increase in population and industrialization, the bay is facing several issues arising from conflicts in the use of the bay and its resources, the continued decline in the quality of the bay (water and sediment), and the rapid deterioration of marine habitats found in the area.

Manila Bay is situated in the western part of Luzon between 14.23° and 14.87° N and 120.53° and 121.03° E (Figure 1). It is bounded by Cavite and Metro Manila on the east, Bulacan and Pampanga on the north, and Bataan on the west and northwest. The southern part of the bay opens to the South China Sea. It has a 190 km coastline, a surface area of 1,700 km^2, and an estimated volume of 2.89×10^{10} m^3. The width of the bay varies from 22 km at its mouth to a maximum of 60 km at its widest point. The average depth of Manila Bay is 17 m. Corregidor and Caballo Islands divide the entrance to Manila Bay into two channels. These channels are named North Channel and South Channel, and both are deep and clear of obstructions.

Manila Bay drains about 17,000 km^2 of watershed area composed of 26 catchments. The biggest catchment is that of the Pampanga River, with the Pampanga River contributing approximately 49% of the net fresh water influx into the bay (Jacinto et al., 2000a). Aside from freshwater, the northern part of the bay also receives lahar discharges. Lahar is an Indonesian word for a rapidly flowing

E. Wolanski (ed.), The Environment in Asia Pacific Harbours, 293–307.

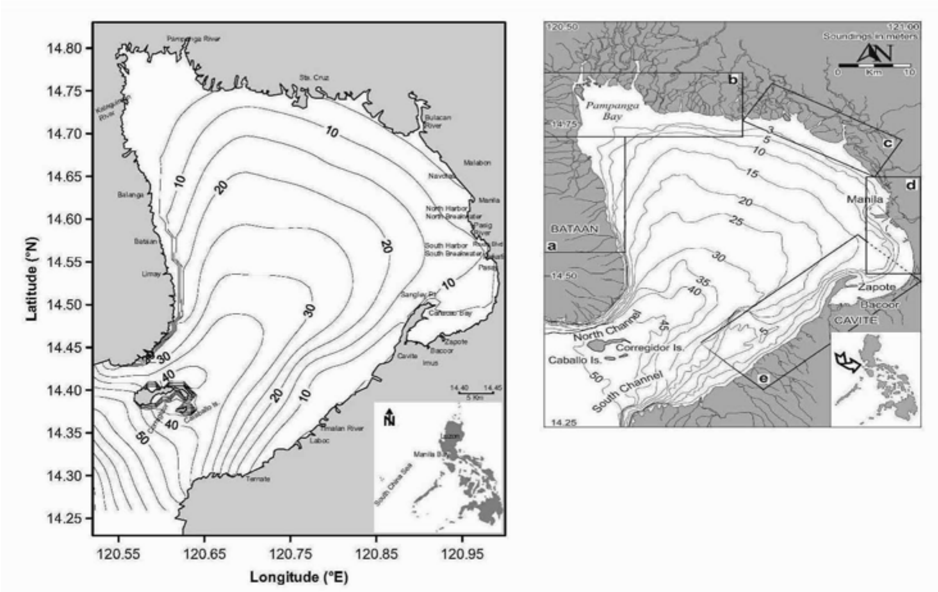

Figure 1. *(Left) Chart of Manila Bay (depth in m), and (right) location maps for Figure 6.*

mixture of rock debris and water that originates on the slopes of a volcano. Lahars are also referred to as volcanic mudflows or debris flows. The western coast receives discharges from the watershed of Bataan, while the eastern side receives a combination of rural and urban river flow from the coastal towns of Cavite and Bulacan. Also, the eastern coast receives the more polluted runoff from the rivers of Metro Manila including the Pasig River. Landsat images clearly show turbid plumes emanating from the rivers around Manila Bay (Figure 2). The Pasig River contributes about 21% of the net freshwater flux, while the remaining rivers contribute 26% (Table 1). The remaining 4% comes from precipitation into the Bay. The population in the drainage area of the bay is approximately 16 million people (NSO, 1996).

2. THE GEOLOGIC BACKGROUND

Until about 3,000 years ago, Manila Bay was connected to Laguna Lake (earlier known as Laguna de Bai), a 949 km^2 freshwater system (See Figure 3; Jaraula and Siringan, pers. comm.). However, recurring episodic uplifts along the West Marikina Valley Fault (Jaraula and Siringan, submitted) separated it from Manila Bay. Currently, the bay-lake interaction between Manila Bay and Laguna Lake is through Pasig River, the only outlet of the lake.

***Table 1**. Estimates of Annual Runoff from the rivers surrounding Manila Bay (Alejandrino et al., 1976).*

River	Drainage Area (km^2)	Runoff ($x\,10^6\ m^3$)
Pampanga	9,759	10,930
Angat	781	873
Balanga	144	161
Pasig	4,678	7,485
Imus	105	168
Ilang Ilang	82	131
Canas	210	336
Laboc	96	154

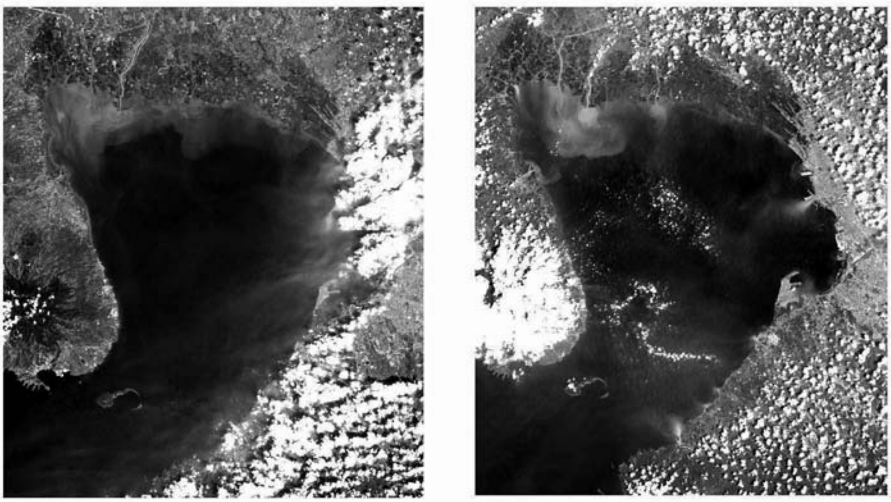

***Figure 2.** Landsat image of Manila Bay in October 1999 (left) and January 2000 (right) showing river plumes at the end of the southwest and northeast monsoons, respectively. (Source: AFMA Ocean Color for Sustainable Fisheries, MSI, DA-BAR)*

3. SHORELINE AND BATHYMETRY CHANGES

There have been recent changes in the physical features of Manila Bay, in particular its shoreline positions. These are largely due to human activities such as land reclamation and the conversion of mangrove and mudflat areas into fishponds (MADECOR and National Museum, 1995; PEMSEA and MBEMP TWG-RRA, 2004). Moreover, environmental processes such as soil erosion, siltation and the combination of local and global sea level rise that cause low retention of sediments near the coast, have also contributed to changing the bay's coastline.

From 1944 to 1991 the coastline of Bataan, a province located west of Manila Bay, has undergone 250-300 m of progradation that resulted in a net land gain of about 2.2 km^2 (Siringan and Ringor, 1997, 1998) (Figure 4a).

In the Pampanga coastline located northwest of the bay, from 1944-1991, most of the coastal segment underwent retreat and lost approximately 0.9 km^2 of land (Fig. 4b). In contrast, the adjacent northern coastline of Bataan, south of Kalaguiman River, experienced a net progradation of as much as 200 m (Fig. 4b). In 1944, the shoreline segments shown in Figure 4b was irregular and became more linear by 1977. This change is due to conversion of the land into fishponds. Reduction in sediment supply through decrease in sediment yield of Pampanga and

Figure 3. *Map showing the connection between Manila Bay and Laguna Lake (Laguna de Bai). False colour image of Pasig River obtained by LANDSAT 7 ETM+ in April 2002 courtesy of Global Land Cover Facility (http://glcfapp.umiacs.umd.edu).*

Binangbang rivers, may have caused shoreline retreat along the Pampanga coast. Pantabangan Dam, the largest multipurpose dam in the Pampanga River basin was constructed in 1971 and became operational in 1974 causing marked decrease in freshwater and discharge.

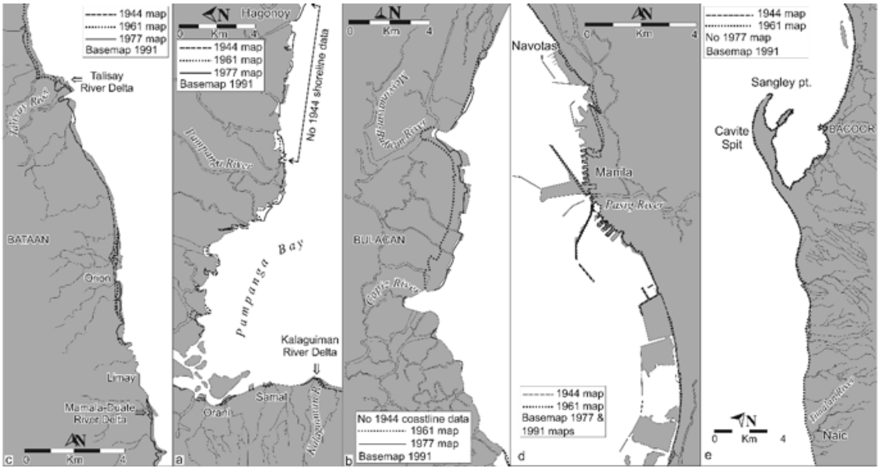

Figure 4. *Changes in shoreline position of different areas in Manila Bay (a) Bataan (b) Pampanga (c) Bulacan (d) Manila (e) Cavite. See location maps in Figure 1.*

An extensive shoreline retreat was noted from 1961 to 1991 between Capiz and Meycauayan rivers in Bulacan, a province northeast of the bay (Figure 4c). From 1961 to 1977, extensive progradation was evident between the coastal segment of Capiz and Meycauayan rivers. Progradation was still dominant from 1977 to 1991, showing a linear coastline which indicates man-made structures such as fish pens. This segment of the coast along Manila Bay shows the largest change in shoreline of the bay where as much as 1200 m of progradation was measured. The dominant change in the shoreline position in this area from 1961-1991 was seaward with an approximate area of 5.7 km^2.

In the Metro Manila shoreline, the changes are mainly due to land reclamation. Approximately 600 m of shoreline have been reclaimed in the northern part of the North Harbor from 1944 to 1991, covering an area of 6.1 km^2 (Figure 4d). Through building of seawalls and breakers, coastal erosion of the reclaimed land has been prevented. Changes in bathymetry of the Pasig River are due to overall shoaling from sediment deposition caused by a net northerly sediment transport along the east and northeast segments of the bay (Siringan and Ringor, 1998). Minimal deepening of the river, by dredging, is limited to small areas.

Shoreline erosion of about 50 to 100m occurred from Timalan River to the Cavite Spit (Figure 4e). Old infrastructures such as railways documented the 1944 map are no longer seen in the field. Anecdotal accounts of local residents indicate that these infrastructures were located approximately 50 m offshore from the present shoreline. Continuing erosion is indicated by exposed roots of coconut trees along the shore. The overall shoreline retreat along the western coast of the Cavite Spit is coupled with deepening in the immediate offshore areas.

4. SEDIMENTATION RATE

Using Cs-137, sedimentation rates in Manila Bay were determined to be 4.3 cm y^{-1}, 4.8 cm y^{-1} and 3.2 cm y^{-1} in Bacoor, Pampanga and Meycauayan, Bulacan, respectively (Acorda, 1985; Santos and Villamater, 1986). In 1990 sedimentation rates were estimated to be greater than 3 cm y^{-1} along the northeastern half of the bay and less than 1 cm y^{-1} in the central and southern portion of the bay (de las Alas, 1990). In 1996, sedimentation rates ranging from 0.64 to 1.5 cm y^{-1} were reported, with the highest rate obtained from a core in Bacoor Bay in Cavite (Furio et al.,1996). A sedimentation rate of less than 1 cm y^{-1} was reported by Santiago (1997a). In Siringan and Ringor (1998), the predominant sediment dispersal pathways and lateral variation in sedimentation rates were determined in Cavite, Manila and Pampanga using secondary sedimentation rate data and changes in water depth from 1901 to 1950. Based on their study, the sedimentation rate north of the Cavite Spit and west-northwest of the Pasig River mouth was determined to be 9 cm y^{-1}. Although estimates of sedimentation rates from several studies differed, the rates in Cavite and Manila were consistently high which imply that they are likely sinks of sediment and possibly pollutants in the bay.

5. CLIMATE

The climate of Manila Bay is classified as Type 1 in the Coronas Classification of the Philippine climate, which is characterized by two pronounced seasons, the wet and dry periods (UNEP/UNDP/EMB-DENR, 1991). The rainy season occurs from June to September, and the dry period is typically from November to April. The annual average rainfall from 1988 to 1999 is 214.3 cm, and the highest reported annual average rainfall is 274.1 cm that occurred in 1999. The rainiest month of the year is August (EMB-DENR, 1991; MMFCP, 2001).

There are three prevailing wind patterns in the bay and its catchment area, the NE monsoon from October to January, Trades from March to May, and the SE monsoon from June to September. The wind speed in the area can range from 2 to 4 m s^{-1} (UNEP/UNDP/EMB-DENR, 1991).

6. CIRCULATION PATTERNS

The tides in Manila Bay are predominantly mixed diurnal-dominant with an average tidal range of 1.2m during spring tide and 0.4m during neap tide. The estimated residence time of water in the bay is ~31 days (Jacinto et al., 2000a).

The three major factors that affect the circulation of Manila Bay are (1) freshwater discharges from the surrounding watersheds, (2) the tides, and 3) the wind stress. The circulation in Manila Bay is strongly influenced by tidal flow in and out of the bay, especially during the Northeast Monsoon season or during spring tides, which is when the waters are vertically homogeneous (Villanoy and Martin, 1997; Villanoy et al., 2001). However, due to the narrow mouth of the bay, recirculating gyres are often observed, particularly during neap tides or when stratification is strong. There appears to be a net counter-clockwise flow characterized by a net inflow into the Bay greater in the southeastern side and an outflow in the northwestern side (Figure 5; Villanoy et al., 2001). Thus there is a marked asymmetry in the tidal flow between Bataan and Cavite.

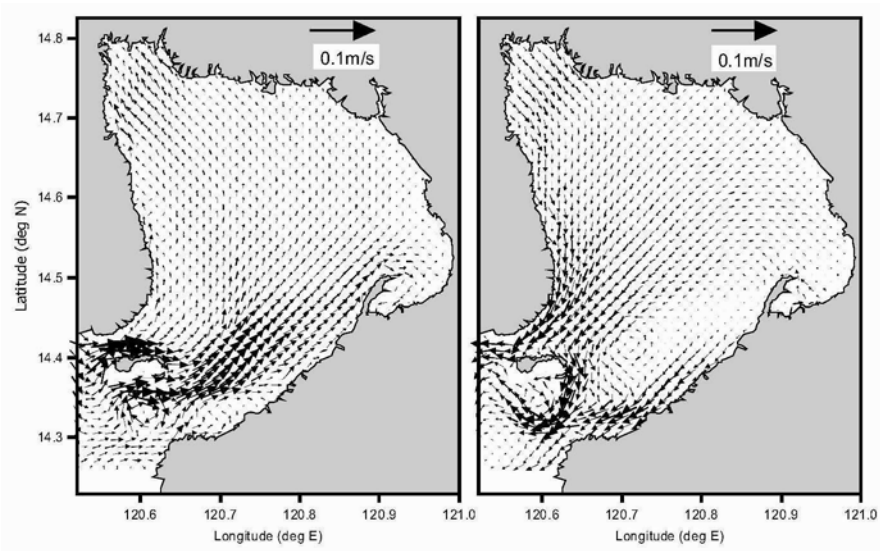

Figure 5. *Circulation pattern of Manila Bay during weak tides, the flow patterns show some variations across the bay.*

The presence of wind forcing introduces significantly higher variability in the surface circulation patterns; nevertheless the asymmetry in the flow between Bataan and Cavite remains. This may be associated with the difference in bathymetry off these two provinces (Figure 6). The shallower depths off Cavite indicate that the current variability in that area is highly influenced by the prevailing winds. The results from circulation models for different wind conditions all show that the currents off Cavite flow predominantly along the direction of the wind (Villanoy et al., 2001). The shallow area off Pampanga exhibits a similar response to wind forcing. The only area, which shows significant variability associated with the tides, is the bay mouth area.

The temperature and salinity distribution in Manila Bay is characterized by strong seasonal variations. Seasonal stratification patterns are strongest during the Southwest monsoon when there is a strong freshwater input in the bay (Villanoy et al., 1996). The precipitation and the freshwater flux from the rivers result in a stratified upper layer with a thickness that can range from 10 m for the shallow and inner part of the bay to 15-20 m in the deeper areas (Velasquez et al., 2002). The water temperature during the southwest monsoon typically ranges between 28 to 31° C; the salinity range is 20 to 36. During the northeast monsoon, the water column becomes homogeneous down to about 40 m depth, due to stong mechanical and convective mixing; during that season, the water temperatures range between 25 to 28° C and salinity from 30 to 36 (Villanoy et. al., 2001). Below 20 m depth, variations in temperature and salinity are small. The strong tidal currents in shallow water can also induce vertical mixing in the water column.

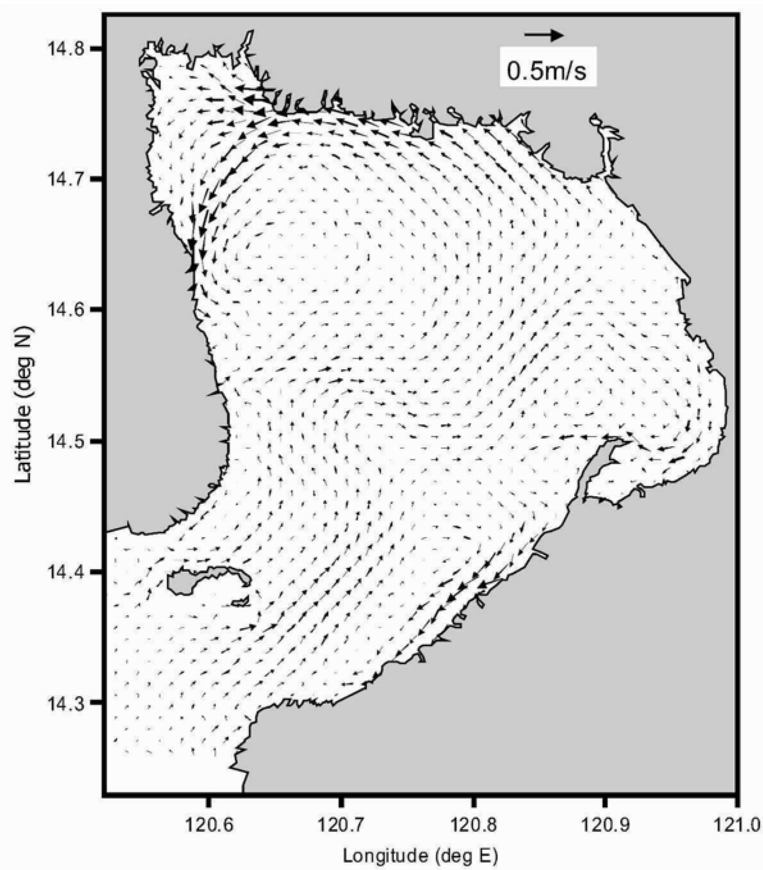

Figure 6. *Combination of wind and tide forcing results in the formation of multiple gyre systems in Manila Bay.*

7. WATER QUALITY

Manila Bay being one of the important bodies of water in the Philippines, and recognized for its uses, importance and natural values by its various stakeholders, has undergone significant changes in its water and sediment chemistry through time. A clear linkage between environmental degradation, increase in population, and other human activities is the reason for the changes in the bay.

7.1. Bathing water quality

The beaches located in the eastern part of the bay already started to shown deterioration in bathing water quality from as early as 1982. The status of the bathing beaches was assessed based on coliform counts. The geometric mean of total and fecal coliform (TC and FC) in Manila, South Breakwater, Pasig River outlet and Bacoor (Cavite) has been very high (37,318 TC and 14,860 FC; 32,350 TC and 15,550 FC; 60,202 TC and 16,550 FC; 10,250 TC and 4,810 FC MPN/100 ml, respectively) even in the early 80's. This has been associated with the discharge of untreated wastes coming from drainage, sewer outlets, and direct discharges from shanties and stilts located in the shoreline area (Acorda, 1985). Ten years later, and with continued increase in population in the coastal areas of Manila Bay, the quality of the water has further deteriorated. Coliform counts have increased by more than five-fold from the 80's to 1998. In general, the hygienic water quality of beach resorts along the bay is not in compliance with the criteria for bathing water quality, which is 1000 MPN/100ml for TC and 200 MPN/100ml for FC (DAO 34, 1990; PRRP, 1998).

7.2. Dissolved Oxygen (DO)

There is a seasonal variation in DO distribution in Manila Bay. In 1985, the average and surface water DO concentration were above the criteria value for Class SC waters (waters suitable for commercial and sustenance fishing) of 5.0 mg/L (DAO 34, 1990). However, values as low as 3.0 mg/L were observed in bottom waters when the water column is highly stratified (Acorda, 1985). The combined data of Jacinto (2000b) and PRRP (1998) from 1995 to 2000 were used to evaluate the changes in DO over time. The average DO values in different stations and several sampling periods from 1995 to 2000 showed a decreasing trend. The lowest value (<5.0 mg l^{-1}), from the surface to the bottom was recorded in 2000. In general, the bottom waters of the bay show continued deterioration in DO levels over the last twenty years, especially in the vicinity of Pasig River and Port Area, Manila (Figure 7).

7.3. Oil and Grease

Due to the continued increase in domestic and municipal discharges of refineries in nearby coastal areas, and shipping activities that include accidental oil spills, the amount of oil and grease in the water is of concern. The recorded oil and grease concentrations at 13 stations within the bay did not show any significant increase from 1985 to 1993 (BFAR, 1995). However, a more recent report in August 2001 from five different stations, showed values exceeding the criteria of 3.0 mg/L (DAO 34, 1990; PEMSEA and MBEMP TWG-RRA, 2004). This trend from the 80's and 90's to the present decade indicates deteriorating quality of water in the bay, with increasing amount of oil and grease.

7.4. Nutrients

Like DO, the dissolved nutrients nitrate (NO_3), ammonia (NH_3) and phosphate (PO_4) exhibit seasonal variations. Higher values of NO_3 and PO_4 were observed during the wet season compared to the dry season and in areas near rivers. However, among the nutrients, PO_4 appears to be the contaminant of concern. From the mid 80's to the late 90's, phosphate values appeared to have continuously increased in the waters of the bay. Figure 8 shows NO_3 and PO_4 values at different stations in the bay from 1994 to 2000. Values exceeding the criteria (ASEAN, 2003) appear to have increased. Likewise, the concentrations at the center of the bay have also increased. The rise in nutrient concentration over time may be due to agricultural runoff, river discharges and fertilizers from nearby fishponds. The rough estimate of

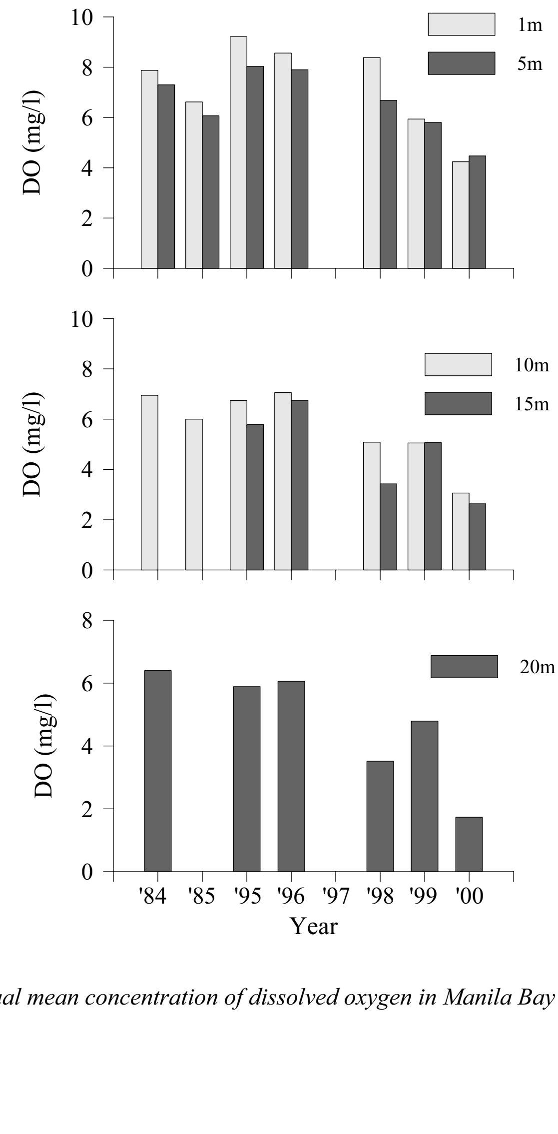

Figure 7. *Annual mean concentration of dissolved oxygen in Manila Bay at various depths.*

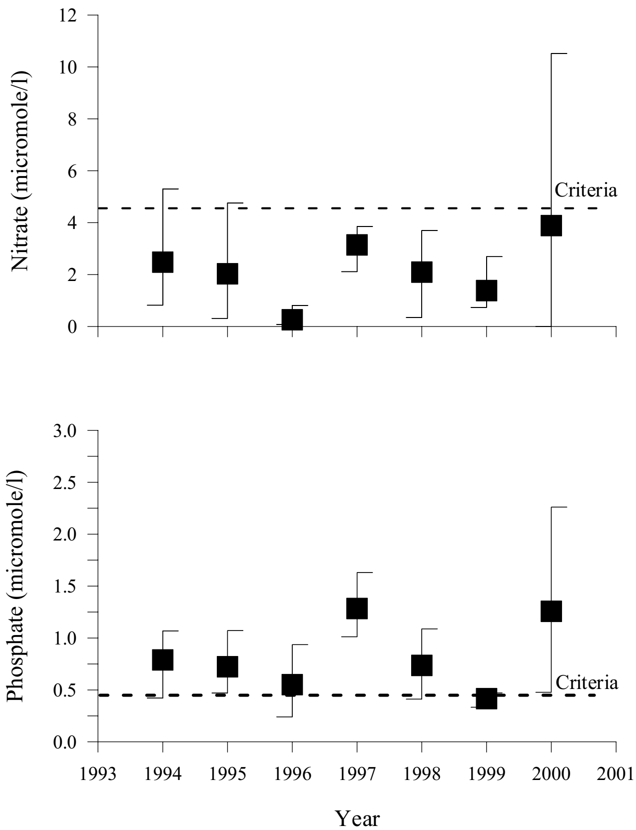

Figure 8. *Maximum, mean and minimum nitrate and phosphate concentrations in Manila Bay from 1994 to 2000.*

inorganic nutrient discharge into the bay is approximately 40 x 10^6 M y^{-1} of inorganic P and 600 x 10^6 M y^{-1} of inorganic N (Jacinto et al., 1999). With increasing population and ineffective waste management, waste loading rates are higher today.

8. SEDIMENT CHEMISTRY

The sediments in Manila Bay are predominantly greyish and clayey-muddy, and are soft and fine in texture. The organic matter content of the sediments ranged between 5 and 19% (PRRP, 1998). There is localized trace metal enrichment in the bay. Since the mid 80's to the late 90's, elevated values of Pb and Zn were consistently observed within the vicinity of the Metro Manila (Acorda, 1985). For the metals Cd, Hg, Zn, Pb, and Cr, the Malabon-Navotas, Paranaque, Pasig and Bulacan rivers are the point sources of these metals into the Bay (PEMSEA and MBEMP TWG-RRA, 2004). Nutrient flux from the sediments, specifically of NH_3 and PO_4, varied with season, with higher fluxes measured near the Pasig, Bulacan and Pampanga Rivers (Azanza et al., 2005; Agustin and San Diego-McGlone, 2003). The total polyaromatic hydrocarbon (TPAH) distribution in Manila Bay showed localized contamination in the eastern part of the bay, which is more commercialized and urbanized (PEMSEA and MBEMP TWG-RRA, 2004). The PAHs identified were petrogenic or from oil discharges from ships and refineries; and pyrolytic or those derived from combustion processes (Santiago, 1997b). These TPAHs may enter the bay through rivers, discharge pipes, outfalls and surface run-off.

9. USES OF MANILA BAY

The bay is considered one of the most strategically and economically important bodies of water in the Philippines. It has a significant socio-economic role for Metro Manila and the surrounding provinces that share its long coastline. It is recognized under the Manila Bay Declaration in 2001 as a source of food, employment and income for the people, the local and international gateway of the country to promote tourism and recreation (MBEMP, 2001). It is also important because of its cultural and historical heritage.

9.1. Fishing

Fisheries and aquaculture are among the major sources of livelihood in the coastal areas of Manila Bay. The whole area of the bay, except those areas near the ports is used as a major fishing ground. Fisheries resources at present suffer from a significant decline from the 1940's onward, primarily due to over-fishing or over-collection. There has been a decline in trawl catch per unit effort or CPUE (kg h^{-1}) from 46 in 1947 to 10 in 1993. The demersal biomass decreased dramatically from 4.61 mt km^{-2} or 8,290 tons in 1947 to 0.47 mt km^{-2} or 840 tons in 1993 (PEMSEA and MBEMP TWG-RRA, 2004).

Shellfisheries are widespread in the southern part of NCR and Cavite; however, in Laguna Lake and inland areas aquaculture is more extensive. Poor management of shellfisheries resulted to unstable production of commercially valuable mussels and oysters, disappearance of the windowpane oyster and contamination of shellfish particularly with fecal coliforms.

9.2. Ports and shipping

Almost completely landlocked, Manila Bay is considered one of the world's great harbours. The biggest shipping ports, ferry terminals, fish port and yachting marina are found in the bay. An average of 30,000 ships arrive and depart from these ports annually to transport passengers, manufactured goods and raw materials (MBEMP, 2001). The bay has two ports, the North and South Harbors, both of which are protected by breakwaters. The Pasig River separates these two harbors. The North Harbor is the smaller of the two harbors and is used solely for inter-island shipping, while the South Harbor is used for large ocean-going vessels.

9.3. Tourism and recreational values

There are a number of beach resorts located along the coast of Cavite and in selected areas in Bataan, which are used for boating and as bathing beaches (Acorda, 1985; MBEMP 2001). Along the stretch of Roxas Boulevard a row of bars and restaurants overlooking the bay also serve as a favourite recreation spot in Manila Bay.

9.4. Receiving site for domestic and industrial sewage

Manila Bay receives solid and liquid wastes from the surrounding watershed through tributaries and major river systems. Large amounts of waste drained into the bay come from domestic discharges, and only 15% of the population is connected to the Metro Manila sewerage system (IMO, 1994). The Metro Manila Water and Sewerage System (MWSS) sewerage facilities are limited to only some areas in the city of Manila and parts of Makati and the sewerage service covers less than 7% of households in the service area (David, 2000). The rest of the population discharges their wastes to septic tanks or directly to the rivers that end up in the bay.

10. ACKNOWLEDGMENT

We thank Dr. Laura David for providing access to satellite images obtained through the AFMA Ocean Color for Sustainable Fisheries project funded by the Bureau of Agricultural Research, Department of Agriculture; and Olive Cabrera and Eileen Peñaflor for their assistance in processing the images.

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CHAPTER 19

MANILA BAY: ENVIRONMENTAL CHALLENGES AND OPPORTUNITIES

G.S. JACINTO, R.V. AZANZA, I.B. VELASQUEZ,
AND F.P. SIRINGAN

1. INTRODUCTION

Manila Bay is considered one of the finest harbours in Southeast Asia, facilitating commerce and trade between the Philippines and neighbouring countries, and offering numerous livelihood opportunities to millions of Filipinos The bay has a watershed area of 17,000 km^2 comprised of 26 catchment areas, and rimmed by a megacity and other heavily populated coastal areas (Figure 1). Several heavy industries, refineries and a power plant are located in Bataan while Metro Manila is highly urbanized with light and heavy industries and factories located in various parts of the metropolis.

Manila Bay is beset with many environmental problems and challenges. A major concern is increased organic and nutrient loading coming largely from untreated domestic and agricultural wastes, and to some extent, generated by industries in the bay's watershed. The enhanced level of nutrients is considered to contribute significantly to episodic hypoxic conditions in the bay waters, increased incidence of toxic and nuisance algal blooms, and higher suspended material in the water column.

An ever-increasing coastal population that relies heavily of marine fisheries as a source of food and livelihood, has also brought tremendous harvesting pressure on the biological resources in the bay. The absence until only recently, of laws, guidelines, and sustainable practices to protect the marine resources has not helped

In this chapter, we examine the major marine environmental issues that confront Manila Bay and consider the initiatives that are expected to bring about the reversal of, or at the very least slow down, the continuing degradation of the bay.

E. Wolanski (ed.), The Environment in Asia Pacific Harbours, 309–328.

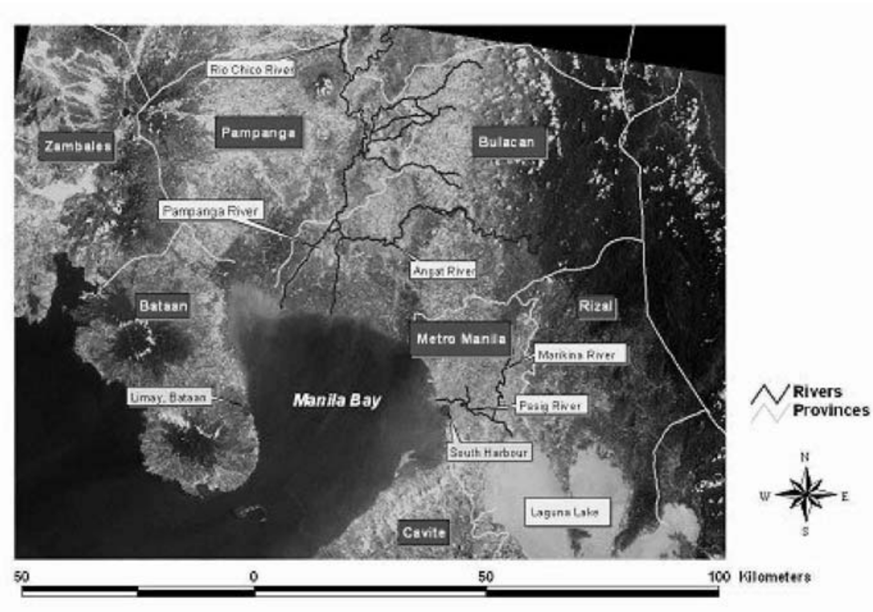

Figure 1.*Manila Bay coastal and watershed areas.*

2. CHALLENGES

2.1. Pollution

Manila Bay is a shallow estuary receiving drainage from immediate watershed through tributaries and major river systems. The population of the cities and municipalities within the bay's catchment areas is estimated at 16 million people (approx. 18% of the population of the Philippines), with 8 million people inhabiting the Pasig River watershed (Jacinto et al., 1998). Large amounts of waste drain into the bay from domestic discharges since only 15% of the population is connected to the Metro Manila sewerage system (IMO 1994). The rest of the population discharge their waste to septic tanks or directly to rivers that end up in the bay.

With the rapid increase in population and industrialization within the Manila Bay region, water quality has deteriorated. Anecdotal reports suggest that Manila Bay is already eutrophic or, at least, eutrophic from time to time. Eutrophication is characterised by an increase in the rate of supply of organic matter to an ecosystem by either primary production by autotrophs within the system (autochthonous carbon) or by production due to an input of organic matter from outside the system (allochthonous carbon) (Hinga et al., 1995). Episodic hypoxia and phytoplankton blooms, phenomena that occur in Manila Bay, are manifestations of eutrophication.

2.1.1. Hypoxia

Episodic or prolonged oxygen depletion may occur in shallow, stratified and eutrophic systems like lakes, rivers and coastal waters (Rosenberg, 1990; Breitburg et al., 1994). This condition normally occurs because of the presence of a stable water column, which separates the bottom water from the surface water, and the deposition of organic matter in the water (Rabalais et al., 2002). Anoxia exists when the oxygen concentration is equal to 0 while hypoxia is present when the dissolved oxygen concentrations are below 2.8 mg l^{-1} (Wu, 2002). Hypoxia is typically an indication of eutrophication (Nixon, 1995). The water is characterized by the increased nutrient inputs, usually in the form of nitrogen and phosphorus, and increased sulphide concentrations near the sediments (Nilsson and Rosenberg, 1994; Sandberg, 1994; Rabalais et al., 2002; Wu, 2002). The persistent occurrence of hypoxic or anoxic water can cause detrimental effects on the aquatic ecosystem. Among others, the condition can result to severe damage to the fisheries and to benthic communities (Nilsson and Rosenberg, 1994; Sandberg, 1994; Karim et al., 2002; Wu, 2002).

In Manila Bay, reduced dissolved oxygen (DO) concentrations were already noted twenty years ago during periods when the bay waters are stratified, while higher values were observed when there is minimal or no stratification in the water (Acorda, 1985).

Average DO profiles from various stations and several sampling periods from 1995 to 2000 showed decreasing trends (Figure 2). The lowest value from the surface to the bottom was recorded in 2000. The mean DO concentrations in 2000 were noticeably below the Department of Environment and Natural Resources (DENR) criteria of 5.0 mg l^{-1} for marine waters (Department Administrative Order 34). A decrease in oxygen concentrations was consistently observed from 10 to 15 m, markedly lower than values obtained in the 1980's (Figure 3). However, there were no observed seasonal oxygen trends from 1995 to 2000. Average oxygen values at 15 m in 1998 and 2000 were < 4.0 mg l^{-1}, sometimes < 2.0 mg l^{-1}. Bottom oxygen concentrations in 1996 to 1999 at various parts of the bay were generally < 5 mg l^{-1}. Values < 2.8 mg l_{-1} were apparent from the center of the bay towards the northwestern section.

The drastic change in the bottom DO concentration in most parts of Manila Bay is most likely due to the increasing organic load in the bay. The apparent shift in the dominant benthic organisms and the decrease in the species diversity of fish in the bay were associated with the continuous decrease in the bottom DO values (Manila Bay-IRA, 2000).

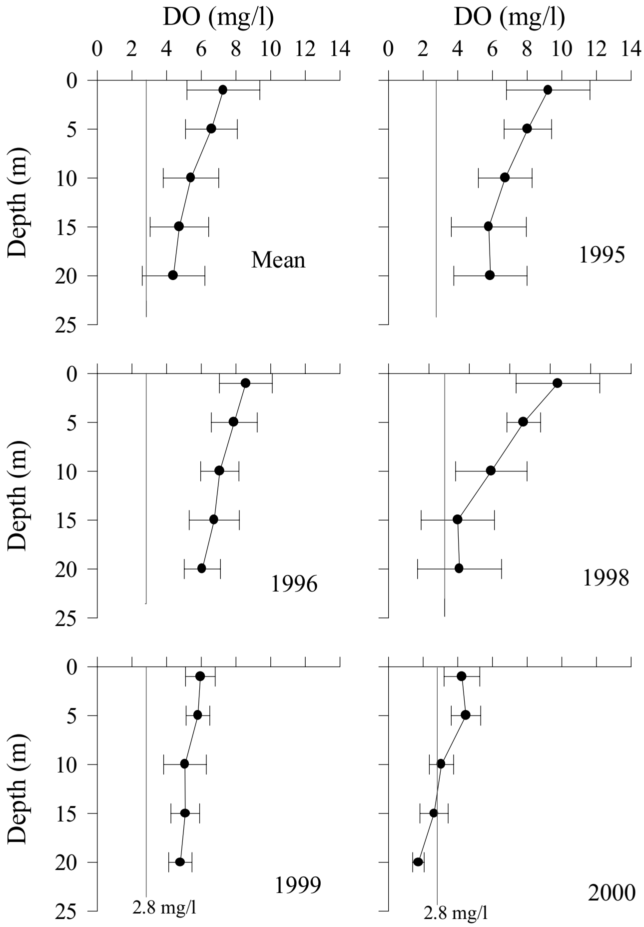

Figure 2. *Average DO profiles from various stations and sampling periods in Manila Bay from 1995-2000.*

2.1.2. Oil Spills

While the primary inputs of oil to Manila Bay are believed to occur from land-based sources (in particular refineries, municipal wastes and urban runoff), sea-based sources (ships and motorized boats) are also contributors. Data on oil spills for the period 1990 to April 2005 from both ships and industries were obtained from the Philippine Coast Guard.

As an operational definition, an oil spill is considered a large spill when the volume of oil discharged is greater than 1,500 metric tons (MT). The highest volume of oil spilled in the bay was 747 MT and falls under small spills. Three of the four spills where the volumes of oil discharged were highest were from ships.

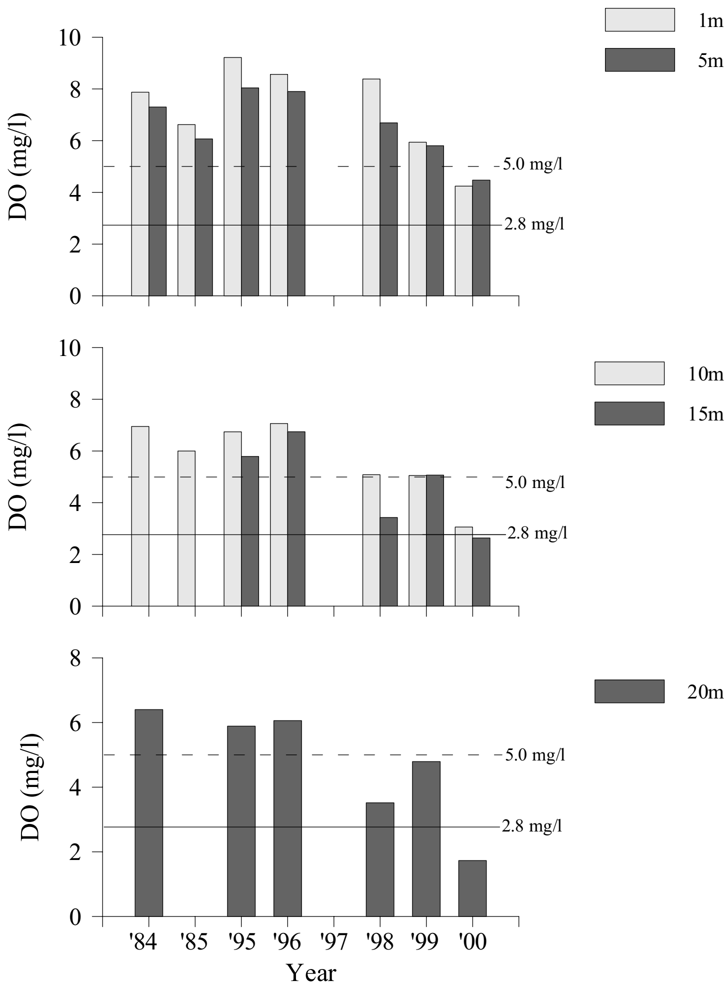

Figure 3. *Average DO values at various depths in Manila Bay in the 1980's and from 1995-2000. The DENR criteria value for Class SC waters of 5 mg l^{-1} and the hypoxia value of 2.8 mg l^{-1} are shown.*

The frequency of oil spill incidents per year is shown in Figure 4. The highest frequency of oil spills (12) was in 1995 while the highest total volume of oil spilled in the bay was from the two oil spills in 1999. These incidents occurred in the Manila South Harbor and Limay, Bataan.

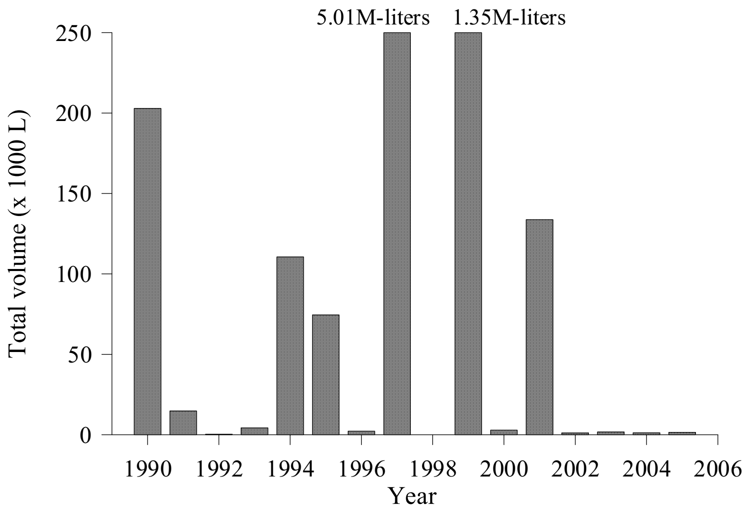

Figure 4. *Volume of accidental discharge of oil from land- and sea-based sources in Manila Bay.*

The greatest number of oil spill occurrences was in the vicinity of Metro Manila with 19 incidents recorded, followed by Bataan with 8 incidents, and Cavite and Rizal with one incident each (Figure 5).

The high frequency of oil spills in Metro Manila area can be due to the large number of ships and maritime activities at the North and South Harbors, the presence of an oil terminal, and discharges from industries located along the rivers (PEMSEA, 2004). Oil spills in Bataan can be due primarily to shipping activities and discharges from industries along the coast.

2.1.3. Trace metals

The heavy metals in Manila Bay come from various sources that range from those originating from land (e.g., domestic sewage, industrial effluents, runoff, combustion emissions, and mining operations) to sea-based sources (e.g., shipping and port operations). In general heavy metal concentrations in the water column were found low (PRRP, 1999; BFAR, 1995; Prudente et al., 1994; Narcise and Jacinto, 1997). A more recent study by Velasquez et al. (2002) on metals in the bay suggests that the Cu, Cd, and Zn at the surface of the water come from point (and likely anthropogenic) sources, with possible contributions from the Pampanga and Bulacan rivers.

Two areas of metal contamination in sediments were identified in Manila Bay - Bataan and Metro Manila (PEMSEA, 2004).

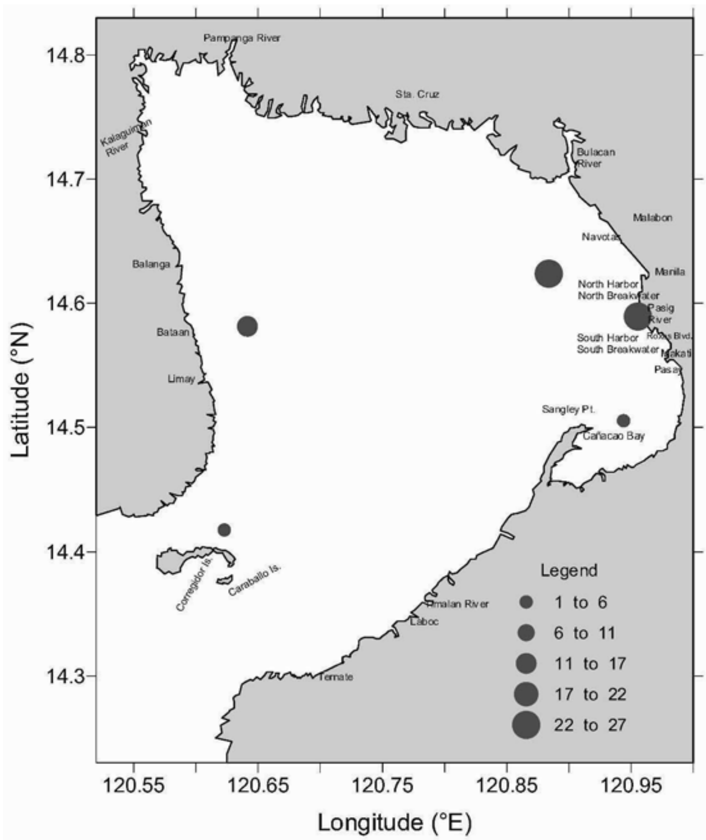

Figure 5. *Oil spill incidents in Manila Bay for the period 1990-2005. The higher the number incidents in the area, the bigger the radius of the circle. (Source: Philippine Coast Guard).*

2.1.4. Organic Contaminants

Concentrations of 16 commonly used pesticides in surface sediment from 10 established monitoring stations were measured in 1996 (PRRP, 1999). Except for alpha-BHC, the values obtained were at or near detection limits. Using the Hong Kong Interim Sediment Quality Values (EVS, 1996), problem contaminants appear to be limited to 4,4'- DDE and 4,4'-DDT.

Levels of the 16 commonly used pesticides in shellfish were generally within acceptable limits assuming an average consumption of 20 g/person/day for shellfish (FNRI, 1987). For fish where a consumption rate of 92 g/person/day was assumed (FNRI, 1987), some health risks for aldrin and heptachlor may be indicated (PEMSEA, 2001).

Santiago (1997) measured PAH's from 19 stations the western section of Manila Bay, and 16 stations on the eastern section of the bay. PAH levels in the eastern section of the bay, which is more urbanized, were higher than the levels on the western side, suggesting the influence of human activities on PAH distribution. The PAH in Manila Bay sediments appear to come principally from petrogenic (e.g., oil discharges from ships, refineries and industries) and pyrolytic sources (from combustion sources) (Santiago, 1997).

2.2. Harmful Algal Blooms (HAB's)

Pyrodinium bahamense var. *compressum*, a thecate chain-forming dinoflagellate (Figure 6), has been reported to have caused the world's greatest number of Paralytic Shellfish Poisoning (PSP) (Azanza and Taylor, 2001). It accounted for 41% of the total global PSP cases, *Alexandrium* spp. for 37%, *Gymnodinium* spp. for 12%, and other unidentified species for the remaining 10% (Azanza and Taylor, 2001).

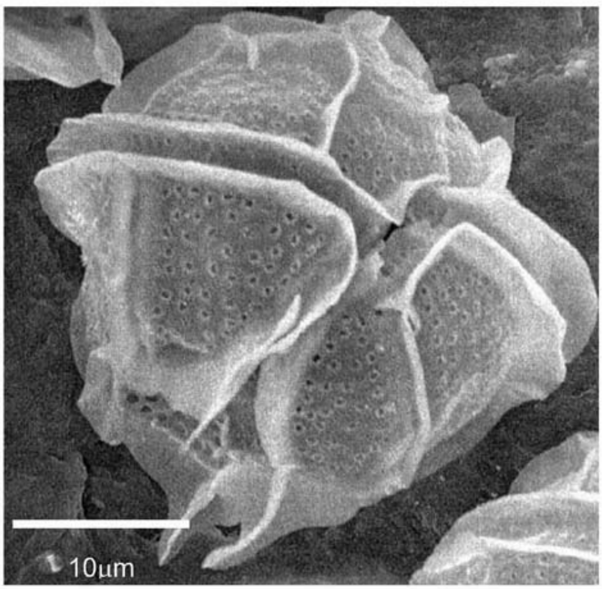

Figure 6. *Scanning electron photomicrograph of the thecate chain-forming dinoflagellate, Pyrodinium bahamense var. compressum.*

In 1988, *Pyrodinium bahamense* var. *compressum* rendered green mussels (*Perna viridis*) and oysters (*Crassostrea* sp.) toxic up to 1,005 mg STX eq/100g during its first documented bloom in Manila Bay (Corrales and MacLean, 1995). In a span of two months, 129 people were hospitalized, four of whom succumbed to PSP. The estimated cost/loss then was US$300,000 day^{-1} at the height of the bloom.

Pyrodinium continued to bloom in the bay every year thereafter and caused 877 PSP cases and 44 deaths in a period of 10 years (1988-1998). However, just as the bloom unexpectedly appeared in 1988, it disappeared as abruptly in 1999 and apparently has not reoccurred since (Azanza and Miranda, 2001; Hansen et al., 2004). Nevertheless, representatives of the species could still be found in the samples collected in the area during the months they are expected to bloom.

2.2.1. Mechanism of Pyrodinium bloom

Results of monitoring and research efforts from 1991 to the present have shown a pattern in bloom formation of *Pyrodinium* and occurrence of PSP toxin in shellfish (Bajarias and Relox, 1996; Azanza and Miranda, 2001; Azanza et al., 2004). The blooms usually start towards the end of summer in the Philippines, i.e., May or June, and peak during early southwest monsoon. Increased thermal stratification and vertical stability of the water column could account for the formation and maintenance of the blooms (Villanoy et al., 1996). *Pyrodinium* blooms in the bay usually end towards the last part of the southwest monsoon (Bajarias and Relox, 1996; Azanza and Miranda, 2001).

As cited by Azanza and Taylor (2001):

> "Villanoy et al. (1996) have hypothesized that during the NE monsoon, high turbulence caused the strongest vertical mixing resulting in strong resuspension of sediments and cysts. During this period, suspended cysts could not start the bloom because of an unfavorable environment, i.e. low temperature and low nutrient level. Reduced light due to strong water mixing could also prevent cyst germination during this period. At the end of summer, germination and start of bloom are possible because of higher temperature and higher nutrient. During the SE monsoon, blooms are maintained by the stable subsurface water and less vertical mixing (Velasquez et al. 1997)".

From a recently concluded study (Villanoy et al., in press), two main sources exist for bloom formation in Manila Bay. One source is fed by the cyst beds in the west (Bataan), which has a relatively higher density of cysts (Corrales and Crisostomo, 1996); the cysts are advected along the west- northwest coast (Bataan-Bulacan). The other source is located in the southeast (Cavite) which feed the east-southeast area (Parañaque-Cavite). Bloom simulation using bio-physical parameters have shown that the direction of the bloom is almost always along the downwind direction. The dispersal distribution increases if the cells are found higher in the water column. Smayda (2002) has concluded that *Pyrodinium* together with its other toxic closer relatives like *Alexandrium* are adapted for "entrainment and dispersal with coastal currents".

Villanoy et al. (in press) also found that "seasonal" variations of temperature and salinity reflect the combined effects of convection and water column stability that can control vertical migration of plankton and other parameters essential to its growth. Wind forcing and tidal currents make possible cyst resuspension and in addition, the wave field during the southwest monsoon indicates that it can contribute to the bottom velocity that dominates particularly in the southeast side.

Competition with other dinoflagellates species like the bigger and heterotophic *Noctiluca scintillans* (Figure 7) has been invoked to be one of the reasons why after 1998, *Pyrodinium* blooms have not been included in Manila Bay (Azanza and Miranda, 2001; Hansen et al., 2004). Since 1998 up to the present, *Noctiluca scintillans* has dominated other phytoplankton species from June to September when *Pyrodinium* used to bloom.

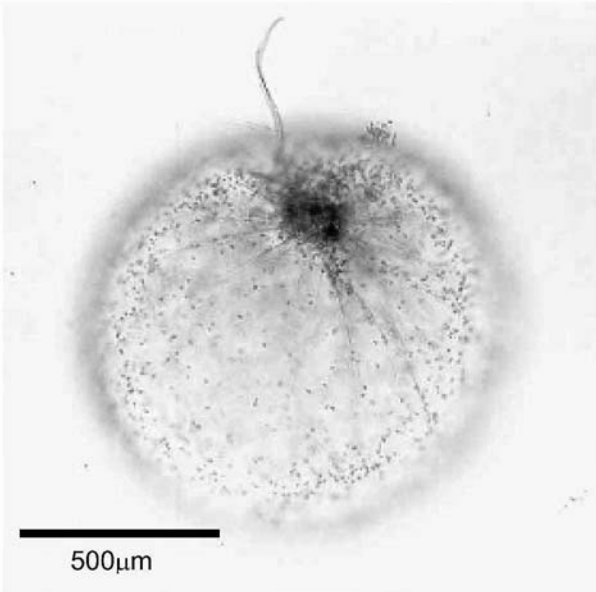

Figure 7. *Photomicrograph of the heterotrophic dinoflagellate, Noctiluca scintillans.*

2.2.2. Phytoplankton Succession

The earliest record of a harmful phytoplankton occurrence in Manila Bay was that of a *Peridinium* bloom in 1908 (Smith, 1908 as cited by Maclean, 1984). The first published account of the phytoplankton succession in Manila Bay was based on a study done from 1997 to 1999. A listing of major phytoplankton species found in the Bay is shown in Table 1. Eight stations in the western side (Bataan) and eastern side (Cavite) were sampled during the three seasons of each year for phytoplankton abundance and species composition (Azanza and Miranda, 2001). There was a pronounced increase in the dinoflagellate population in waters off Bataan from northeast to the southwest monsoon. This pattern was less obvious in Cavite. A bloom of the heterotrophic *Noctiluca scintillans* was observed in the two sites after the bloom of *Pyrodinium* in 1999. This organism was found to be present in Bataan and Cavite during the three-year study period. Later, in a series of experiments, it was shown that *N. scintillans* can feed on *Pyrodinium* (Hansen et al., 2004) and results have also suggested that *N. scintillans,* when it occurs at bloom concentration in nature (1-10 cells mL^{-1}), may have a significant impact on the bloom dynamics of *Pyrodinium bahamense* var. *compressum*.

Gymnodinium catenatum and *Alexandrium* spp. are the other HAB species recorded in the bay, but no bloom and toxicity have been attributed to them. Phytoplankton composition in the bay, which showed strong seasonal

Table 1. *Phytoplankton species identified in Manila Bay, Philippine (after Azanza and Miranda, 2001).*

Family	*Species*
Dinoflagellates	
Gonyaulacaceae	*Alexandrium* sp.
	Goniodoma sp.
	Gonyaulax spinifera
	Pyrodinium bahamense var. *compressum*
Ceratiaceae	*Ceratium breve*
	Ceratium carriense
	Ceratium furca
	Ceratium fusus
	Ceratium macroceros
	Ceratium trichoceros
	Ceratium tripos
	Ceratium vultur
Dinophysiaceae	*Dinophysis caudata*
	Dinophysis rotundata
	Dinophysis miles
Gymnodiniaceae	*Cochlodinium* sp.
	Gymnodinium catenatum
	Gymnodinium sanguinaeum
	Gyrodinium sp.
	Gyrodinium spirale
	Polykrikos kofoidii
Prorocentraceae	*Prorocentrum lima*
	Prorocentrum micans
Noctilucaceae	*Noctiluca scintillans*
Peridiniaceae	*Diplopsalis* sp.
	Protoperidinium sp.
	Protoperidinium conicum
	Protoperidinium depressum
	Protoperidinium divergens
	Protoperidinium leonis
	Protoperidinium pellucidum
Pyrocystaceae	*Pyrocystis* sp.
Pyrophacaceae	*Pyrophacus steinii*
Diatoms	
Chaetocerotaceae	*Bacteriastrum* spp.
	Chaetoceros spp.
Coscinodiscaceae	*Coscinodiscus* spp.
Rhizosoleniaceae	*Rhizosolenia* spp.
	Guinardia spp.
Melosiraceae	*Melosira* spp.
Naviculaceae	*Navicula* spp.
Bacillariaceae	*Pseudonitzschia* spp.
Thalassiosiraceae	*Skeletonema* spp.
	Thalassiosira spp.
	Thalassionema spp.
	Thalassiothrix spp.
Cyanobacteria	
Oscillatoriaceae	*Trichodesmium* spp.
Flagellates	
Chatonellaceae	*Chatonella* sp.
Dictyochaceae	*Dictyocha* sp.

variability with diatoms dominating the northeast monsoon and dinoflagellates during the southwest monsoon, could be attributed to physico-chemical changes. The yearly bloom of *Pyrodinium* from 1988 to 1998 appears to be caused by long-term changes in the water column. Long-term data on nutrient concentration may explain the shift of dinoflagellate dominance from autotrophic to heterotrophic types as well as the change of zooplankton size class towards the juveniles (Azanza et al., in prep.). More detailed and longer-term phytoplankton and zooplankton studies in this bay are needed to understand better their roles on HAB's in the area.

2.3. Subsidence and Groundwater Extraction

Besides issues of shoreline erosion described by Jacinto et al. (2005), another potentially problematic situation in Manila Bay is subsidence. The tide gauge record in Manila South Harbor indicate that the relative sea level rose about 2 mm y^{-1} from 1902 to the early 1960's, essentially the rate of eustatic rise. The rate increased by an order of magnitude from 1963 to 1980, to 2.35 cm y^{-1}, a trend that correlates to the increase in groundwater withdrawal over the same period (Siringan and Ringor, 1998).

Over the coastal towns of the Pampanga delta plain, north of Manila Bay, the rate of subsidence was estimated from the emergence of water wells and changes in flood levels due to high tide alone. The emergence of wells gave typical rates that were between 2–4 cm y^{-1} with an average of 2.5 cm y^{-1} (Siringan and Rodolfo, 2003). The rates derived from the emergence of wells are consistent with the estimates based on accounts that some areas that stood above tide levels in the 1970s are now frequently flooded, to a depth of a meter, during high tide.

A consequence of subsidence is that floods during the rainy season have also become more frequent, higher, and takes longer to subside. In response, many roads and houses on the delta plain have been raised more than once in the past few years. Agricultural lands have been transformed into aquaculture ponds.

2.4. Overexploitation of Fishery Resources

Among the problems affecting the fishers of Manila Bay is the apparent decrease in fish catch and the shift from high value to low value fish products. This is manifested in the decline of demersal fisheries in the bay.

The trawl fishery in Manila Bay in the second half of the 1950s was assessed to have reached its maximum sustainable yield, such that an increase in the number of fishing vessels resulted in a decrease in their annual average landings (Ronquillo et al., 1960). A recent assessment by Armada (2004) suggests that the decline in demersal biomass is not the only effect of excessive fishing of these resources by trawls and other gears but also results in changes in species composition. An increase in the catch of invertebrates, shrimp and squid, and a decrease in large-size fish species was already noted in the early 1980's (Silvestre et al., 1987). For instance, invertebrates were not part of the catch in trawl surveys done in the 1940's (Manacop, 1950) but four decades later invertebrates contributed 25% of the catch. The fish catch, classified into families, showed a significant change in composition (Armada, 2004).

This is apparently an illustration of ecosystem overfishing described by Pauly et al. (1989) where "major changes in catch composition include an increase in abundance of squids, shrimps and small pelagic species like herrings and anchovies; disappearance of turbots and lactarids; and substantial declines in the abundance of large commercially valuable species like snappers, sea catfish and Spanish mackerels."

Silvestre et al. (1987) showed the need to reduce the fishing effort on the demersal stock in Manila Bay to one-third of the 1983-1984 level to be able to attain an economic rent of US$1.5-4.8 million. Further, Armada (1994) showed that the demersal trawlable biomass of Manila Bay decreased tenfold, from 4.61 t km^{-2} in 1947 to 0.47 t km^{-2} in 1992.

2.5. Habitat Conversion and Degradation

A recently conducted Refined Risk Assessment of Manila Bay (PEMSEA, 2004) provided an overview of the issues related to habitat conversion and degradation and is particularly relevant:

> "**Mangroves.** The assessment of mangroves done through a Resource and Ecological Assessment of Manila Bay (BFAR, 1995) indicated that there were around 54,000 hectares of mangrove forests in Manila Bay at the turn of the century (1890). Subsequent estimates showed that, 100 years later (in 1990), only 2,000 hectares were left. This was further reduced in 1995 to 794 hectares. The decrease in mangrove cover was caused by clearance for conversion into aquaculture and salt beds, land reclamation for human settlement, industrial development and other development activities. Physical removal for fuel wood was also cited as a cause of decline."
>
> "**Coral Reefs.** No long-term studies have been conducted on coral reefs in Manila Bay, although there are anecdotal reports of the significant decline of this habitat. For instance, a large section of the reefs at the entrance of the bay, particularly the *Acropora sp.* has been damaged (BFAR, 1995). In that same study conducted in the early 1990's, the live coral cover estimates ranged from 20% in Mariveles and Corregidor Island up to 80% in Cavite (Limbones Cove). There were about 14 families of hard corals, one family for soft coral composed of 38 genera and 53 species that were considered in ecologically poor condition. Structure-wise, those found were of the fringing type composed of generally encrusting forms and massive in habit with no solid stands, and are mostly dispersed and occurring in patches. Most colonies were found to be young and small."
>
> "The sparse distribution of coral reefs, mostly occurring in patches and in young and small colonies, in the bay could be attributed to various factors including physical destruction (blast fishing), cyanide/poison fishing in the reef area, siltation, gathering, use of fishing gears and attachments (trawls and motorized push nets), increase in boat anchorage, and pollution from metals, pesticides and oil spills."

3. OPPORTUNITIES

3.1. Legislation

The Philippines is replete with environmental laws, with about 118 environment related laws enacted by the Philippine government (Oposa, 1996). Many of these laws have a direct bearing on Manila Bay's resources and inhabitants. The 1987 Constitution provides the primary legal basis for environmental protection, particularly Article II, Section 16 which provides that "the State shall protect and advance the right of the people to a balanced and healthful ecology in accord with the rhythm and harmony of nature."

However, implementation of the laws is beset with many problems. Among them, and as is common in many developing countries, are conflicts of jurisdiction at the central level between the DENR and other national agencies; and at the provincial levels and between the DENR and provincial governments (these are the local government units, or LGU's). Problems of enforcement of environmental laws are also caused by lack of political will (especially by LGU's), jurisdictional overlaps and confusion, insufficient understanding of legislation, and lack of financial resources.

Certain initiatives that take advantage of current laws, however, have been taken recently which may help make a difference for Manila Bay.

In what was apparently a landmark case, the Regional Trial Court of Imus, Cavite (a province located on the southern part of Manila Bay) issued in 2002 a historic judgment ordering 12 government agencies to clean up Manila Bay. The Court directed the Department of Environment and Natural Resources (DENR) as the lead agency to "prepare a consolidated and coordinated action plan for the restoration of Manila Bay to make it fit for swimming and other forms of contact recreation."

Represented by environmental law students of the University of the Philippines (UP), residents of Manila Bay filed the action, using an antiquated and little-known law requiring "concerned government agencies to clean up a polluted body of water."

During trial, it was proved that while the standard for fecal coliform for a swimmable body of water is only 200 MPN L^{-1}, the waters in Manila Bay contain as much as 100,000 MPN L^{-1}.

In yet another development in the country, an environmental Ombudsman was created in February 2004 to monitor and prosecute cases involving environmental violations in the country. "The Ombudsman is an office created by the Constitution to investigate acts of any person or public official, which may be "illegal, unjust, improper, or inefficient."

It may also investigate any public official or government office in order "to order, prevent, and correct any abuse or impropriety in the performance of duties."

The special investigative arm of the Ombudsman will monitor the compliance of public officials with the protection of the country's environment and natural resources, and consider complaints against public officials who fail to enforce the country's environmental laws or to act on such complaints.

In May 2005, and in one of the first attempts to take advantage of the recently created Environmental Ombudsman, the Philippine Bar Association filed complaints against three mayors of Metro Manila for failing to enforce the provisions of Republic Act 9003 (Ecological Solid Waste Management Act of 2000) that calls for the establishment of material recovery facilities in every barangay or cluster of barangays where waste segregation, composting and recycling could be made. Fines of as much as P500,000 (~USD10,000), imprisonment of up to two years and preventive suspensions may be slapped on the mayors for ignoring the law.

No one has yet been imprisoned as a result of the cases filed or decisions made by the courts, but the message is being sent to government officials, at the national and local levels, that they would be taken to task for their failure to implement and abide by environmental laws.

3.2. Sewage Treatment

Undoubtedly, the discharge of largely untreated sewage and sewage sludge coming from domestic and industrial sources has contributed significantly to the deterioration of Manila Bay. Sewage generated within Metro Manila is managed by household or communal septic tank systems and, only to a lesser extent, sewerage collection and treatment systems.

Supported by funding from the World Bank and the Asian Development Bank (ADB), and following the privatization of the Manila Waterworks and Sewerage System (the Government-owned water and wastewater utility of Metropolitan Manila) in the late 1990's, the sanitation strategy for Metro Manila was formulated to radically expand the septage management program in concert with the rehabilitation and new construction of small conventional sewage treatment plants serving medium and high-rise housing establishments, which are the current trend in the housing industry.

Following this strategy, one of the two private concessionaires, the Manila Water Company, Inc. (MWCI) will expand its sanitation services so that all cities and municipalities in the concession area will be 100% covered with either sewerage and/or sanitation. The World Bank and the ADB are also integrally involved in implementing this strategy through the Manila Second Sewage Project (MSSP), the Manila Third Sewage Project (MTSP), and the Pasig River Rehabilitation Project (PRRP).

Started in 1998, the World Bank's MSSP and MTSP activities in the MWCI area over the next few years have involved the following: (1) an improved sewage collection and disposal system, which includes the provision of septage desludging trucks and construction of two centralized septage treatment plants within the concession area; (2) rehabilitation of the Makati sewerage system and of the Magallanes STP and; (3) construction of compact STPs for individual housing developments in Quezon City currently served by communal septic tanks (IFC Projects,
http://www.ifc.org/ifcext/spiwebsite1.nsf/0/4021913ed665ec6b85256e6700631f57?OpenDocument).

The ADB's PRRP includes a wide range of measures to improve the water quality in the Pasig River and environmental conditions in the vicinity. Activities to be completed include construction of a third septage treatment plant, and provision of over 30 additional septic tank desludging tankers.

The three sewage treatment plants currently operated by MWCI continue to produce treated effluent that meets discharge limits imposed by the DENR according to monitoring carried out by both MWCI and DENR. As part of the MSSP, construction of 36 compact sewage treatment plants is ongoing, and MWCI was to have commissioned these plants by 2004.

3.2.1. Manila Sewarage Projects

The Manila Second Sewerage Project (MSSP) is intended to help the government improve the quality of sanitation services in Metro Manila and enable the Metropolitan Waterworks and Sewerage System to radically expand its septage management program and establish the conditions needed for medium-term low-cost improvement of sewerage services in Metro Manila and; to reduce pollution in Metro Manila waterways and in Manila Bay, thereby reducing the health hazards associated with human exposure to excreta; see (http://www.lguportal.org/lgupor/Products/Ongoing/water-mssp.htm).

The project includes the building of three large loading stations for septage disposal at sea; building of a pilot septage treatment plant; rehabilitation of the Central and the Ayala Sewerage Systems, the Ayala treatment plant, and construction of about 10,000 individual sewer connections and; provision of maintenance equipment, tools, spare parts and laboratory instruments; provision of community and sanitation through on-site treatment.

The Manila Third Sewerage Project (MTSP) aims to increase the coverage and effectiveness of sewerage service delivery in participating areas of Metro Manila through an integrated approach involving septage management, sewage management, and heightened consumer awareness of water pollution problems and their solutions. It will establish the financial and technical viability of new approaches for sewage management in Metro Manila. MTSP will improve sanitation and sewer coverage in the East Zone, from less than 8 percent in 2004 to an estimated 30 percent in 2010. The project consists of three components: (i) Sewage Management, (ii) Septage Management, and (iii) Institutional Strengthening.

3.3. The Manila Bay Environmental Management Project

The Manila Bay Environmental Management Project (MBEMP) is a local component of the Regional Programme on Building Partnerships in Environmental Management for the Seas of East Asia (PEMSEA), a UNDP-GEF funded project with the Department of Environment and Natural Resources (DENR) as the host institution in the Philippines and the International Maritime Organization (IMO) as the executing agency. The project was established to help address the numerous environmental problems confronting Manila Bay. The objectives of the project are:

> "to apply environmental risk management processes in addressing transboundary environmental issues and its effects in the coastal and watershed areas under stress; and to establish Manila Bay as a pollution hotspot/demonstration site by applying Integrated Coastal Management (ICM) for the systematic and effective management and use of land and water resources".

The Project consists of various components, namely:

- Environmental Risk Assessment [Initial Risk Assessment, Refined Risk Assessment (IRA, RRA)]
- Environmental Investments (EI)
- Integrated Environmental Monitoring Program (IEMP)
- Manila Bay Coastal Strategy (MBCS)
- Civil Society Participation (CSP)

- Operational Plan for Manila Bay Coastal Strategy (OPMBCS)
- Integrated Information Management System (IIMS)
- Manila Bay Information Network (MBIN)
- Coastal Land and Sea Use Zonation Plan
- Manila Bay Oil Spill Contingency Planning
- Environmental and Resource Valuation for Manila Bay Area (ERVMB); and,
- Institutional Studies

So far, the project has developed the Manila Bay Coastal Strategy, which provides a comprehensive environmental management framework, targeted outcomes and a series of action programs involving the participation of both government and non- government sectors.

A report entitled "Manila Bay: Refined Risk Assessment" was completed in 2004 and presents the findings and outcome of the assessment of Manila Bay undertaken by an inter-agency, multi-disciplinary Technical Working Group (TWG). The report draws attention to the priority risks and areas of concern in the bay area.

In addition, the project is beginning to implement an Operational Plan for Manila Bay (OPMB) in partnership with national government agencies and stakeholders both from public and private sectors; see
http://www.emb.gov.ph/mbemp/mb_ovrview.htm.

4. CONCLUSION

Manila Bay has a wide range of environmental problems that need to be addressed - from land-based and sea-based sources of pollution to harmful algal blooms, subsidence and groundwater extraction, overexploitation of fishery resources, and habitat conversion and degradation. However, there are reasons to be optimistic. There is greater accountability expected of public officials vis-a-vis environmental laws, significant and increasing infrastructure investments to treat and reduce domestic sewage discharges into the bay, the implementation of the Manila Bay Environmental Management Project, and the adoption the concept and practice of ICM by local government units and communities around Manila Bay.

The Manila Bay Coastal Strategy's response to the many issues confronting the bay is articulated as follows:

> "Manila Bay stakeholders are partners in: raising public awareness and participation, protecting human welfare, ecological, historical, cultural and economic features, mitigating environmental risks, implementing effective policies and environment management and governance, and developing areas and opportunities in a sustainable manner."

Time will tell if the envisioned response will be pursued and continued so that Manila Bay will revert to be a clean, safe, wholesome, and productive ecosystem for the present and future generations.

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CHAPTER 20

CARBON FLUX THROUGH BACTERIA IN A EUTROPHIC TROPICAL ENVIRONMENT: PORT KLANG WATERS

CHOON–WENG LEE AND CHUI–WEI BONG

1. INTRODUCTION

The term "microbial loop" was introduced more than two decades ago (Azam et al., 1983) to describe the importance of the microbial food web on the recycling and mineralization of organic matter in aquatic habitats. As bacteria are the most abundant component responsible for the transformation of organic matter (Cole et al., 1988), bacterial production (BP) becomes a key process in dissolved organic matter (DOM) flux. Heterotrophic and autotrophic processes are the two most fundamental metabolic processes in aquatic ecosystems. In oligotrophic systems, planktonic primary production (PP) is the main source of DOM, and there is tight coupling between BP and PP (Williams, 1998). However in coastal waters, terrigenous DOM is often suggested as an alternative source of organic matter that could be utilized by bacteria, and to explain the "uncoupling" that sometimes occurs between PP and BP (Tranvik, 1992). Understanding these auto- and heterotrophic processes is central to the study of biogeochemical cycles especially the carbon (C) cycle.

Another essential information for C cycle study is the bacterial growth efficiency (BGE). BGEs have great relevance to C dynamics because most aquatic metabolism is microbial (del Giorgio et al., 1997). BGE is an important parameter to evaluate the fate of organic carbon inputs, and whether the bacteria act as a link (recyclers) or a sink (mineralizers) depends on the BGE (del Giorgio and Cole, 2000). BGE is essentially the ratio of net production over bacterial carbon demand (BCD), where BCD has been measured as BP plus bacterial respiration (BR) (Lee et al., 2002) or DOM utilization (Amon and Benner, 1996) or both (Cherrier et al., 1996). Although BP data are available, there are relatively few BR and BGE data, and most studies use a common BGE value (i.e. 30%) for their carbon flux calculations (e.g. Bano et al., 1997; Lee et al., 2001b). However studies of temperate ecosystems have shown that BGE varies in time and space (e.g., Rivkin and Legendre, 2001; Biddanda and Cotner, 2002; Lee et al., 2002), and the independent measurement of BGE is

E. Wolanski (ed.), The Environment in Asia Pacific Harbours, 329–345.

required to avoid uncertainty in conversion factors. Furthermore, estuarine systems (e.g. Port Klang waters; Figure 1) receive inputs of allochthonous organic matter from terrestrial runoff, and BGE becomes a crucial parameter to evaluate the fate of allochthonous carbon inputs. Other than the work carried out by Pradeep Ram et al. (2003), very little information is available on BGEs from tropical aquatic systems.

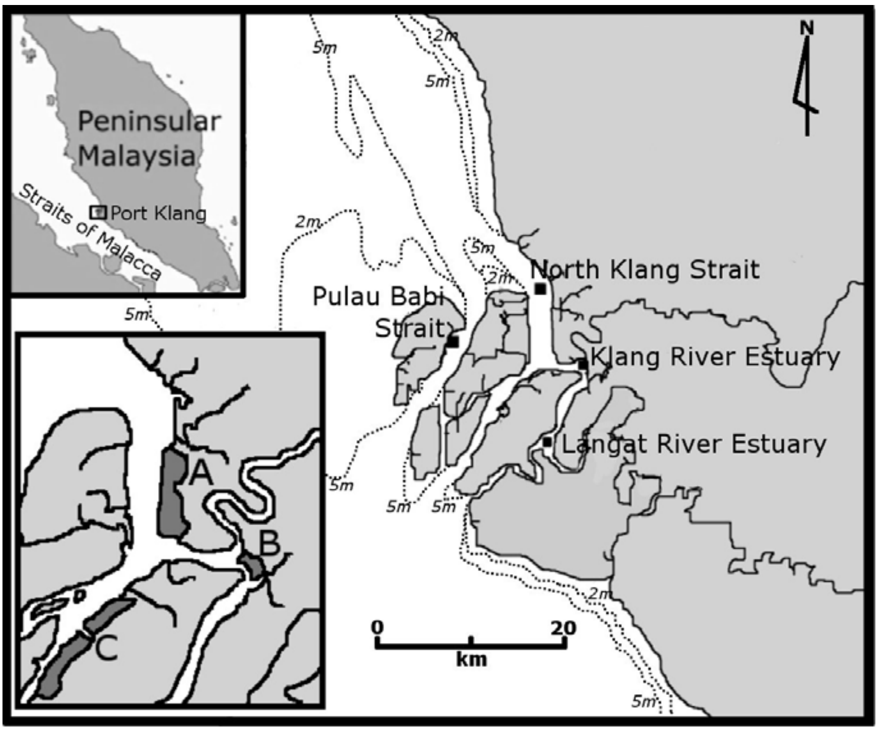

__Figure 1.__ Map showing the location of Port Klang, and the sampling stations in this study. Isobaths for 2 and 5 m depths are delineated by dotted lines. Lower left inset shows the (A) Northport, (B) Southport and (C) Westport of Port Klang.

Research on the microbial food web in Malaysian waters is limited to culturable-specific bacteria and its participation in the nitrogen cycle (Shaiful et al., 1986; Thong et al., 1993). Most of the scientific literature focuses on the biology and ecology of penaeid prawn and fish communities in mangroves (e.g. Chong et al., 1990 and 2001). In Malaysia, data on metabolic processes are scarce (Alongi et al., 2003). This study was part of a research initiative to understand the microbial food web in coastal Malaysian waters. It aims to determine the significance of bacterial processes in material and energy fluxes in different aquatic environments. In this paper, microbial process rates including BGE values from a eutrophic tropical aquatic system are reported, as well as the amount of BP grazed, and how much BP was transferred onto higher trophic levels. A major goal of this research was to evaluate temporal variation in both autotrophic and heterotrophic processes in Port Klang waters, and to determine how these processes affect Port Klang water quality.

2. SITE DESCRIPTION

Port Klang (Figure 1) is a multipurpose Malaysian gateway port located strategically mid-way on the west coast of Peninsular Malaysia overlooking the Straits of Malacca. It offers the first mainline port of call eastbound on the Europe-Asia leg and last port of call westbound on the Asia-Europe leg. Port Klang began more than 100 years ago as a small railway port. Consistent with the rapid growth of the Malaysian economy in the 1970s–1990s, there was a rapid expansion of demand for port facilities at Port Klang. Port Klang now comprises Northport (covering an area of 241 ha), Westport (510 ha) and Southport (48 ha, see Figure 1). Port Klang handled a total of 5.2 million twenty-foot equivalent units (TEUs) in 2004, 12th in the World Port Rankings (Barrock, 2005).

3. METHODS

Sampling was carried out regularly from September 2004 until February 2005 at the Klang River Estuary station (03º00'04 N, 101º23'24E, Figure 1). This station is located at the mouth of the Klang River, near to the Southport. Seawater samples were collected about 0.1 m below seawater surface, and kept in a cooler box until processing within three hours. *In-situ* measurement of salinity was carried out using a salinometer (Atago S/Mill-Σ, Japan) whereas pH and temperature were measured with a pH meter (Jenway 3071, UK). For dissolved oxygen (DO) determination, samples were collected in 50 ml DO bottles, and fixed immediately with manganous chloride and alkaline iodide solutions. DO concentration was then determined by the Winkler titration method (Grasshoff et al., 1999). The theoretical 100% saturation value at the time of sampling was also calculated according to Weiss (1970). One sample for the determination of microbial abundance was obtained each time, and preserved with filtered (0.2 μm pore size) glutaraldehyde (1% final concentration).

3.1. Chemical parameters

In the laboratory, seawater sample was filtered through pre-combusted (500 °C for 3 h) Whatman GF/F filters, and the filtrate was kept frozen (–20 °C) until nutrient analysis. Filters for chlorophyll *a* (Chl *a*) and total suspended solids (TSS) were also kept frozen until analyses. Both dissolved inorganic nutrients [nitrate (NO_3), nitrite (NO_2), ammonium (NH_4), phosphate (PO_4) and silicate (SiO_4)] and Chl *a* analyses were carried out according to Parsons et al. (1984). Chl *a* was extracted overnight with 90% ice-cold acetone, and its absorbance at different wavelengths (using the trichromatic method) was measured with a spectrophotometer (Beckman DU7500i, US). TSS was measured as the filter weight increase after drying (70 ºC until no more weight loss). The same filter was later combusted in a microwave furnace (CEM MAS7000, US), and the weight loss after combustion was calculated as particulate organic matter (POM).

3.2. Microbiological parameters

Bacterial abundance was determined by epifluorescence microscopy on samples filtered onto a black polycarbonate filter (0.2 µm pore size), and then stained with 4'6-diamidino-2-phenylindole (DAPI) (0.1 µg l^{-1} final concentration) for 7 min (Kepner and Pratt, 1994). More than 300 cells or a minimum of seven fields were counted for each sample using an epifluorescence microscope (Olympus BX60, Japan) with the U–MWU filter cassette (excitor 330–385 nm, dichroic mirror 400 nm, barrier 420 nm). For protist, 10 ml of sample was filtered onto a black polycarbonate filter (0.8 µm pore size), and then stained with the fluorochrome primulin (Bloem et al., 1986). Observation was also carried out using the U–MWU filter cassette. The abundance of phototrophic picoplankton (PPico) was determined by filtering 5 mL of sample onto a black polycarbonate filter (0.2 µm pore size). No fluorochrome was used, and the autofluorescence of the PPico was observed using the U–MWG filter cassette (excitor 510–550, dichroic mirror 570 nm, barrier 590 nm). For the determination of bacterial biovolume, measurements of bacterial diameter and length were obtained using a digital imaging system, analySIS® version 3.2 (Soft Imaging System, Germany). These measurements were then used to calculate the bacterial biovolume. Bacterial biovolume was measured either as a sphere $[(\pi A^3)/6]$ or an ellipsoid $[(\pi AB^2)/6]$ where A is the diameter or length, and B is the width of the cell (Kellar et al., 1980). To obtain a carbon conversion factor, the bacterial biovolume was converted into carbon biomass using an equation derived from Simon and Azam (1989):

$$\text{fg C cell}^{-1} = 75.9(\mu\text{m}^{-3}\text{cell}^{-1})^{0.59} \quad (1)$$

3.3. Bacterial production rate

Bacterial specific growth rate (µ) was measured using a dilution culture method (Lee et al., 2001a). Sample was filtered through pre-combusted Whatman GF/F filters, and diluted five-fold with 0.2 µm filtered sample. Incubation was carried out for 12 h in the dark at *in-situ* temperature. Sub-samples were collected regularly at 4 h intervals, and the bacterial abundance increase over incubation time was measured. µ was calculated using the least-squares method as the slope of the regression analysis of natural logarithmic bacterial cell increase over time. Bacterial production (BP) was then estimated by multiplying µ by the bacterial abundance (BA): BP=µBA. At each sampling, an additional batch of incubation was carried out to examine whether any nutrient limitation occurred. Before incubation was started, the sample was enriched with nutrients at final concentrations of 60 μM Glucose, 15 μM NH_4Cl and 2 μM KH_2PO_4.

3.4. Bacterial respiration rate

To measure bacterial respiration (BR), seawater sample was filtered through pre-combusted Whatman GF/F filters to remove particles and bacterial grazers, and then siphoned into acid-washed 50 ml DO bottles. These bottles were incubated in the dark at *in-situ* temperatures for 12 h. Change in DO concentration was measured in sets of five DO bottles in a four time-point analysis. BR was calculated by the least-squares method as the rate of DO decrease with incubation time.

3.5. Bacterial growth efficiency

In this study, bacterial growth efficiency (BGE) was determined by comparing gross with net bacterial production where the gross bacterial production or bacterial carbon demand (BCD) was measured as bacterial production (BP_{resp}) plus bacterial respiration (BR) (e.g. Lee et al., 2002):

$$BGE = BP_{resp}/(BP_{resp}+BR) \qquad (2)$$

BP_{resp} was determined by measuring the increase in bacterial abundance in another set of bottles incubated simultaneously.

3.6. Protist grazing rate

In order to determine the protist grazing rate on bacteria, the size-fractionation method was used. Seawater sample was size-fractionated into both <0.7 μm and <20 μm fractions. These were then incubated for 12 h, and changes in both bacterial and protist abundance in each fraction was determined. The <0.7 μm fraction was essentially a grazer-free environment for the bacteria, and represented the bacterial growth rates without any grazing pressure ($\mu_{0.7}$) whereas the <20 μm fraction contained both bacteria and protist. The bacterial growth rate in the <20 μm fraction (μ_{20}) presumably represented the product of both growth and grazing (McManus, 1993). Bacterial activity was assumed the same for all the fractions, and grazing rate (h^{-1}) was then estimated using the following equation: $\mu_{0.7}-\mu_{20}$. Protist grazing rate was also expressed as bacteria eaten per protist per h (cell protist^{-1} h^{-1}).

3.7. Gross Primary Production and Community Respiration

To measure gross primary production (GPP) and community respiration (CR), seawater samples were siphoned into acid-washed 50 ml DO bottles. These bottles were then incubated under light (for GPP) and dark (for CR) conditions at *in-situ* temperatures for 12 h. Change in DO concentration was measured in sets of five DO bottles in a four time-point analysis. The rate of DO increase or decrease was analyzed using the least-squares linear regression method. CR was the rate obtained from "dark bottles" whereas GPP was calculated as the rate of DO change in "light bottles" minus "dark bottles". A photosynthetic quotient of 1.2 and a respiratory quotient of 1.0 was used in the conversion to carbon units (Parsons et al., 1984). For BR, GPP and CR, the multipoint and replicate measurements of DO over a short incubation time is able to yield realistic estimates of respiration (Pomeroy et al., 1994; Biddanda and Cotner, 2002).

4. RESULTS

Surface seawater temperature was relatively invariant, ranging between 29.2 and 30.1 (Coefficient of Variation, *CV*=1%) whereas salinity varied over a wider range (16–29, *CV*=22%) (Figure 2A). Salinity was lowest in late December 2004 and January 2005. pH range was 7.08–7.86 (*CV*=4%), and correlated positively with salinity (R^2=0.678, *n*=6, *p*<0.05). DO concentration was generally low (<200 μM, *CV*=42%) (Figure 2B), and decreased to 37 μM in February 2005. Percentage DO saturation can be used to indicate physiological stress (Breitburg, 2002). Although DO saturation value is dependent on both temperature and salinity (Weiss, 1970), temperature plays a more important role. As temperature was stable, DO saturation did not vary much (200–204 μM, Figure 2B). Using both DO saturation and *in-situ* DO concentration, the percentage DO saturation calculated ranged 18–97%. TSS or particulate >0.7 μm size was consistently high, and ranged 260–290 mg l^{-1} (*CV*=4%) (Figure 2C). In comparison to TSS, POM was more variable, fluctuating between 6–13 mg l^{-1} (*CV*=33%). Chl *a* averaged 3.58±1.86 μg l^{-1}, and was between 2.06–3.64 μg l^{-1} throughout most of the sampling period but doubled to 7.16 μg l^{-1} late December 2004.

Figures 3A and 3B show the nutrient concentration measured in this study. Of the three nitrogen species measured, NH_4 (Figure 3A) was consistently highest (>67% of dissolved inorganic nitrogen, DIN). NH_4 (mean ± S.D.=11.76 ± 9.06 μM) fluctuated within 5.46–10.39 μM for most months but increased nearly three fold to 29.78 μM early December 2004. Relative to NH_4, NO_3 (2.78±1.99 μM) and NO_2 (2.39±1.93 μM) made up only 18 and 15% of DIN, respectively. In our study, PO_4 varied nearly ten fold (0.41–5.68 μM), and was also highest early December 2004 (5.68 μM) whereas SiO_4 fluctuated the least (*CV*=39%), and was always >3.51 μM (Figure 3B).

Microbial abundance showed less than one order fluctuation throughout the sampling period (Figure 3C). Bacteria were more than two orders higher than other microbes, and dominated the microbial community. Bacterial abundance ranged 2.5–9.8×10^6 cells mL^{-1} whereas PPico and protists ranged 1.2–6.4×10^4 cells mL^{-1} and 1.3–3.5×10^3 cells mL^{-1}, respectively. Bacterial abundance was relatively stable (2.5–4.0×10^6 cells mL^{-1}) but more than doubled in January 2005 to 9.8×10^6 cells mL^{-1}. For the determination of bacterial biovolume (n=650), the carbon content per bacterium ranged 4.6–97.2 fg C $cell^{-1}$, and the average carbon content per cell (32.8 fg $cell^{-1}$) was used as our constant carbon conversion factor.

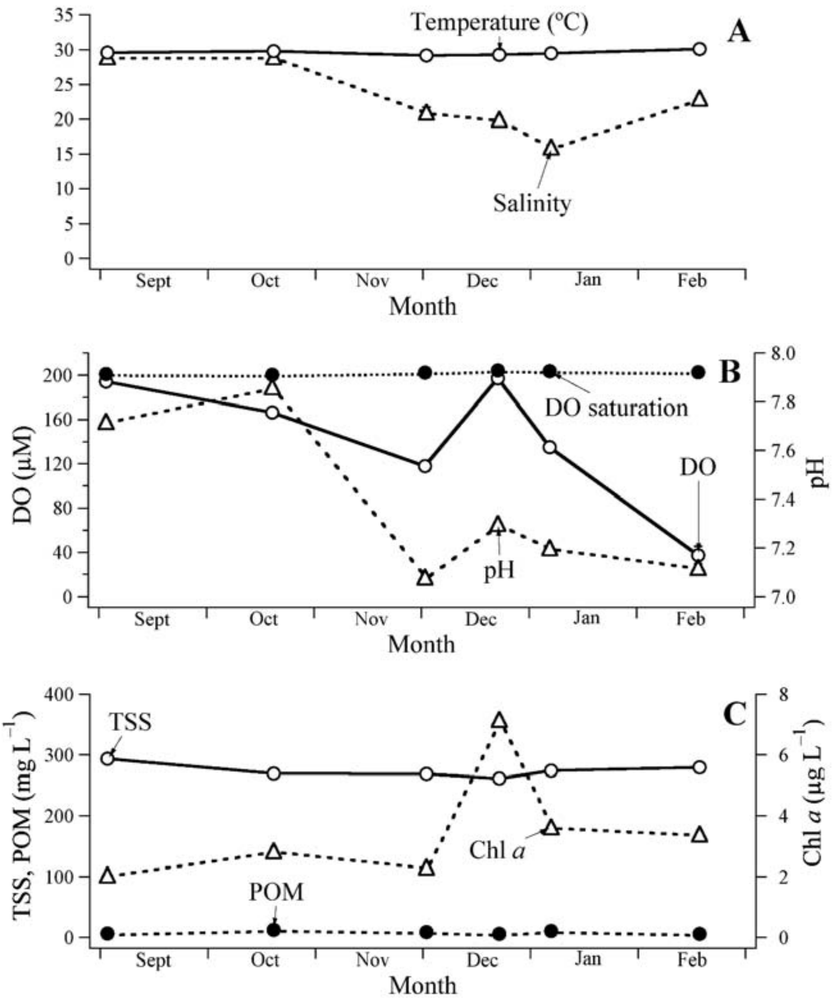

Figure 2. *Temporal variation of (A) temperature and salinity; (B) dissolved oxygen (DO), DO saturation and pH; (C) total suspended solids (TSS), particulate organic matter (POM) and Chlorophyll a (Chl a) at the Klang River Estuary throughout the sampling period. Error bars (±Standard Deviation, S.D.) are shown for DO, except where they are smaller than the symbols.*

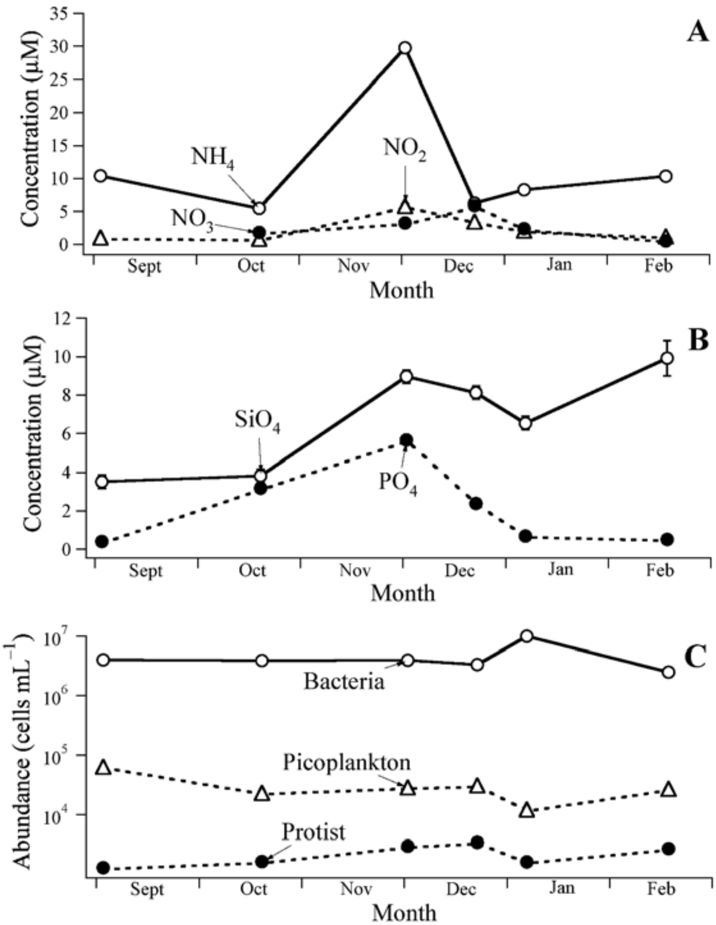

Figure 3. *Temporal variation of (A) NH_4, NO_3 and NO_2; (B) SiO_4 and PO_4; (C) bacterial abundance, protists and phototrophic picoplankton at the Klang River Estuary throughout the sampling period. Error bars (±Standard Deviation, S.D.) are shown for NH_4, NO_3, NO_2, SiO_4 and PO_4, except where they are smaller than the symbols.*

In this study (Figure 4A), GPP fluctuated between 103–265 µg C L^{-1} h^{-1} except late December 2004 when it was highest (407 µg C L^{-1} h^{-1}) and lowest in February 2005 (6 µg C L^{-1} h^{-1}). CR ranged 0.931–7.301 µM O_2 h^{-1} or its carbon equivalent of 11–88 µg C L^{-1} h^{-1}, and was highest late December 2004. CR was lower than GPP on all occasions except February 2005. Bacterial growth rate or µ ranged 0.086–0.221 h^{-1}, and BP ranged 10–71 µg C L^{-1} h^{-1} (Figure 4B). Both µ and BP were highest in January 2005. We also found that adding nutrients did not significantly increase µ (0.098–0.249 h^{-1}) (Student's t-test: t=0.357, df=10, p>0.70). Of this BP, each protist consumed about 17.8–71.9 bacteria h^{-1}. Protist grazing was about one order lower than BP, ranging 1.5–5.8 μg C L^{-1} h^{-1} (Figure 4C).

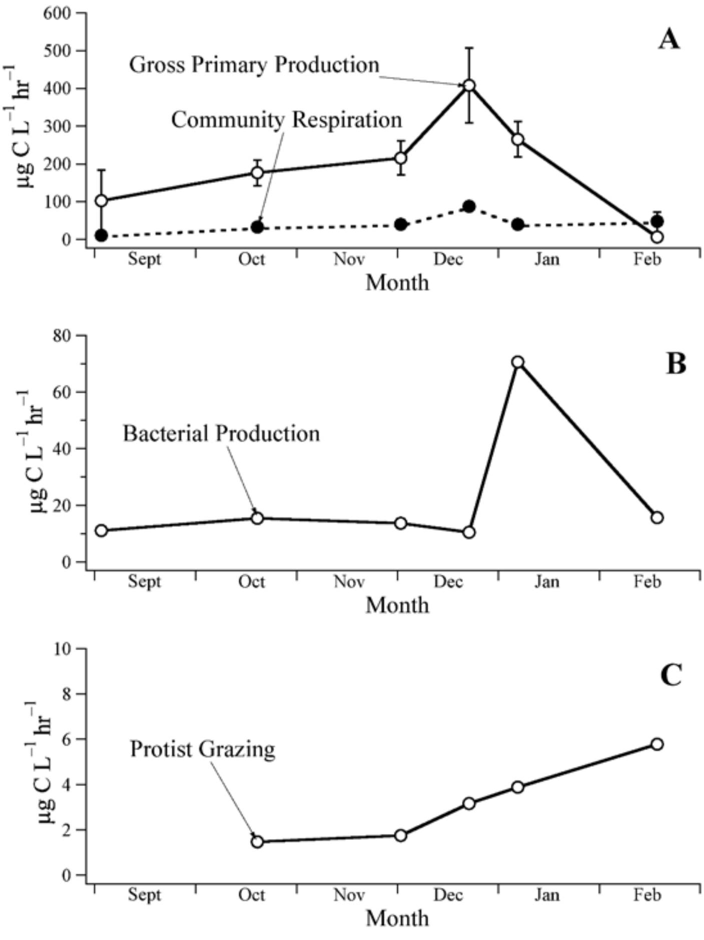

Figure 4. *Temporal variation of (A) Gross Primary Production and Community Respiration; (B) Bacterial Production; (C) Protist Grazing at the Klang River Estuary throughout the sampling period. Error bars (±Standard Deviation, S.D.) are shown for Gross Primary Production and Community Respiration, except where they are smaller than the symbols;*

Table 1 shows both the BR and concurrent BP (BP_{resp}) measured in this study and the BGE estimated for each sampling. In this study, the significant linear change in DO concentration for all the respiration experiments carried out showed there was no "undesirable effects of confinement" (Pomeroy et al., 1994). This could be due to the short incubation time (≤12 h). BR ranged 1.542 to 7.295 μM O_2 h^{-1} or 19.49–87.51 μg C L^{-1} h^{-1} whereas BP_{resp} ranged 3.90–6.08 μg C L^{-1} h^{-1}. The DO decrease in respiration experiments was due to biological activity because incubation of the <0.2 μm fraction showed no significant DO change (Lee and Bong, *submitted*). Using both BR and BP_{resp} values, BGE calculated ranged 6.4–22.9%. Comparison between the bacterial growth rate in the BR experiments and the dilution culture BP experiments showed no significant difference (Student's t-test: t=0.470, df=7, p>0.30). This indicated that the sample preparation carried out for the respiration experiments did not significantly affect bacterial activity.

***Table 1**. (a) Bacterial respiration (BR) rates ± standard error (μM O_2 h^{-1}±S.E.) at Klang River Estuary. (b) Concurrent bacterial growth rates (μ±S.E., h^{-1}) and bacterial production (BP_{resp}). df=degrees of freedom, p=significance level for the regression analysis, S.D.=standard deviation, BGE=bacterial growth efficiency.*

(a) Bacterial respiration (BR)	μM O_2 h^{-1}	±S.E.	df	p	BR±S.D. (μg C L^{-1} h^{-1})	
02 September 2004	–1.542	0.151	19	<0.001	18.49±8.09	
19 October 2004	–2.217	0.108	19	<0.001	26.59±5.82	
01 December 2004	–2.558	0.152	19	<0.001	30.69±8.16	
22 December 2004	–3.781	0.257	19	<0.001	45.35±13.80	
06 January 2005	–1.843	0.132	19	<0.001	22.10±7.11	
16 February 2005	–7.295	0.445	19	<0.001	87.51±23.88	
(b) Bacterial growth	**μ, h^{-1}**	**±S.E.**	**df**	**p**	**BP_{resp}±S.D. (μg C L^{-1} h^{-1})**	**BGE (%)**
02 September 2004	0.132	0.015	3	0.012	5.49±1.24	22.9
19 October 2004	0.167	0.030	3	0.031	3.90±1.40	12.8
01 December 2004	0.124	0.019	3	0.022	6.08±1.85	16.5
22 December 2004	0.134	0.019	3	0.020	5.55±1.60	10.9
06 January 2005	0.154	0.025	3	0.026	3.99±1.31	15.3
16 February 2005	0.187	0.018	3	0.009	5.92±1.11	6.4

5. DISCUSSION

The tropical climate in Peninsular Malaysia is strongly influenced by both North-East (NE) and South-West (SW) monsoons. Towards the end of the year, there is heavier rainfall that coincides with the NE monsoon; otherwise weather conditions are relatively stable with rainfall throughout the year (Uktolseya, 1988). Although our sampling period was only six months, it covered both wet and dry months. We found that salinity was highly variable compared with temperature. Changes in salinity were strongly influenced by freshwater run-off from large rivers and high rainfall (Uktolseya, 1988). The salinity decreased from early December 2004, and this coincided with the wetter months. The amount of freshwater that flowed from the Klang River diluted the salinity at the Klang River Estuary. This also decreased the pH as freshwater generally has a lower pH than seawater.

The water quality in Port Klang was poor due to the low DO concentration (<200 μM), and high TSS (>260 mg l^{-1}). High TSS levels is a pervasive problem in Malaysia (Dow, 1995). TSS can comprise of both biogenic and non-biogenic particulates. The high TSS here was mainly inorganic (POM <5% of TSS). This could be attributed to land clearing activities for construction projects, mining, agricultural and forest industries, and dredging operations (Dow, 1995). Moreover, the Klang River basin covers the Klang valley that represents the most rapidly developing part of Malaysia.

A good indicator of aquatic health is DO concentration as all respiring organisms require oxygen. Further, low DO concentration (or oxygen deficiency) causes stress response in fish and other aquatic organisms. This stressful level of oxygen-depletion is known as hypoxia. Although there is no universally accepted DO levels to describe hypoxia, the consensus from laboratory or field observations is 125 μM (Rabalais et al., 2002). Another convenient threshold for detecting physiological stress is the <50% DO saturation (Breitburg, 2002). Our study showed two episodes of hypoxia occurring, in early December 2004 and February 2005. In the latter date, DO was close to anoxia (no oxygen). If the Breitburg (2002) threshold is used, only in February 2005 did the oxygen deficiency cause stress (18% of DO saturation). At other periods, the DO levels were 58–97% of DO saturation.

Based on the marine water quality data collected by Department of Environment (DOE) of Malaysia, water quality at and around Port Klang (Pulau Babi Strait, North Klang Strait, Klang River Estuary and Langat River Estuary stations, see Figure 1) deteriorated from 1990 to 2003. Although there were gaps of two to five years in this compilation, TSS increased significantly (F=134.4, df=140, p<0.001) (Figure 5A) whereas DO decreased significantly (F=11.8, df=122, p<0.001) (Figure 5B). From the linear regression equations, we estimated that over the decade (1994–2003), TSS increased 132 mg L^{-1} whereas DO decreased 48 μM. When we included our data in the analysis, TSS and DO continued the existing trend.

Inorganic nutrients were within the range previously reported for estuarine waters in Malaysia (Nixon et al., 1984). NH_4 was the most dominant nitrogen species (mean=67%). This reflects a reducing environment where NH_4 accumulates, and is typical of mangrove waters (Alongi et al., 2003) or waters with low DO. Compared with Redfield's NO_3:PO_4 ratio of 16 (Redfield et al., 1963), the NO_3:PO_4 ratio observed here was extremely low, ranging 0.05–0.38. This suggested a nitrogen limiting condition for phytoplankton which concurred with Law et al. (2001) where they reported that the nutrient limiting factor in the Straits of Malacca is probably nitrogen. SiO_4 was persistently high in this study, and is typical of coastal stations with large river systems (for example the Klang River) as freshwater is a source of SiO_4 (Nixon et al., 1984).

Chl *a* concentration is frequently used as an indicator of phytoplankton biomass (Falkowski et al., 1998) as all primary producers have this photosynthetic pigment.

Chl *a* measured in this study was within the range previously reported for the Klang River (Lee et al., 1984). We observed a small phytoplankton bloom late December 2004 where the Chl *a* concentration doubled. Bacterial abundance measured here was within the range reported for the Indus River delta, Pakistan (Bano et al., 1997), and peaked in January 2005 whereas PPico abundance was within the range for most aquatic systems (Sherr and Sherr, 2000). Although bacteria are closely coupled to primary producers (Cole et al., 1988), we observed a lag in the bacterial increase

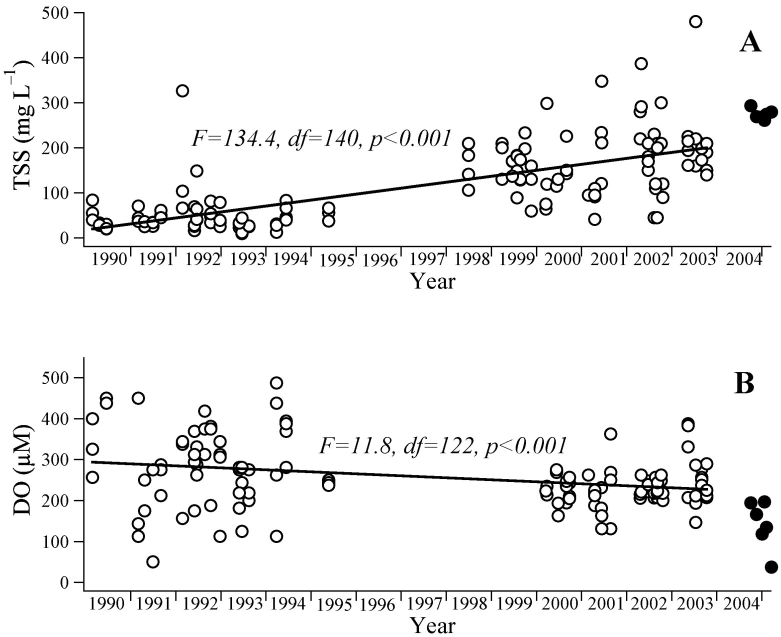

Figure 5. *Temporal variation of (A) Total Suspended Solids; (B) Dissolved Oxygen for stations Pulau Babi Strait, North Klang Strait, Klang River Estuary and Langat River Estuary. Data from 1990 until 2003 are from Department of Environment (DOE), Malaysia (open circles) whereas data from this present study are represented by closed circles. Linear regression line for the DOE data are also shown.*

when compared with phytoplankton bloom late December 2004. This could be due to the fact that bacterial biomass increases slower than phytoplankton biomass (Sanders et al., 1992; Simon et al., 1992).

Protist abundance was three orders lower than bacteria, and this conformed to the observation by Sanders et al. (1992) that in specific systems, there are usually 1000 bacteria per one protist. In this study, the carbon conversion factor of 32.8 fg C cell^{-1} was used, and this is comparable to other direct measurements of coastal samples (Fukuda et al., 1998). However it is about twice the factor used for open ocean systems (Caron et al., 1995). The wide range of conversion factor shows the necessity of measuring the carbon content per bacterium for different ecosystems.

Our GPP and CR results showed the eutrophic nature of Port Klang waters where both were one order higher than that reported for a mangrove estuarine system at Matang, Malaysia (Alongi et al., 2003). The phytoplankton bloom observed late December 2004 coincided with a two times increase in GPP. There was also a 70% increase in DO that could be attributed to primary production as photosynthesis releases oxygen (Grasshoff et al., 1999). However there was hypoxia in February 2005 due to reduced photosynthesis as GPP was lowest, and was 40 times less. This drastic reduction in GPP could be due to nutrient limitation. We found that GPP correlated significantly with NO_3 (R^2=0.867, n=5, p<0.05). The low NO_3 concentration in February 2005 could have limited GPP, and indirectly triggered hypoxia. This was further evidence of nitrogen limitation for primary producers. GPP is the basis of aquatic food web from where all autochthonous organic matter originates (Valiela, 1995). Excluding the data in February 2005, we found a highly significant correlation between GPP and CR (R^2=0.956, n=5, p<0.01). However in February 2005, there was uncoupling between primary production and heterotrophy as CR was probably supported by allochthonous organic matter from the Klang River.

Bacterial growth rates were within the range reported for an estuarine mangrove ecosystem at Matang, Malaysia (Alongi et al., 2003) but BP was up to six times higher. This was because the bacterial abundance here was generally higher, and the bacterial carbon conversion factor adopted by Alongi et al. (2003) was lower (25 fg C cell^{-1}). Although GPP was limited by nutrients, adding nutrients to our BP experiments did not stimulate bacterial growth rates indicating that there was no nutrient limitation for bacteria. This showed the catabolic flexibility of the bacterial community in obtaining its essential nutrients. Bacterial growth rates were not significantly stimulated even with the addition of glucose.

In marine environment, protists are important grazers or consumers of bacteria (Valiela, 1995). In this study, the consumption rate ranged 5.5–26.9×10^4 cells mL^{-1} h^{-1}, higher than that reported for coral reef waters (Ferrier-Pagés and Gattuso, 1998). This consumption rate was only 19±14% of BP, lower than the 60–70% reported by Ferrier-Pagés and Gattuso (1998). Ingestion rates ranged 18–72 cells protist^{-1} h^{-1}, higher than 9–36 cells protist^{-1} h^{-1} reported in North Atlantic waters (Weisse and Scheffel-Möser, 1991). By observing the temporal change in GPP, BP and grazing, we observed that there was a lag response to earlier events. Maximum GPP was in late December 2004, followed by maximum BP in January 2005 and then by maximum grazing in February 2005.

BR was higher than the range 5–25.44 μg C L^{-1} h^{-1} reported for tropical coastal waters near Goa, India (Pradeep Ram et al., 2003). BR:CR was about 0.29±0.16, and this was on the lower end of the range reported by Biddanda and Cotner (2002). Earlier studies have shown that BR:CR tends to decline as nutrient concentration increases (Schwaerter et al., 1988; del Giorgio et al., 1997). We determined the BGE at each sampling as BGE can have a wide range (del Giorgio and Cole, 2000). BGE was within the range reported for tropical estuarine and coastal waters (Pradeep Ram et al., 2003), and was generally >10%, but decreased to 6% in February 2005. The variability of BGE reflects both substrate quality (del Giorgio and Cole, 2000) and water temperature (Rivkin and Legendre 2001). As temperature was relatively stable, the sudden reduction of BGE in February 2005 was probably caused by the consumption of a more refractory organic matter. This further pointed to heterotrophy uncoupling from primary production.

With these BGE values, we calculated the carbon consumed by bacteria or the bacterial carbon demand (BCD). Table 2 shows the BCD, and the amount of carbon flux through the bacterial component. BCD ranged 1.16–11.08 g C m^{-3} d^{-1}. To determine whether there was net heterotrophy (BCD>GPP) and net autotrophy (BCD<GPP) at each sampling, the ratio BCD:GPP was estimated. It ranged over more than two orders. The lowest ratio (0.47) occurred when there was a phytoplankton bloom in late December 2004, and primary production alone could have supported BCD. The highest ratio (77.38) was in February 2005 when primary production was limited by NO_3, and could not support BCD. Uncoupling between bacteria and phytoplankton occurred. BCD:GPP ratio was also high (3.48) in January 2005, a post-bloom period. There was elevated BCD (highest at 11.08 g C m^{-3} d^{-1}) that was probably supported by excess primary production that occurred during the phytoplankton bloom earlier (Biddanda and Cotner, 2002). On other occasions, the BCD and GPP seemed in balance, and BCD:GPP ratio averaged 1.03. Of the C consumed by bacteria, only 2% was consumed by protists. Although microbial loop (organic matter–bacteria–protists) is essential to recycling of organic matter (Azam et al., 1983), we observed that in this eutrophic and tropical coastal system, it was not an efficient pathway as a substantial amount of carbon was lost.

***Table 2**. Amount of carbon flux through the bacterial component. GPP=Gross Primary Production ((g C m^{-3} d^{-1}), BP=Bacterial Production (g C m^{-3} d^{-1}), BGE=Bacterial Growth Efficiency (%), BCD=Bacterial Carbon Demand (g C m^{-3} d^{-1}), n.d.=not detectable.*

	GPP	BP	BGE	BCD	Grazed	Amount of bacteria consumed (%)	Amount of BCD transferred to higher trophic level (%)
02 Sep. 2004	1.23	0.27	22.9	1.16	n.d.	n.d.	n.d.
19 Oct. 2004	2.12	0.37	12.8	2.90	0.04	9.5	1.2
01 Dec. 2004	2.59	0.33	16.5	1.99	0.04	12.8	2.1
22 Dec. 2004	4.89	0.25	10.9	2.30	0.08	30.3	3.3
06 Jan. 2005	3.18	1.70	15.3	11.08	0.09	5.5	0.8
16 Feb. 2005	0.08	0.38	6.3	5.92	0.14	37.0	2.3

6. CONCLUSION

Port Klang waters is eutrophic and long term data showed that its water quality is deteriorating. We observed the occurrence of hypoxia due to very low GPP that was limited by NO_3. Although primary production is the basis of aquatic food web, and supported both CR and BCD, episodes of uncoupling were observed. Uncoupling occurred especially when GPP was very low, and generally BCD and GPP were balanced. Our results showed that only 2% of C consumed by bacteria were passed onto protists. This suggested that the microbial loop was not an efficient pathway to recycle organic matter as a substantial amount of carbon was lost.

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CHAPTER 21

PHYTOPLANKTON STRUCTURE IN THE TROPICAL PORT WATERS OF SINGAPORE

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1. INTRODUCTION

Singapore is an island located at the confluence of the Malacca Straits, the South China Sea and the Java Sea (Figure 1). Its unique geographical position and deep waters have enabled it to grow as a thriving port city and it is currently one of the busiest ports in the world. The challenge which Singapore and other coastal cities face is how to effectively manage the coastal impacts in such a way that sustainable development can be pursued.

One of the concerns is the occurrence of cultural eutrophication and harmful algal blooms. Cultural eutrophication is caused by excessive nutrient inputs which lead to the proliferation of phytoplankton. Several problems may arise, including the depletion of dissolved oxygen as the plant biomass decays and/or the production of toxins from harmful algal blooms (HABs). The incidence of eutrophication of coastal waters in South East Asia has increased dramatically in recent years, coinciding with increases in loading from domestic and industrial effluents. For example, occurrences of harmful algal blooms have been reported in Hong Kong (Lam and Ho, 1989, Ho and Hodgkiss, 1995), Philippines (Estudillo et al., 1984, Bajarias and Relox, 1996), Brunei (Jaafar et al., (1989), Papua New Guinea (Maclean, 1989), Sabah in East Malaysia (Ting and Wong 1989) and possibly the Malacca Straits of West Malaysia (Usup et al., 2002) and Indonesia (Azanza and Taylor, 2001). The need to establish baseline characteristics and to understand the potential for eutrophication is particularly important for Singapore as it continues to expand its coastal developments.

The control of nutrient inputs to the coastal environment is a key factor in determining the extent of eutrophication. For Singapore, the main anthropogenic

E. Wolanski (ed.), The Environment in Asia Pacific Harbours, 347–375.

***Figure 1**. Map of Singapore showing the locations of sampling sites in the Singapore Strait and Johor Strait.*

inputs into the coastal zone include wastewater effluents, storm runoff and vessel discharges. 100% of Singapore's population is served by modern sanitation; all wastewater (both domestic and industrial) is required to be discharged into the public sewerage system where it undergoes secondary treatment. Industrial wastewater, however, can be discharged into a watercourse (if a public sewer is not available), but it must be treated to a specified standard before discharge. Singapore has six water reclamation plants which not only treat wastewater but also reclaim water for non-potable use. However, to meet Singapore's long-term needs for the 21st century, the construction of a Deep Tunnel Sewerage System has been initiated. This consists of replacing the existing sewerage systems with two centralized, state-of-the-art, water reclamation plants to be located at either end of the island. The treated effluents from these plants will then be pumped through 5 km long outfall pipes 30 m below the sea surface, into the Straits of Singapore where they will be diluted and dispersed by currents.

Another major source of nutrients into the coastal zone is storm runoff. Singapore has a land area of 650 km^2 and sustains a population of about 4 million people. Most of Singapore's catchments are highly urbanized and channelized. However, to date there are few studies quantifying the amount of nutrient loading through the catchments. Major changes in the future are also anticipated given that existing estuaries will be converted to freshwater reservoirs to increase Singapore's water supply. This will have an impact on the nutrient loads from river and estuaries.

In this chapter, we present an overview of the phytoplankton composition in Singapore coastal waters and their relationships with nutrient and environmental conditions. A variety of techniques were employed to determine the structure of the phytoplankton community, in addition to overall biomass levels. These include sophisticated methods, such as flow cytometry and high performance liquid chromatography (HPLC) but also traditional methods, such as microscopy and extracted chlorophyll measurements. While field measurements are important for understanding baseline conditions and explaining past trends, they are not as useful for prediction. Thus, a numerical model was also developed to serve this purpose and to assist coastal managers in the assessment of eutrophication issues.

2. GEOGRAPHICAL LOCATION AND CLIMATOLOGY

The island of Singapore is located between latitudes 1°09'N and 1°29'N and longitudes, 103°38'E and 104°06'E and is bounded by the Johor Strait in the north and the Singapore Strait in the south (Figure 1). The Singapore Strait is located slightly 1°N of the equator and is about 16 km wide, separating Singapore and Indonesia. The Johor Strait to the north of Singapore Island is an even narrower channel, typically 1.8 km wide. The Johor causeway connects Malaysia and Singapore and divides the strait into the eastern and western parts. The causeway limits the flow of water and essentially divides the strait into two separate semi-enclosed water bodies, i.e., the West Johor Strait and East Johor Strait with the large Malaysian city of Johor Bahru situated along the northern shoreline.

The climate of Singapore is typically wet equatorial, with high temperatures and large amounts of rainfall throughout the year. Although the local climate is relatively uniform, it is modified by the Southeast Asian monsoon regime, which introduces variations in wind speed and direction, cloudiness, rain and dry seasons over the year. The North-East Monsoon period falls between December and February and is characterized by heavy rains and winds from the north-east. The South-West Monsoon, between the months of June and August, is the drier monsoon with winds driven from the south and south-west. The two Inter-Monsoon periods (March-May; September-November) are periods of relative calm with intermittent rains and weak and variable winds. The oceanic circulation patterns driven by the monsoon winds play an important role in determining the distribution of plankton and other environmental parameters in the Singapore and Johor Straits.

3. PHYTOPLANKTON STRUCTURE

The first recorded survey of plankton in Singapore waters was undertaken in the 1950s (Tham, 1953); a dominance of diatoms in the Singapore Strait was observed, with higher concentrations found in the Inter-Monsoon period, between April and May (2,500 cells L^{-1}). Similar concentrations were also found in the Johor Strait and this was attributed to inflows from the Singapore Strait to the Johor Strait. A later study by Chou and Chia (1991) also confirmed the dominance of diatoms but since then, there has been no published data on phytoplankton until recently. In the last six years, detailed studies of the dynamics and composition of the phytoplankton community were undertaken using a variety of methods, ranging from microscopy, size-fractionated chlorophyll, high performance liquid chromatography (HPLC) to flow cytometry (Gin et al., 2000; Gin et al., 2003). These different approaches help to bring out the different features of the phytoplankton community.

Microscopy is the earliest and most common method for phytoplankton identification and enumeration. While detailed size and species information can be acquired, the method is very labor-intensive, often taking several hours per sample. In addition, skilled personnel in taxonomy are needed to analyze natural samples with assorted phytoplankton; and many delicate cells may be damaged or overlooked and consequently be underestimated. An alternative to the use of conventional light microscopes is the spectrophotometric determination of total chlorophyll *a* to estimate algal biomass levels (Yentsch and Yentsch, 1989). Although it is simpler and less tedious, the measurement does not give information on community structure; neither does it give the responses of different taxa within the phytoplankton assemblage to external environmental perturbations. To avoid this limitation, size fractionation of chlorophyll can be undertaken, where the phytoplankton are categorized into different size classes. Phytoplankton size is important because it regulates phytoplankton growth and loss rates (Harris, 1986), thereby significantly affecting phytoplankton abundance (Agusti et al., 1990) and its contribution to community biomass. The variability and distribution of the size-fractionated phytoplankton biomass and productivity have important implications in the path of carbon produced in the euphotic zone, and in the pelagic food chain structure (Ryther, 1969; Walsh, 1976).

In the last twenty years, flow cytometry has made a significant impact in the analysis of environmental microbial populations. With flow cytometry, individual particles can be examined at very high speeds (millions of cells in several minutes) and discriminated on the basis of their light scattering and fluorescence properties. Phytoplankton, in particular, is ideal for flow cytometric analysis because they are naturally autofluorescent by virtue of their photosynthetic pigments. With blue light (488 nm) excitation, chlorophyll *a* will fluoresce red (>650 nm) while accessory pigments, such as phycoerythrin (PE), fluoresce orange (575 nm). Based on autofluorescent signals and light scattering properties, biological oceanographers have been able to typically divide the phytoplankton in to *Synechococcus*, *Prochlorococcus*, pico-eukaryotic phytoplankton and large eukaryotic phytoplankton (Olson et al., 1990; Campbell et al., 1994; Partensky et al., 1996; Campbell et al., 1998).

Flow cytometry is a particularly useful tool for phytoplankton analysis in the open ocean, where the picoplankton fraction is dominant. However, in coastal waters where larger eukaryotic phytoplankton, such as diatoms and dinoflagellates, may dominate, their relatively low concentrations, presence of chain-forming species and homogenous spectral properties sometimes make it difficult to discriminate the sub-populations. High performance liquid chromatography (HPLC) provides an additional framework to profile whole phytoplankton communities in the marine ecosystem (Wright et al., 1991). Measuring the relative and absolute concentration of phytoplankton pigments is also important for understanding the relationships between phytoplankton species and their environment. Studies have shown that areas with different hydrographic conditions and trophic state can be characterized by selected groups of phytoplankton with different pigment type and concentration (Matta and Marshall, 1984; Gould and Fryxell, 1988). One advantage of HPLC-derived pigment analysis is that the proportion of major phytoplankton groups (classes) contributing to the total chlorophyll *a* can be estimated, via the ratio of the accessory pigment to chlorophyll *a* (Everitt et al., 1990; Ondrusek et al., 1991). However, these ratios can vary to some extent with depth and also regionally, although the majority of ratios appear to be constrained within a narrow range of values.

3.1. Patterns of Total Chlorophyll

In most temperate environments, phytoplankton communities respond dramatically to seasons, with phytoplankton biomass and productivity often changing by several orders of magnitude over the seasons (Cebrian and Valiela, 1999). The pronounced spring outbursts (diatom blooms) in these waters are driven by stratification of the water column in deep waters and increasing light intensity in the surface layer (Harris, 1986). Dramatic seasonal changes are thought to be absent in subtropical and tropical areas (Blackburn, 1981). In particular, for tropical waters where there is a more or less permanent thermocline, the concentration of chlorophyll in surface waters is often found to be continuously low (Harris, 1986). Exceptions do exist, however, especially for those areas subject to strong monsoonal forcing. For example, the tropical Arabian Sea is characterized by regular, seasonal oscillations in phytoplankton biomass: during the South-West Monsoon, the wind-induced upwelling processes bring nutrient-enrich subsurface water into the surface euphotic zone, thereby stimulating new production (Bauer et al., 1991). In addition, the influence of seasonal monsoons on size-fractionated chlorophyll has been reported for the coastal upwelling zone in the north-west Indian Ocean (Savidge and Gilpin, 1999). During the late South-West Monsoon, maximal chlorophyll and production values were recorded and the phytoplankton community were dominated by the >18 μm size fraction (diatoms). However, during the inter-monsoonal season, both chlorophyll concentrations and production decreased and were dominated by picoplankton (0.2-2 μm size fraction).

Relatively lower chlorophyll concentrations (average 1.4 $\mu g\ l^{-1}$) were recorded in the Singapore Strait during the North-East Monsoon of November 1997 to February, 1998 (Gin et al., 2000; Lin, 2000). Higher chlorophyll levels coincided with the South-West Monsoon in July (average 2.5 $\mu g\ l^{-1}$) and August (average 3.3 $\mu g\ l^{-1}$), particularly along the east coast of Singapore. A significant increase ($p<0.05$) in chlorophyll concentrations by about 5-10 times was observed in July and August, 1998 (5-10 $\mu g\ l^{-1}$) (South-West Monsoon) compared the North-East Monsoon (0.8 to 3 $\mu g\ l^{-1}$). Using pooled field measurements from September, 1996, to December, 1998, chlorophyll levels were found to be statistically higher ($p<0.05$) during the South-West Monsoon (average 2.6 $\mu g\ l^{-1}$) and lower during the North-East Monsoon (average 1.4 $\mu g\ l^{-1}$). A similar seasonal trend was also reported by Brock et al. (1993) in the tropical western Arabian sea; chlorophyll concentrations were generally low throughout the year (<0.3 $\mu g\ l^{-1}$), however, the chlorophyll levels increased to about 4 $\mu g\ l^{-1}$ during the South-West Monsoon.

In the shallow waters of the East Johor Strait, frequent algal blooms with very high chlorophyll levels (>40 μg l^{-1}) were observed. These blooms seemed independent of the seasons and were likely due to variable anthropogenic inputs, coupled with suitable tidal conditions (e.g. neap low tides generally resulted in higher levels of chlorophyll).

3.1.1. Relationships between Phytoplankton, Nutrients and Water Quality

Levels of nitrogen and phosphorus in the Singapore Strait are generally low compared to the more eutrophic Johor Strait. Based on measurements taken from November, 1997, to December, 1998, phosphate concentrations were generally less than 0.02 mg l^{-1} and comprised about 40% to 95% of total phosphorus (TP) concentrations (Table 1). Nitrite+nitrate levels ranged from 0.02 to 0.07 mg l^{-1}, while ammonium ranged from 0.01 to 0.03 mg l^{-1}. Earlier studies in the 1970s showed that the nutrient concentrations in the Singapore Strait were of a similar range, with phosphate ranging from 0.003 to 0.033 mg l^{-1} and nitrate ranging from 0.048 to 0.082 mg l^{-1} (Tham, 1973). Nitrogen in inorganic forms was less abundant than organic forms, with dissolved inorganic nitrogen (DIN) (average 0.058 mg l^{-1}) typically comprising only about 7% of total nitrogen (average 0.78 mg l^{-1}). This ratio increased in the East Johor Strait, with DIN (average 0.49 mg l^{-1}) occupying about 30% of total nitrogen (average 1.68 mg l^{-1}). In comparison, studies in similar tropical conditions in Thailand showed a much lower fraction of DIN (less than 1% of total nitrogen) (Wattayakorn et al., 1994).

Table 1. *Average, range and number of samples (n) for nutrients, chlorophyll, temperature and salinity measured in the Singapore Strait and East Johor Strait. Data was collected from November, 1997, to December, 1998 (n.d. means non-detectable).*

		Nitrite + Nitrate (μg l^{-1})	*Ammonia (mg l^{-1})*	*Phosphate (mg l^{-1})*	*Total Nitrogen (mg l^{-1})*	*Total Phosphorus (μg l^{-1})*	*Chlorophyll a (μg l^{-1})*	*Temp. (°C)*	*Salinity*
a	*1*	36	0.015	0.009	0.55	16	1.7	29.7	30.6
	2	**5-78**	**n.d.-0.053**	**n.d.-0.024**	***0.21-1.1***	**5-31**	**0.4-10.5**	**28.3-31.2**	**28.7-32.2**
	3	140	92	140	98	140	140	205	196
b	1	0.146	0.098	0.04	1.6	75	21.5	29.8	28.0
	2	**0.013-0.4**	**n.d.-0.178**	**0.007-0.075**	**1.3-2.1**	**20-167**	**1-60**	**27.6-32.2**	**19-33**
	3	22	22	22	22	22	22	102	102

The nutrient levels in the East Johor Strait are much higher and approximately 5-10 times that of the Singapore Strait ($p<0.05$), with no obvious temporal trends ($p>0.05$). Comparable results have also been reported by Koh (1998) and can be attributed to variable anthropogenic inputs from the rivers and outfalls that flow into it from both Singapore and Malaysia. Consequently, wide and irregular fluctuations in phytoplankton abundance are generally observed in the East Johor Strait, with no seasonal pattern evident. The frequent algal blooms result in elevated chlorophyll concentrations, sometimes reaching as high as 60 μg l^{-1}. In comparison, the chlorophyll concentrations are generally low (0.5 to 4 μg l^{-1}) in the Singapore Strait, due in part to the relatively well-mixed waters and good flushing characteristics.

Using data collected from November, 1997 to December, 1998, correlations between the different physical, chemical and biological parameters were obtained. For correlations between temperature and salinity, the overall relationship using pooled data was not significant. However, strong correlations were observed during the South-West and North-East Monsoons in the Singapore Strait (Table 2). This

Table 2. *Correlations (r[n]) between temperature, salinity, nutrients and chlorophyll in the Singapore and East Johor Strait (r- correlation coefficient; n- number of data; ALL- all data from November, 1997, to December, 1998; SW-data from the South-West Monsoon; NE-data from the North-East Monsoon; ns-no significant correlations,*significant at 0.01<P<0.05,**highly significant at P<0.01).*

		T	S	Nitrate+ Nitrite	Ammonia	Phosphate	Total Nitrogen	Total Phosphorus
a	T							
	All		0.1 ns [100]	-0.046 ns [60]	-0.393 * [38]	0.457 ** [53]	0.148 ns [39] -0.594 ** [22]	0.394 ** [60]
	SW		0.527** [35]	-0.524 * [21]	-0.370 ns [21]	0.199 ns [21]	-0.321 ns [17]	-0.516 * [21]
	NE		0.541** [65]	-0.323 * [39]	-0.267 ns [17]	0.263 ns [32]		0.242 ns [39]
	S							
	All			-0.574 ** [60]	0.107 ns [38]	-0.313 * [53]	-0.678 ** [39] -0.467 * [22]	-0.391 ** [60]
	SW			-0.458 * [21]	-0.258 ns [21]	0.007 ns [21]	0.136 ns [17]	-0.261 ns [21]
	NE			-0.447 ** [3]	-0.014 ns [17]	0.062 ns [32]		0.121 ns [39]
	C							
	All	0.158 ns [59]	-0.246 ns [59]	-0.080 ns [60]	0.232 ns [41]	0.221 ns [57]	0.343 * [41] 0.160 ns [22]	0.519 ** [64]
	SW	-0.12 ns [20]	-0.056 ns [20]	-0.479 * [21]	0.321 ns [22]	0.054 ns [22]	0.615 ns [19]	0.579 ** [22]
	NE	-0.032 ns [39]	-0.056 ns [39]	0.102 ns [39]	0.788 ** [19]	0.144 ns [35]		0.326 * [42]
b	C			-0.643 ns [8]	-0.236 ns [8]	-0.529 ns [8]	-0.280 ns [8]	-0.074 ns [8]
a & b				0.524 ** [68]	0.566 ** [49]	0.486 ** [65]	0.517 ** [49]	0.646 ** [72]

indicates that the water masses passing through the Strait during the two monsoon seasons may have different physical characteristics. In general, temperature was positively correlated with phosphate and TP, but negatively correlated with nitrogen in the Singapore Strait. In comparison, salinity was negatively correlated with nutrients. The exception was ammonium which did not show any significant correlation with salinity. These relationships between nutrient, temperature and salinity suggest that the major trends in nutrient concentrations may be partially caused by circulation from the surrounding seas.

From this study, neither temperature nor salinity seems to play a major role in determining the chlorophyll abundance ($r=0.158$ and -0.246 respectively). Although temperature is known to influence phytoplankton growth rates directly through increase in uptake rates and half-saturation constants (Goldman and Carpenter, 1974), the effects of temperature on cell size are less predictable. The small range in temperature recorded in this tropical ecosystem (3° C) compared to that from temperate coastal waters is one reason why we do not observe much temperature dependence for chlorophyll concentrations. Similarly, the range in salinity measured in the Singapore Strait is also small and hence, no significant relationship with total chlorophyll was observed ($r=-0.246$, $n=59$). However, for estuarine environments, it has been reported that phytoplankton (especially picoplankton) concentrations showed positive correlations with salinity (Iriate, 1993).

Nutrient levels were generally positively but poorly correlated with chlorophyll concentrations in the Singapore Strait (except for ammonium during the North-East Monsoon ($r=0.788$, $n=19$). However, for nitrate+nitrite, higher chlorophyll readings were recorded at stations with lower nitrate+nitrite concentrations ($r=-0.479$, $n=21$).

3.1.2. Limiting Nutrients in the Singapore Strait and East Johor Strait

To identify the limiting nutrient in a particular ecosystem, the empirical relationship between the major plankton constituents is needed. The Redfield ratio for dissolved nitrogen and phosphorus (N/P) is generally taken to be 16:1 (by atoms) (Redfield et al., 1963); although recent studies have indicated that the Redfield ratio is an average of species-specific N:P ratios and that the optimal N:P ratio can vary from 8.2 to 45 depending upon ecological conditions (Klausmeier et al., 2004). In spite of this difference, the N:P ratio can be used as a first approximation to estimate the nutrient that could potentially limit growth in the system. Based on the field measurements, the DIN:DIP ratio in the Singapore Strait is relatively low, ranging from 6 to 18 (by atoms) and averaging 11, implying possible limitation by nitrogen.

We also conducted simple nutrient enrichment tests using water from both the Singapore Strait and East Johor Strait, to determine whether nitrogen or phosphorus was limiting production. Concentrations of nutrients were prepared according to f/2 media (Guillard, 1975) and seawater samples were incubated in 1 L glass bottles at ambient light and temperature in the laboratory for one week (Lin, 2000). Chlorophyll in each bottle was measured at the start and end of the experiment and compared with un-enriched controls. The results showed that the samples enriched with nitrate gave higher chlorophyll concentrations, whereas those enriched with phosphate differed little from the controls (Table 3). In addition, the bottles enriched with all nutrients except phosphate showed higher chlorophyll yield, suggesting that nitrogen was probably the limiting nutrient. However, according to Koh (1998), similar nutrient enrichment tests for East Johor Strait waters showed that phosphorus could also be limiting. Our field data showed that the DIN:DIP ratio in the East Johor Strait ranged from 16:1 to 32:1, presumably due to the variable inputs of river, sewage effluents and stormwater runoff into the East Johor Strait. This variability suggests that either nitrogen or phosphorus could limit primary production, depending on the nature of the anthropogenic sources at that point in time.

Table 3. *Results of nutrient enrichment test conducted on waters from the Singapore Strait and East Johor Strait (N-nitrogen, P-phosphorus, All-all nutrients according to F/2 media (Guillard, 1975)).*

		Chlorophyll a (μg l^{-1})		
		Replicate 1	Replicate 2	Average
Singapore Strait	Control	1.5	0.9	1.2
	Only N added	6	8	7.0
	Only P added	1.5	2.4	2.0
	All except N	4	5	4.5
	All except P	80	97	88.5
	All	100	141	121
Johor Strait	Control	28	49	38.5
	Only N added	77	50	63.5
	Only P added	36	37	36.5
	All except N	17	26	21.5
	All except P	111	101	106
	All	191	233	212

3.2. Size-Fractionated Chlorophyll

Size-fractionated chlorophyll measurements showed that, in general, the picoplankton (<1 μm) contributed a small percentage of the total chlorophyll, typically between 1% to 20% (total chlorophyll ranging from 0.5 to 60 μg l^{-1}) (Gin et al., 2000, Lin, 2000). In comparison, it has been reported that this size fraction contributes more than 70% of the total chlorophyll for oligotrophic open oceans in the Sargasso Sea (Goerick, 1998) and Arabian Sea (Jochem and Zeitzschel, 1993) where total chlorophyll is about 0.5 μg L^{-1}. However, our data is comparable to data from Southampton coastal waters where total chlorophyll ranged from 1.2 to 12.7 μg L^{-1} (Iriate, 1993). These observations can be partially explained by the greater efficiency of small cells in absorbing nutrients from the environment due to their higher surface area to volume ratio (Smatechek, 1985), especially at very low nutrient concentrations where the diffusion rates of molecules towards the cell surface may limit the nutrient supply to the cell (Pasciak and Gavis, 1974; Hudson and Morel, 1991).

With respect to larger size classes, cells in the <8 μm fraction generally occupied about 40% of the chlorophyll stock, when total chlorophyll was less than 3 $\mu g\ l^{-1}$ in the Singapore Strait. This percentage decreased significantly ($p<0.05$) to around 20% during bloom conditions, e.g. in the South-West Monsoon when chlorophyll increased to 5-10 $\mu g\ l^{-1}$ along the east coast of Singapore. In the particular case in the East Johor Strait where total chlorophyll exceeded 45 $\mu g\ l^{-1}$, the proportion of the <8 μm size fraction decreased further to 6% ($p<0.05$). During these blooms when the phytoplankton assemblage was skewed to larger size classes, a significant fraction of total chlorophyll was contributed by the larger 8-20 μm or 20-100 μm size classes, or both. This is consistent with earlier studies, which showed that as chlorophyll biomass is added to the coastal ecosystem, it is usually contributed by the larger nano- or microplankton (Robles-Jarero and Lara Lara, 1993). The abundance of large phytoplankton in nutrient-enriched coastal areas can be partly explained by selective grazing by microzooplankton. Riegman et al. (1993) showed how during a coastal spring bloom, substantial grazing by microzooplankton on the smaller algae resulted in biomass accumulation of the larger algal species. Models have also been used to simulate the seasonal succession of the phytoplankton community (Moloney et al., 1991; Taylor et al., 1993). The results show that the first phytoplankton groups to grow at the beginning of a bloom are the small-sized phytoplankton, but increases in these groups are soon halted by the rapid growth of microzooplankton and heteroflagellates, whereupon they are succeeded by diatoms.

The phytoplankton size structures have implications on the trophic state and health of the water body. In lake ecosystems, the percentage of picoplankton in relation to total chlorophyll has been found to be related to trophic state, i.e., approximately 10% for oligotrophic lakes, 5% for meso-eutrophic lakes and 0.5% in hypereutrophic lakes (Rojo and Rodriguez, 1994). For marine ecosystems, measurements have also shown that in going from oligotrophic to eutrophic waters, the proportion of small cells decreases relative to larger cells. This was clearly shown in our measurements where higher nutrient levels in the East Johor Strait led to higher chlorophyll levels and hence, a relative decrease in abundance of the small size classes (<8, <5, <1 μm size fractions) (Figure 2). However, a poor relationship existed between the proportion of large cells (i.e., 8-20, 20-100 and 100-200 μm phytoplankton) and total chlorophyll, suggesting that a wider range of factors may influence the relative contribution of these large cells.

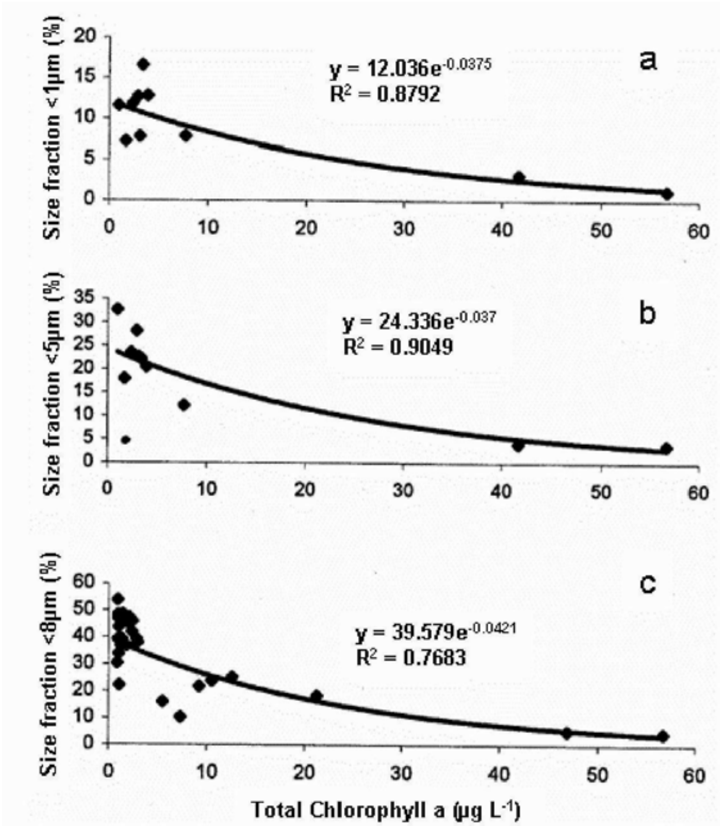

Figure 2. *Percentages of a) <1 µm, b) < 5 µm and c) < 8 µm size classes as a function of total chlorophyll a. Data were pooled from the Singapore and Johor Straits, from February to December, 1998.*

In terms of the absolute values of size-fractionated chlorophyll, studies in the Mediterranean Sea have shown that the upper limits reached for the <1, <3 and <10 µm size classes were 0.5, 1 and 2 µg l^{-1} respectively, when total chlorophyll was less than 5 µg l^{-1} (Raimbault et al., 1988). Based on measurements in Singapore waters, size-fractionated chlorophyll for the <1 µm size class ranged from 0.3 to 0.6 µg l^{-1} when total chlorophyll was less than 10 µg l^{-1}; and increased to 1.3 µg l^{-1} when total chlorophyll exceeded 40 µg/L (Figure 3). This is larger than the upper limit reported by Raimbault (1988). For total chlorophyll less than 10 µg l^{-1}, measurements for Singapore waters showed that the upper chlorophyll limit in the <8 µm size class was less than 2.5 µg l^{-1}, comparable to Raimbault el al.'s study. However, for total chlorophyll greater than 10 µg l^{-1} (i.e. beyond the range reported by Raimbault et al., 1988), maximum chlorophyll in the <8 µm size class increased to about 4 µg l^{-1} (total chlorophyll between 10 to 25 µg l^{-1}) and subsequently decreased to about 2.5 μg l^{-1} (total chlorophyll > 45 μg l^{-1}). Hence, while there does appear to be an upper limit to this particular size fraction at the lower and upper end of the chlorophyll scale, some variability exists in the intermediate range between 10-25 μg l^{-1}. One reason for an upper limit to the different size categories could lie in the metabolic constraints of size. As nutrient concentrations increase under constant light intensity, the growth rate of a particular size class of phytoplankton increases until eventually some maximum is reached, according to Michaelis Menten kinetics (Thingstad and Sakshaug, 1990). Maximum growth rates are known to be inversely related to size, following a power law function (Peters, 1983). Once growth rates of small cells are saturated, ambient nutrient concentrations can increase substantially to allow larger phytoplankton (with larger half-saturation constants) to grow.

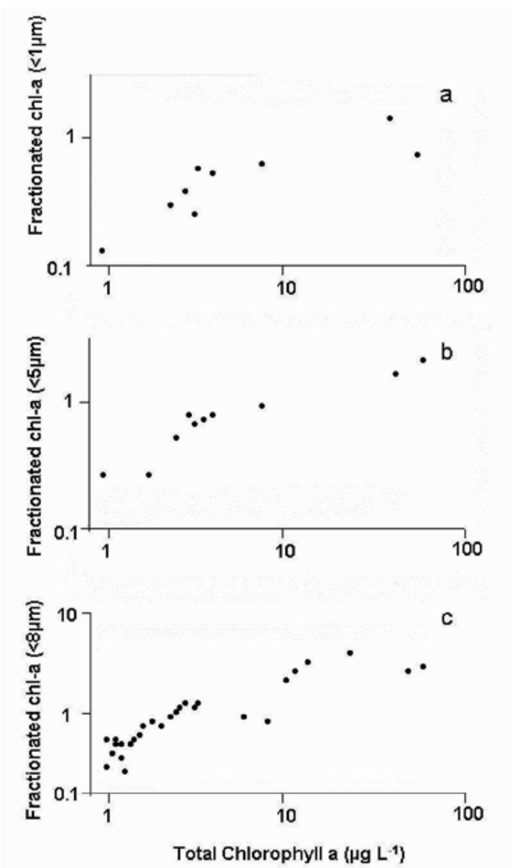

Figure 3. *Size-fractionated chlorophyll a (μg l^{-1}) of a) < 1 μm, b) < 5 μm and c) < 8 μm size classes as a function of total chlorophyll a. Data were pooled from the Singapore and Johor Straits, from February to December, 1998.*

3.3. Microscope and HPLC Pigment Analysis

Field samples for the microscope analysis were collected from August, 1998, to November, 1998 (Zhang, 2001). As expected from total chlorophyll measurements, total phytoplankton counts of net phytoplankton (>10 μm) using microscopy in the East Johor Strait (>600,000 cells l^{-1}) far exceeded that of the Singapore Strait (<100,000 cells l^{-1}) (Zhang, 2001). In both waters, diatoms clearly dominated, in particular, chain-forming species such as *Skeletonema sp*. (Figure 4) and *Chaetoceros sp.*; although, a range of solitary centric and pennate diatoms were also common (e.g. *Cyclotella* sp., *Eucampia* spp., *Nitzschia* spp. and *Rhizosolenia* spp.). Along with diatoms, a diverse range of armoured and unarmoured dinoflagellates, raphidophytes (*Chattonella* spp., *Fibrocapsa japonica*, *Heterosigma akashiwo*) and small flagellates were also abundant and dominant at various times, especially in the Johor Straits. Although *Skeletonema sp.* and *Chaetoceros sp.* were dominant species in the Johor Strait, their proportions in the total phytoplankton community (13.6% and 24.3% respectively) differed from the Singapore Strait (34.7% and 14.3% respectively). In addition, ciliates such as *Tintinnopsis sp.,* were also abundant in the Johor Strait, but were relatively rare in the Singapore Strait (Gin et al., 2000).

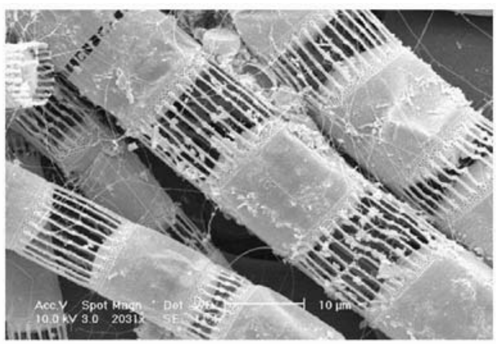

Figure 4. *Scanning electron micrograph of the chain-forming centric diatom Skeletonema sp. This is the dominant net-phytoplankton species in Singapore waters.*

HPLC analysis of seawater samples were conducted on bimonthly samples collected from the West Johor Strait and monthly samples from the Singapore Strait during the period July, 1999, to November, 1999. Details of the HPLC method can be found in Zhang (2001) and Gin et al. (2003). The results showed that all the major chemotaxonomic pigments including fucoxanthin, diadinoxanthin, zeaxanthin, chlorophyll *a*, chlorophyll *b*, β-carotene and pheopigments were readily identified in the field samples. Fucoxanthin and diadinoxanthin were prominent in the Singapore and Johor Strait, indicating diatoms as the major primary producer, consistent with the microscope results. However, the absolute concentrations of fucoxanthin and diatoms differed in the two straits. In the Singapore Strait, the average fucoxanthin concentration was 0.356 μg l^{-1} (range 0.115–0.779 μg l^{-1}), whereas in the Johor Strait, it was 2.294 μg l^{-1} (range 0.634–5.716 μg l^{-1}). Larger fluctuations in both fucoxanthin and chlorophyll *a* were also observed in the Johor Strait, corresponding to the variable anthropogenic nutrient loading into this region. Strong correlations between fucoxanthin and chlorophyll *a* were observed for both the West Johor Strait (r^2=0.964) and the Singapore Strait (r^2=0.902). Using accessory pigment:chlorophyll *a* ratios from the literature (Everitt et al., 1990; Ondrusek et al., 1991), diatoms consisted of approximately 72% of total chlorophyll *a* in the Singapore Strait whereas in the West Johor Strait, they comprised 88%.

Unlike the Johor Strait, the more oligotrophic Singapore Strait showed the significant presence of other accessory pigments (e.g. zeaxanthin, alloxanthin), suggesting a more diverse phytoplankton community. The average concentration of zeaxanthin measured in the Singapore Strait was 47.5 ng l^{-1} (range 18–102 ng l^{-1}). *Synechococcus* was the main source of zeaxanthin in this region (confirmed by flow cytometry), with only a minor contribution from green algae (<2.5%). Alloxanthin was also measured in the Singapore Strait, with an average concentration of 26.9 ng l^{-1} (range 12.3-55.5 ng l^{-1}). Alloxanthin is an unequivocal marker for only one group of algae, the cryptophytes. This phytoplankton appeared to be more abundant in the Singapore Strait compared to the West Johor Strait, where they were generally undetected by the HPLC method. In contrast, peridinin (an indicator of dinoflagellates) was detected more frequently in the Johor Strait compared to the Singapore Strait, although it only formed a minor fraction of the total chlorophyll *a*.

Traditionally, chlorophyll *b* is used as an indicator of green algae since these organisms are hard to distinguish because of the absence of morphological features. In the Singapore Strait, the contribution of green algae to total chlorophyll *a* was quite stable (8.6%), with an average concentration of 35.7 ng l^{-1} (range 23-66 ng l^{-1}). In contrast, chlorophyll *b* concentrations in the West Johor Strait fluctuated dramatically, from undetectable to 273 ng l^{-1}. Other chlorophyte markers such as lutein, pasinoxanthin and violaxanthin were not detected, suggesting that the type of green algae dominant in Singapore coastal waters could be prasinophytes which lacked significant amounts of prasinoxanthin, as shown by studies elsewhere (Everitt et al., 1990; Barlow et al., 1997). Trace amounts of a chlorophyll *b*-like pigment were also detected at two of the stations in the Singapore Strait, with a concentration of approximately 14 ng l^{-1} and contributing up to 5% of total chlorophyll *a*. We suspected that this chlorophyll *b*-like pigment was divinyl chlorophyll *b*, since the bathochromic shift was similar to that found in other studies (Goericke and Repeta, 1993). Divinyl chlorophyll *b* is a unique signature of the prokaryotic picoplankton, prochlorophytes. However, we were unable to confirm the presence of this organism by flow cytometry.

3.4. Flow Cytometric Analysis

In addition to the above, flow cytometry was used to analyze the phytoplankton community according to: 1) the pico (0.2-2 μm) and ultraphytoplankton (~2-10 μm) and 2) the net plankton (>10 μm fraction collected with a plankton net). The temporal and spatial samplings were conducted in a similar manner to the HPLC samples and details of the flow cytometric method can be found in Zhang (2001) and Gin et al. (2003). A typical result of the pico and ultraphytoplankton is shown in Figure 5, where populations can be discriminated from non-algal particles because the former fluoresce red. The average concentration of pico and ultraplankton measured in the Singapore Strait was 8.2×10^4 cells ml^{-1} (range 5.4×10^4-11.5×10^4 cells ml^{-1}) and more than 70% of the cells in this group were comprised of *Synechococcus*. *Synechococcus* was discriminated based on its ability to fluoresce orange (due to the presence of phycoerythrin) in addition to red. Another two populations with higher forward scatter (i.e. larger cell size) and both orange and red fluorescence were identified as cryptophytes, consistent with the measurement of alloxanthin by HPLC. The abundance of these two populations (about 4489 cells ml^{-1}) was significantly lower than that of *Synechococcus*.

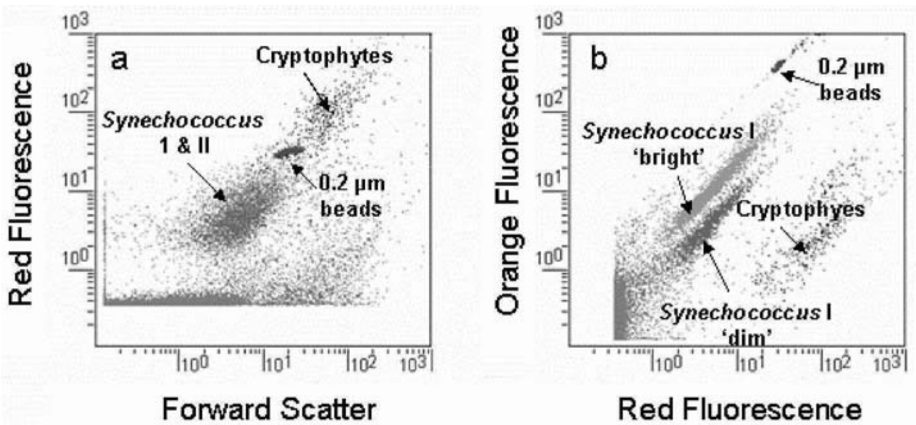

Figure 5*. Typical signatures of picophytoplankton and ultraphytoplankton from the Singapore Strait using flow cytometry: (A) Cytogram of Red Fluorescence (chlorophyll a) versus Forward Scatter (surrogate for size) and (B) Cytogram of Orange Fluorescence (phycoerythrin) versus Red Fluorescence. Two populations of Synechococcus ("bright" and "dim") and Cryptophytes could be discriminated based on their orange and red fluorescence (B). Standard beads (2µm) were used as reference and internal standards.*

In contrast, the average concentration of total pico and ultraphytoplankton in the Johor Strait was 2.7×10^4 cells ml^{-1}, only about one third that of the Singapore Strait. For net plankton (>10 µm), individual species could not be distinguished because the commonly encountered large eukaryotic phytoplankton (e.g. diatoms and dinoflagellates) absorb in similar regions of the spectrum and all emit red fluoresce from chlorophyll *a* (Figure 6). However, their total abundance could be determined.

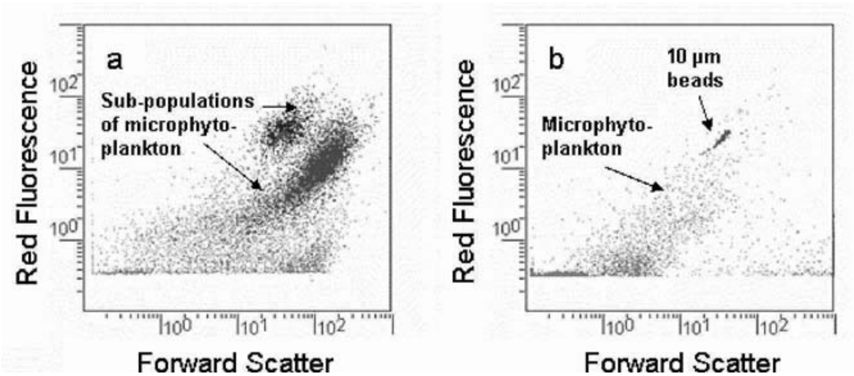

Figure 6. *Typical signatures of microphytoplankton and large nanophytoplankton (10 µm-100 µm) using flow cytometry from: (A) west coast and (B) east coast of the Singapore Strait. Standard beads (10µm) were used as reference and internal standards. The phytoplankton population in (A) is distinct from (B) in terms of its higher forward scatter mean value (i.e. cell size), higher red fluorescence (i.e. chlorophyll a content) and total concentration.*

In the Singapore Strait, the average concentration of net plankton was 2.1×10^4 cells ml^{-1}, whereas in the Johor Strait, the value was ten fold higher at 1.5×10^5 cells ml^{-1}. (Note that only cells ranging from 10 to 60 µm were enumerated using flow cytometry and hence, their abundance was less than the microscope counts). The mean forward light scatter for the phytoplankton population in the Johor Strait was also considerably higher than that in the Singapore Strait, reflecting the larger cell sizes in the more eutrophic waters of the Johor Strait.

3.4.1. Synechococcus

Using flow cytometry, the picophytoplankton, *Synechococcus* was found to be abundant in the tropical waters of Singapore, supporting the ubiquity of this algal species in many areas of the world's oceans. *Synechococcus* populations (5.5-15×10^3 cells ml^{-1}) have been measured in oceanic waters off Japan (Kudoh et al., 1990), the Arabian Sea (~5.3×10^5 cells ml^{-1} during the early North-East Monsoon) (Campbell et al., 1998), the North Pacific Ocean (Iturriaga and Mitchell, 1986), North Atlantic Ocean (Olson et al., 1990) and Kaneohe Bay, Hawaii (Landry et al., 1984). In the Singapore Strait, the concentration averaged 5.6×10^4 cells ml^{-1} (range 4.0×10^4-7.4×10^4 cells ml^{-1}) while in the Johor Strait, average concentrations were about 22% lower, at about 1.2×10^4 cells ml^{-1}. Earlier studies have shown that *Synechococcus* is generally a poor competitor in eutrophic ecosystems (Gin, 1996), whereas it usually thrives in nutrient-depleted environments due to its higher surface area to volume ratios, as well as its lower subsistence quota and high growth rate (Chisholm, 1992). Other studies have also shown that the rapid growth of *Synechococcus* in oligotrophic waters is supported by the development of a high-affinity nutrient uptake system (e.g. phosphate uptake), with a similar mechanism as some heterotrophic gram-negative eubacterial cells (Scanlan et al., 1993). The lower nutrient availability in the Singapore Strait (phosphate ~0.009 mg l^{-1}) could explain the relatively higher abundance of *Synechococcus* there compared to the Johor Strait (phosphate~0.04 mg l^{-1}).

In the Singapore Strait, we found two different strains of *Synechococcus*, defined by their phycoerythrin chromophore composition and distinguished by their flow cytometric fluorescence signatures (Figure 5b). Similar dual populations have been observed in the North Atlantic and Pacific Oceans (Olson et al., 1990), and the Arabian Sea (Campbell et al., 1998). *Synechococcus*, with a high phycourobilin (PUB): phycoerythrin (PEB) ratio ('bright'), emit brighter orange fluorescence than those with a low PUB:PEB ratio ('dim'), since 488 nm excitation is absorbed by PUB more efficiently (Olson et al., 1988; Olson et al., 1990). None of the "bright" *Synechococcus* were found in the Johor Strait and the relative mean orange fluorescence for the "dim" *Synechococcus* was much lower than its counterpart in the Singapore Strait. One reason to explain these results is the difference in optical characteristics. In the Johor Strait, higher turbidity and dissolved substances will lead to the rapid attenuation of blue light and a preferential transmission of green-to-yellow light (Olson et al., 1991). At the same time, larger species containing pigments that more efficiently absorb light in the green wave band (e.g. diatoms with fucoxanthin) will thrive, which in turn further increases the turbidity. Hence, only "dim" *Synechococcus* with low PUB:PEB ratios would be able to sustain growth by absorbing green light. In contrast, the existence of "bright" *Synechococcus* in the Singapore Strait demonstrates a less turbid environment where more blue light can penetrate the water column and where the "bright" strain (with higher PUB:PEB ratio) can outcompete the "dim" strain.

Overall, it appears that the ratio of "bright" and "dim" *Synechococcus* is inversely correlated with eutrophication (and subsequent blue light attenuation) of coastal waters. An inverse correlation is observed between total nitrogen concentration and the "bright":"dim" *Synechococcus* ratio (r^2=0.8024) (Figure 7), suggesting that this ratio could be used as a potential indicator for assessing trophic states in coastal waters.

4. HARMFUL ALGAL BLOOMS (HAB)

HAB span a broad range of scenarios including: (1) hypoxia produced from the microbial decomposition of high biomass blooms, (2) the production of highly potent toxins that directly kill marine organisms, and (3) the production of toxins that accumulate through marine food chains to poison marine mammals, seabirds and humans (e.g., the toxins that cause the seafood diseases known as paralytic shellfish poisoning (PSP), diarrhetic shellfish poisoning (DSP), neurotoxic shellfish poisoning (NSP), amnesic shellfish poisoning (ASP), azaspiracid shellfish poisoning (AZP) and ciguatera). These three potential scenarios are examined in relation to Singapore waters.

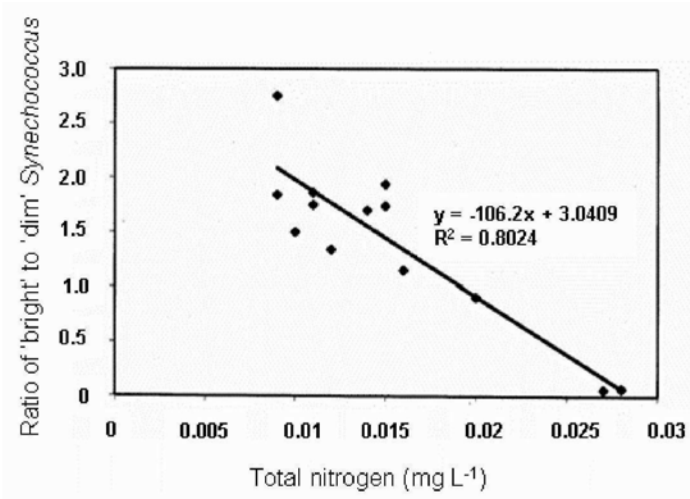

Figure 7. *Inverse correlation between the ratio of "bright" to "dim" Synechococcus and total nitrogen (Singapore Strait).*

(1) Loss of dissolved oxygen from the decay of phytoplankton blooms is most likely in the Johor Strait because of the high biomass (> 40 μg l^{-1} chl-*a*) blooms that frequently occur in this eutrophic waterway. Small fish kills do occur, especially in or near the small-scale aquaculture farms that exist along this waterway, although there is no direct evidence of their cause. Oxygen levels near the bottom of Johor Strait can fall to less than 2 mg l^{-1} in some parts of this Strait. Hypoxia is much less likely in the less nutrient-enriched and better-mixed waters of the Singapore Strait.

(2) A number of dinoflagellate species produce toxins that kill marine organisms, with probably the best known being the brevetoxins produced by *Karenia brevis* in the Gulf of Mexico that are responsible for killing fish and manatees (Steidinger et al., 1998). However, in recent years increasing numbers of these small, unarmoured "gymnodinoid" dinoflagellates, such as *K. brevisulcata*, *K. digitata*, *K. mikimotoi* and *Karlodinium veneficum,* have been linked to major fish kills. In South East Asia, *Karenia digitata* was responsible for massive losses of aquacultured fish in Hong Kong (Yang et al., 2000). Singapore waters contain many of these small "gymnodinoid-like" species which sometimes form blooms in both the Singapore and Johor Straits. These include species of *Karenia* and *Karlodinium* (YZ Tang, personal communication) but as yet we have no evidence of their causing fish kills. Since the revision of the genera *Gymnodinium* and *Gyrodinium* by Daugbjerg et al. (2000), and the subsequent erection of several new genera (including *Karenia* and *Karlodinium*), this has become a "hot" field of research with many new species being discovered. Work is underway to identify and characterize the fish-killing ability of these isolates from Singapore. Another unarmoured dinoflagellate, *Cochlodinium* sp. is commonly found in the Johor Straits and this species has been linked to fish kills in this waterway (Khoo and Wee, 1997). The species that occurs in the Singapore waters has not yet been identified although *C. polykrikoides* has

been associated with fish kills in North East Asia, possibly through production of superoxide radicals (Kim et al., 1999).

Raphidophytes are golden-brown bi-flagellated cells, a few species of which are associated with fish-killing ability (Hallegraeff and Hara, 2003). Singapore waters contain at least four species; *Chattonella marina*, *C. subsala*, *Fibrocapsa japonica* and *Heterosigma akashiwo* with all of these species except *C. subsala* having been linked to fish-kills in various parts of the globe. The fish killing mechanism is poorly understood with suggestions including the physical clogging of gills by mucous secretion, production of haemolysins, brevetoxins, hydroxyl and superoxide radicals and combinations of these (Hallegraeff and Hara, 2003; Marshall et al., 2003). Kills of caged fish from the East Johor Strait in 1989 were attributed to blooms of the dinoflagellate *Gymnodinium catenatum* but a *Chattonella* species was apparently also present (Khoo and Wee, 1997). However, cultures of the four raphidophyte species isolated from Singapore waters several years after this fish-killing incident were found to be non-toxic to fingerlings of Asian seabass, *Lates calcarifer* (Tang and Holmes, 2004), although all four species do produce copious amounts of mucous. However, as blooms of these species occur in the Johor Strait, a waterway used for fish aquaculture (Chou and Lee, 1997), continued monitoring is necessary. This is not simple for raphidophytes (and unarmoured dinoflagellates) because they preserve poorly in most fixatives.

(3) Singapore waters contain a number of marine dinoflagellate species that produce toxins that can accumulate through food chains to cause human poisoning (reviewed by Holmes and Teo, 2002). In South East Asia, the major HAB problem has been paralytic shellfish poisoning (PSP) caused by contamination of shellfish by toxins produced by the dinoflagellate *Pyrodinium bahamense* var. *compressum* (Hallegraeff and Maclean, 1989). The PSP-toxins comprise a family of more than 20 analogs of highly potent, water-soluble neurotoxins, known collectively as saxitoxins because saxitoxin was the name given to the first structural analog elucidated (Luckas et al. 2003). These toxins are produced by dinoflagellate species belonging to the genera *Alexandrium*, *Gymnodinium* and *Pyrodinium,* although each toxin-producing species typically only synthesizes a small subset of the total number of known toxins. Although *P. bahamense* var. *compressum* has caused the majority of PSP cases in South East Asia, this species has not been observed in Singapore waters (Holmes and Teo, 2002), or from the adjacent Straits of Malacca (Usup et al., 2002). The distinctive morphology of this species makes it difficult to miss in plankton monitoring programs so it is likely that it is truly absent from the Singapore region although blooms recur almost annually on the eastern side of the South China Sea, along the west coast of Sabah (Usup et al., 2002). PSP has occurred in Singapore waters with five people poisoned and two deaths from eating green mussels collected from the Johor Straits (Tan and Lee, 1987) but the causative organism was never identified. The unarmoured, chain-forming dinoflagellate *Gymnodinium catenatum* (Figure 8) produces PSP-toxins and this species has been found intermittently in Singapore waters since 1997 (Holmes and Teo, 2002, Holmes et al., 2002) but with blooms possibly seen 10-years earlier (Khoo and Wee, 1997). However, the early identification was made before the existence was known

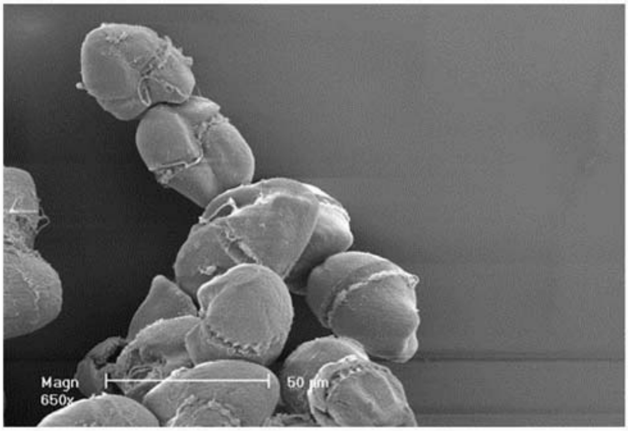

Figure 8. *Scanning electron micrograph of the chain-forming, unarmoured dinoflagellate Gymnodinium catenatum from the port of Singapore. This species produces paralytic shellfish poisoning (PSP) toxins.*

of a morphologically similar, but non-toxic, chain-forming species, *G. impudicum*. Holmes and Teo (2002) reported the presence of *G. impudicum* blooms in Singapore waters and at least in recent years, blooms of chain-forming *Gymnodinium* species can mostly be attributed to *G. impudicum*. However, as no other PSP-producing dinoflagellates have been yet found from Singapore waters, *G. catenatum* remains the most likely source for the lethal poisonings that occurred in the 1980's. Two other potential candidate species are *Alexandrium minutum* and *A. tamiyavanachii* since the former has been found from the north east coast of Peninsula Malaysia and the latter from the west coast along the southern Straits of Malacca (Usup et al., 2002). However, to-date, the only *Alexandrium* species found in Singapore waters has been the non-PSP producing *A. leei* (Tang and Holmes, unpublished).

The discovery of *G. catenatum* in Singapore waters was initially surprising although the species had been reported from the Gulf of Thailand and the Philippines. This is because the species is generally associated with temperate waters (Hallegraeff and Fraga, 1998). The species causes major PSP problems on the Atlantic coast of the Iberian Peninsula, the Pacific coast of Mexico, Tasmania in southern Australia and Southern Japan (Hallegraeff and Fraga, 1998). A possible explanation for the disjunct distribution of *G. catenatum* is that it may have been translocated by ship ballast water (Hallegraeff, 1993; McMinn et al., 1997). A relatively recent global spread of this species (in geological terms) is supported by the discovery of identical large subunit rDNA sequences for isolates of all global populations so far examined (from Australia, China, Japan, Korea, New Zealand, Singapore, Spain and Uruguay) (Holmes et al., 2002). As Singapore has one of the busiest ports in the world, it is possible that the species has been introduced via ballast water. However, the toxin profile (i.e., relative proportions of each toxin analog) of the Singapore isolates is unique, being dominated by carbamoyl derivatives of saxitoxin (especially gonyautoxins-1 and -4), unlike all other global populations so far examined which are dominated by less potent sulfocarbamoyl derivatives (Holmes et al., 2002). As the toxin profiles produced by *G. cat*enatum isolates are thought to be stable (Oshima et al., 1993), it suggests that if the Singapore strain has been translocated to this port, then the source population is different from those so far examined around the globe. In any case, the carbamoyl-dominated toxin profile may be a useful signature to help identify any spread from the port of Singapore (Holmes et al., 2002). However, any spread via ballast water would likely be from transfer of the resting cyst stage (hypnozygote) of the life cycle of this species since vegetative cells are unlikely to survive for extended periods in ballast tanks (Hallegraeff, 1993, McMinn et al., 1997), and as yet no cysts have been observed from Singapore isolates (Holmes and Bolch unpublished).

Dinoflagellates that produce diarrhetic shellfish poisoning (DSP) toxins also occur in Singapore waters although as yet, no cases of DSP have been identified from Singapore (Holmes and Teo, 2002). DSP is caused by eating shellfish contaminated by a family of lipid-soluble toxins that are structural analogs of okadaic acid (Quilliam, 2003). DSP-toxins are produced by a number of planktonic dinoflagellates belonging to the genus *Dinophysis*/ *Phalacroma* as well as some benthic species belonging to the genus *Prorocentrum* (Holmes and Lewis, 2002). The *Dinophysis* species that produce these toxins are thought to be the major cause of DSP globally with only one benthic species, *P. lima* so far implicated in contamination of shellfish (reviewed by Holmes and Teo, 2002). Four dinophysoid species have been reported from Singapore waters, *D. caudata*, *D. ovum*, *D. sacculus* and *P. rotundum* (Holmes and Teo, 2002), although *D. sacculus* may be a misidentification of a *Metadinophysis* species (Tang and Holmes, unpublished). Holmes et al. (1999) were the first to identify *D. caudata* as a producer of DSP toxins although the cellular concentrations they found were extremely low. They were also the first to prove the accumulation of DSP toxins in tropical shellfish when they found that green mussels from the Johor Straits were persistently contaminated with low concentrations of okadaic acid. The green mussels accumulate these toxins by feeding upon *D. caudata*, which is the most abundant dinophysoid species in Singapore waters (Holmes et al., 1999). Subsequently, it was discovered that *D. caudata* (and *D. miles*) from the Philippines can produce much higher toxin concentrations than so far found from Singapore samples (Marasigan et al. 2001). *Dinophysis ovum* is the only other dinophysoid species from Singapore so far tested for DSP toxins but okadaic acid could not be detected (Holmes and Teo, 2002). Although *D. caudata* is the most abundant dinophysoid species in Singapore waters, to-date no large blooms have been observed.

5. EUTROPHICATION MODELING

While field measurements are necessary for characterizing baseline conditions, they are limited in that measurements are discrete, expensive and labour-intensive to obtain. Numerical modeling can help to avoid some of these problems and also allow the prediction of impacts and future scenarios, given perturbations to the system. A three-dimensional, multi-level eutrophication model was developed to simulate the water column characteristics and sediment nutrient fluxes in the tropical coastal waters of Singapore (Gin et al., 2000). The model takes into account six interacting systems – the nitrogen, phosphorus and carbon cycles, phytoplankton and zooplankton dynamics and dissolved oxygen balance. The conceptual framework for the eutrophication kinetics in the water column is based on the WASP model and BEM model (Ambrose et al., 1993; HydroQual et al., 1995) with some modifications. In total, thirteen state variables were used to simulate the above interacting systems: ammonia, nitrate, phosphate, phytoplankton, carbonaceous biochemical oxygen demand, dissolved oxygen, particle organic nitrogen, dissolved organic nitrogen, particle organic phosphorus, dissolved organic phosphorus, particle organic carbon, dissolved organic carbon and zooplankton. In addition, interactions between the water column and sediments were accounted for using a two-layer sediment model, representing the aerobic and active anaerobic layers of the sediment (Di Toro and Fitzpatrick, 1993). Advective transport velocities and eddy viscosity parameters were provided using a three-dimensional multi-level hydrodynamic model.

The eutrophication model has been calibrated and validated with field data from the Southwest Monsoon and Northeast Monsoon, so as to reproduce the general features of Singapore's coastal waters. In the model calibration, a 14-day hydrodynamic period generated from the three-dimensional hydrodynamic model was used. The hydrodynamic condition corresponded to one of the Monsoon periods and included a spring tide and a neap tide. The model parameters were divided into two sets. One set of parameters was directly obtained from experiments or field measurements from the coastal waters of Singapore (Tham, 1953; Gu, 1998; Cheong, 1999). These values were kept unchanged in the process of model calibration. The second set of model parameters were initially estimated from the literature (Thomann and Fitzpatrick, 1982; Ambrose et al., 1993; Gu, 1998). These were adjusted or tuned until a reasonable reproduction of the field data (nutrients and plankton) at observation stations was obtained. The calibrated model was then verified with data from the Northeast Monsoon. The chi-square goodness of fit test was used to assess the model calibration results (Schnoor, 1996) and some representative results are shown in Table 4. These results show that all the selected variables pass the goodness of fit test at a 0.01 significance level.

Overall, the computed results of nutrient and phytoplankton concentrations agree reasonably well with the observed data. The simulated results of nutrients and phytoplankton showed that Singapore's coastal waters were relatively well-mixed and the differences in concentration with depth for all state variables were generally less than 20%. The nitrogen, phosphorus and phytoplankton concentrations in the northern part of Singapore (Johor Strait) were much higher than those in the Singapore Strait, whereas dissolved oxygen showed the opposite trend, consistent with the observations in field data. One of the problems, however, with the modeling study was the lack of field data for calibrating sediment fluxes. Hence, future studies will require sediment data for better calibration of the model.

Table 4. *Comparison between field data and numerical results for data from the Northeast Monsoon. The Chi-square (χ^2) goodness of fit test was used to assess the degree of fit between the model results and field data. In order to accept the model results as a good fit, $P(\chi^2 \leq \chi_o^2) = 1-\propto$, where $\propto$ is the confidence level and χ_o^2 is the Chi-square distribution value for(n-1) degrees of freedom.*

Variable	Measured[a] (mg l^{-1})	Numerical[b] (mg l^{-1})	Error (mg l^{-1})	Relative error (%)	Chi-square critical values (χ_o^2)	Chi-square statistic (χ^2)	Degree of freedom (n-1)	Confidence level ($\propto$)
Ammonia	0.024	0.020	0.004	17	2.558	0.0247	10	0.99
Nitrate	0.036	0.048	-0.012	-33	2.558	0.0868	10	0.99
Phosphate	0.011	0.009	0.002	18	1.646	0.0220	8	0.99
Organic Nitrogen	0.668	0.726	-0.058	-7	2.558	0.1300	10	0.99
Organic Phosphorus	0.005	0.005	0	0	1.646	0.0116	8	0.99
Phosphate	0.098	0.095	0.003	3	2.558	0.1930	10	0.99

[a]obtained from horizontally averaged field data at all stations
[b]obtained from 14-day horizontally averaged numerical results at stations

One of the benefits of having such a model lies in the prediction of impacts when potential new nutrient sources are introduced into the marine environment. The model allows for the prediction of a number of scenarios based on different treatment options, source location, etc, in order to minimize negative impacts and optimize benefits for all coastal users. Control measures can thus be implemented to prevent algal blooms and detrimental low levels of dissolved oxygen. Apart from ocean outfall applications, the eutrophication model can also be applied to waste discharges from ships. Some of the major concerns include accidental spills and ballast water exchange. In the latter case, the transport and exchange of toxic/nuisance organisms through ship ballast water has become a major concern for international ports in recent years. These foreign species include potential "harmful algal bloom" species, which can proliferate if given the right environmental conditions and there is already evidence for the successful translocation of at least two toxic HAB organisms via ballast water (McMinn et al., 1997, Lilly et al., 2002).

The current model can be used to simulate the environmental conditions that could trigger such a bloom. However, more research is needed to determine the life cycles of the species involved, their survival in ballast water and interaction with indigenous organisms before a complete model can be developed for risk assessment.

6. CONCLUSIONS

In the last 30 years, the coastal waters of Singapore have shown gradual increases in primary production in line with its social and economic growth. Using data collected in the last 6 years, the phytoplankton structure in the Singapore Strait has been shown to differ considerably from that of the Johor Strait. In general, waters in the Johor Strait are more eutrophic than the Singapore Strait, with chlorophyll levels reaching as high as 60 $\mu g\ l^{-1}$, consistent with the higher nutrient concentrations measured in the Johor Strait. This can be partially explained by the high and variable levels of anthropogenic inputs, poor circulation and shallow waters in the Johor Strait and partially explained by the better flushing characteristics found in the Singapore Strait. Lower dissolved oxygen levels were also observed in the Johor Strait compared to the Singapore Strait, especially for bottom waters. Using simple enrichment tests, we found that Singapore waters were generally nitrogen limited. However, for the Johor Strait, variable anthropogenic inputs and a N:P ratio close to the Redfield Ratio implies that nutrient limitation can easily switch to phosphorus. The size structures of phytoplankton in the Johor Strait were skewed to larger microplankton whereas in the less nutrient-enriched Singapore Strait, smaller pico and nano-plankton dominated. In particular, we observed significant populations of *Synechococcus*, the small ubiquitous picoplankton typically found in tropical open oceans. The ratio of "bright" to "dim" *Synechococcus* (determined by flow cytometry) may be used as a potential indicator for assessing the trophic state of coastal waters.

Compared to the Singapore Strait, algal blooms are a frequent occurrence in the Johor Strait. In spite of the frequent blooms, there are few documented cases of harmful algal blooms (HAB). However, many toxic and potentially toxic species of dinoflagellates and raphidophytes do exist in these, and neighboring waters, with the paralytic shellfish poisoning (PSP) toxin-producing dinoflagellate *Gymnodinium catenatum* probably of most concern at present. Both PSP and diarrhetic shellfish poisoning (DSP) toxins have been found in mussels, indicating the need for continued monitoring.

The strategy for preventing eutrophication problems lies in the strict control of nutrient discharges coupled with comprehensive monitoring and numerical modeling capabilities for sound management of the coastal environment. As Singapore's economy continues to expand, the successful management of Singapore's marine environment is critical. Land reclamation, expanding coastal and port developments, ocean outfalls and damming of rivers and estuaries are expected to increasingly impact the coastal environment in future years. A field monitoring programme is already in place to establish trends and also to provide data for continuous upgrading of the numerical model. These integrated measures, together with cooperation of government agencies will allow for the continued successful management of Singapore's coastal environment.

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CHAPTER 22

MARINE HABITATS IN ONE OF THE WORLD'S BUSIEST HARBOURS

LOKE MING CHOU

1. INTRODUCTION

Singapore's limited marine territory of approximately 600 km^2 supports one of the world's busiest harbours. Over 133,000 vessels (above 75 gross tonnes) called at the port in 2004. Thirty-five percent comprised regional ferries linking Singapore with various ports of Indonesia's Riau province. Other vessels were mainly containers and tankers which facilitate movement of 1 billion gross tonnes of cargo through the port. To accommodate such intense shipping activity and provide safe navigation to the heavy vessel traffic, the port waters cover 82% of Singapore territorial waters. Most of the port provisions are located in the seas south of the main island, particularly the south-western sector (Figure 1). Wharf and berthing facilities dominate the southwestern coastline, which have been transformed by coastal reclamation. Fairways, anchorages and vessel manoeuvring areas intermesh between the fifty-odd smaller offshore islands scattered mostly in the southern waters.

A spectrum of marine habitats typical of tropical regimes is found in this heavily utilized marine environment. Fringing and patch reefs are present among the southern islands together with seagrass meadows and mangrove stands, all varying in areal extent. A range of seashore types including rocky, sandy and muddy exists. The seafloor is now mostly dominated by mud bottoms as most of the sand deposits have been exhausted for reclamation while the continuous rain of suspended sediment adds more silt.

Threats from shipping include vessel movement, grounding and accidental spillage of hazardous materials. Other activities such as land reclamation, seabed dredging and the dumping of dredged spoils contribute to increased suspended sediment, modification of hydrodynamic patterns and changed tidal speeds at different locations. High energy wash from increasing numbers and operating frequency of fast-going vessels such as high-speed passenger ferries scour exposed shores and reef flats. Large tracts of razed corals on reef flats provide evidence of grounding by flat-bottom barges, often connected with a nearby development activity. The spillage of 28,500 tonnes of heavy marine fuel oil into the sea

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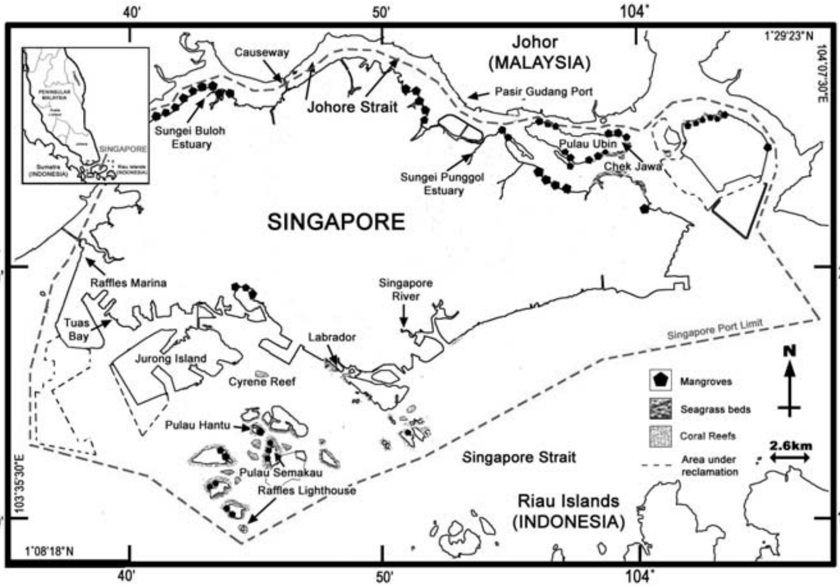

Figure 1. *A map of Singapore and its port limit. Its share of the northern Johor Straits is very limited. Waters to its south are used extensively to support shipping and port-related services. (Inset shows location of Singapore in relation to Malaysia and Indonesia).*

resulting from the collision of two oil tankers, the "Evoikos" and the "Orapin Global" in October 1997, the largest in Singapore's history, firmly emphasised that this ever-present threat can happen at anytime even under fine weather conditions and with sufficient warning given to the approaching vessels. In August 2000 and July 2001, two separate incidents of phenol spills occurred in the Malaysian waters of the east Johor Straits, killing fish and mussels in Singapore waters as well. Both resulted from capsized cargo vessels transporting the toxic chemical from the Malaysian port of Pasir Gudang.

Shipping also introduces the problem of invasive species. The exotic *Mytilopsis sallei*, which is of Caribbean origin, is now the most commonly-occurring bivalve in Singapore (Sachidhanandam and Chou, 1996), dominating walls of intertidal monsoon canals. Tolerant of large salinity fluctuations, they were possibly introduced by ships coming from across the Panama Canal (Tan and Chou, 2000). Imposex in females of muricid gastropods indicate organotin contamination, which is not surprising considering the shipping intensity (Tan, 1999). Exposed to the diversity of impacts, Singapore's marine habitats face three major problems: habitat loss, habitat degradation and habitat modification.

2. MARINE LIFE

The richness of the country's marine biodiversity has been well documented for groups that received close attention. Whilst not all groups are well studied, historical records point to a rich biodiversity (Chou, 1994). At the same time, updated collections of many groups studied in the past repeatedly revealed species missed earlier, demonstrating the need for more exhaustive field surveys. Earlier records of gastropods and bivalves from 1843 to 1878 formed a small fraction of all species established by Chuang (1973a), supporting the conclusion that earlier collections were not sufficiently comprehensive. For groups that were studied more thoroughly in the past, the indications are little to no decline in species richness, but an evident depreciation of abundance. Marine fish species composition between 1934 and 1973 showed no loss of species but definitely less abundance (Tham, 1973).

Despite an expansion of activities impacting the marine environment since the late 1960s, leading to habitat loss and degradation, the reduction in species diversity appears to be less than expected, although frequency occurrence and abundance of many species have declined. Coastal modification has replaced natural habitats along almost the entire shoreline of the southern half of the main island and the northeastern sector. These modifications have resulted in sandy beaches with a steeper gradient along the southeastern shores designated for recreation use, and concrete structures including seawalls along most of the southwestern shoreline to support port operations (Figure 2). These new human-made habitats have a strong influence on the structure of biological communities as they displace species established at the original habitats in favour of those able to thrive in the changed environment.

Figure 2. *Part of the container port along the southwest coast. The original coastal habitat is buried by coastal reclamation and replaced by concrete structures.*

2.1. Natural habitats

All habitats would have been influenced to some degree by the various anthropogenic impacts and those described as pristine are in reality, comparatively better in species diversity and abundance than those with higher exposure to impacts.

2.1.1. Open water

There is more open water south of the mainland, but the large smattering of offshore islands in the south punctuates it with shallow inshore waters blurring the distinction. Tham (1973) suggested that fish populations could be conveniently divided into those commonly occurring in inshore waters up to 18.5 m, in waters beyond 18.5 m, in coral reefs, and in the Johor Straits. The inshore waters are fairly rich with over 150 species representing at least fifty families, and there were some differences in species composition between the north and the south. He concluded that compared to 1934, there was no change in species composition and no species loss, but a decline in abundance based on catch data.

The narrow Johor Straits separating Singapore from Malaysia is relatively sheltered. A causeway linking both countries severely curtails circulation resulting in thermal stratification that suggests stagnant waters (Lim, 1984a and b). The east Johor Straits in particularly is impacted by coastal development of both countries and increased vessel traffic sailing to and from the Malaysian port of Pasir Gudang, which began operations in the early 1990s. A recent study of inshore fish from the east Johor Straits revealed 133 species from 46 families, comprising juveniles primarily and adults of small-sized fish (Hajisamae and Chou, 2003). Seasonal variations in abundance and species richness occurred and the Straits served an important nursery ground for numerous species in spite of the anthropogenic impacts. A wide variety of prey dominated by calanoid copepoids was available (Hajisamae et al., 2003). Habitat restoration on the Singapore side included artificial sandy beaches and mangrove reforestation. Fish density was higher at the reclaimed sandy shore while the reforested mangrove supported greater species richness (Jaafar et al., 2004).

No recent studies were conducted on fish in waters beyond the 18.5 m depth, but occasional news reports over the last ten years of large fish such as a 5.3 m live sawfish caught by local fishermen, confirm their presence in spite of the intense vessel traffic.

Sightings of the Indo-Pacific Humpbacked dolphin, *Sousa chinensis*, are fairly common within port waters, but that of other marine mammals are rare. A 3.75 m female false killer whale, *Pseudorca crassidens*, appeared trapped for a week within Tuas Bay, a large man-made bay, in January 1994. After refusing attempts to guide it out of the bay, its carcass was found outside the bay with body injuries. Dugongs are rarely seen but confirmed by occasional beached carcasses on the northern shore, often with injuries consistent with propeller cuts.

2.1.2. The seafloor

Much information on soft-bottom benthic communities was generated from the ASEAN-Australia Living Coastal Resources project from 1986 to 1994. The surveys provided a better understanding of benthic communities and established suitable baseline data of what was previously largely unknown. First records of 29 polychaete species were described (Tan and Chou, 1993) together with reports of three new species subsequently (Tan and Chou, 1994, 1996, 1998). New records of three rare brachyuran and one new anomuran crabs were published (Chia and Ng, 1993; Ng and Nakasone, 1994). These reveal that with the discovery of new species in an impacted marine environment even at this advanced stage, others that are potentially new to science could likely have been exterminated by the rapid physical changes.

Studies indicated differentiation between estuarine and offshore locations (Khoo, 1990). Riverine bottoms were mainly mud with polychaete diversity highest, followed by bivalve or gastropod diversity (Chong and Loo, 1990; Chua and Loo, 1990; Goh and Loo, 1990; Quek and Chua, 1990). Offshore habitats which were sand or sandy-mud showed variation in faunal diversity dominance between polychaetes and crustaceans with evidence of more epifaunal diversity than infaunal (Chou and Loo, 1994; Chung and Goh, 1990; Lim and Gan, 1990; Lim and Koh, 1990).

Community structure change in response to localised impacts was rapid. Benthic community diversity at Pulau Semakau, a southern offshore island, declined between 1989 and 1993 (Chou and Loo, 1994). This was attributed to the dumping of dredged spoils which covered the bottom with thick layers of silt. In a further study in 1996, when an extensive rock bund was constructed to enclose the sea on its eastern side for use as a sanitary landfill, changes in the abundance, familial diversity and abundance of benthic invertebrate fauna were correlated with sedimentation rate, sediment composition and ammonia concentration (Chou, Yu and Loh, 2004). In the mangrove estuary of Sungei Buloh, distinct changes in benthic community structure were detected between 1990, when 87 hectares were gazetted as a nature park, and 1992. Prior to 1990, all farming activities (pig, poultry and shrimp) around the area were steadily phased out. The changes were indicative of a response to improved environmental quality (Goh et al., 1994). Similarly, a study of macrobenthic infauna at Sungei Punggol, an estuarine river in the north, showed that coastal reclamation had a damaging effect on the community structure of macrobenthos within the vicinity but resulted in significant increase in familial diversity and abundance away from the impact (Lu et al., 2002).

2.1.3. Seashores

The seashore conditions in the 1970s were adequately described by Chuang (1973b). Since then, the shore profiles have been extensively modified by coastal reclamation. About the only original beach along the southern coastline is a short stretch of rocky shore at Labrador with seagrass patches and a reef community. Its existence remains threatened by future extension of the container port to its west.

The beach was used until the recent past for the docking of small tankers to discharge oil through pipelines to storage tanks further inland. A section of the beach was reclaimed and transformed into a recreational park. At the opposite end, thermal effluent was discharged from a power generating plant, which has since been decommissioned. These past impacts would have affected the biological communities, but what is still present demonstrates some degree of ecosystem resilience or remarkable recovery ability.

Very few natural beaches remain and they are located mostly along the northwestern sector of the main island and some of the offshore islands. A rich intertidal flat that represents what most intertidal flats were like in the past is Chek Jawa, located at the eastern tip of Pulau Ubin in the Johore Straits. In spite of coastal reclamation adjacent to it, sediment carried down the Johor River, and the wash from passing vessels, the flat supports seagrass, corals and a rich diversity of marine life.

2.1.4. Mangroves

From early history, mangroves were cleared from the Singapore River for the development of port facilities. Most of the mangroves along the main island's southwestern coast have been converted for industrial and ship-related activities. Less than 1% of the original mangrove forests remain in patches along the north coastline and the offshore islands (Low and Chou, 1996) (Figure 3). Establishment of mangroves appears to be opportunistic in the southern islands with *Rhizophora* seedlings rooting within crevices of sloping seawalls. Mangrove stands in the southern islands remain healthy in spite of the heavy shipping traffic. Growth is also vigorous (due mainly to the warm equatorial climate) based on reforestation projects.

A large reforestation programme was initiated in the late 1990s at Pulau Semakau. The construction of a containment seawall for a landfill site destroyed much of the mangrove forest on its eastern shore. Reforestation was carried out at both, north and south of the connection to replace the loss. Growth of seedlings was vigorous and the effort can be considered successful.

2.1.5. Seagrass

Seagrass habitats along the coast of the main island have largely been lost to coastal reclamation and are restricted in distribution (Loo et al., 1996). Nine species have since been reported, but it is suspected that the diversity is higher than this. Larger seagrass beds are associated with the offshore islands, both in the north and south. They are extensive on some reef flats such as Cyrene Reef and west Semakau. A large and somewhat pristine seagrass bed exists at Chek Jawa on the eastern tip of Pulau Ubin in the north. Relatively unknown, its presence was highly publicised when the entire area was about to be reclaimed. Increased traffic of ships calling at the Malaysian port of Pasir Gudang and passing close to the site has apparently not

Figure 3. *Relatively pristine mangrove forests occur in the northern offshore islands. They are now exposed to waves created by large vessels sailing through the Johor Straits to the Johor port of Pasir Gudang.*

much impact on this habitat. Plans to reclaim Chek Jawa were suspended and National Parks is presently responsible for its management as a nature area.

In the artificial lagoons of some of the southern islands, colonisation by seagrass has occurred but at a slow pace. A study is in progress to review the current status of this habitat. Shipping does not seem to have a large impact but the threat from accidental oil spills remains.

2.1.6. Coral reefs

Many investigations were carried out on Singapore reefs and a monitoring programme since 1987 indicates the response of reefs to the diversity of impacts. Approximately 60% of reefs have been lost to reclamation and the remaining ones subjected mainly to sediment impact. Regular dredging of shipping channels and dumping of the spoils further offshore add to the suspended sediment load generated from land reclamation. The construction of berthing facilities and jetties has also caused degradation of some reefs. Direct shipping impacts come from waves generated by vessels, particular the high speed ferries in recent years. Reefs crests and flats in the path of these routes are exposed to increased wave energy, which affects benthic stability. Another recent observation is direct physical damage of shallow reefs by construction or transport barges, which destroy considerable tracts of reefs.

Rapid light attenuation of the water column has prevented coral survival below 6 m depth. Growth is now restricted to the upper reef slope and reef crests support the best reef development. Long-term monitoring since 1986 indicated an overall decline in live coral cover (Table 1) as well as a reduction in the abundance of reef-associated invertebrates, although a few reef sites showed improvement. No comprehensive studies on species extinctions have been attempted but of the close to 200 hard corals, only two, *Stylophora pistillata* and *Seriatopora hystrix* have not been seen in the past fifteen years. There is evidence of morphological change in some coral species in response to sedimentation (Todd et al., 2001, 2004). Reef processes however are not totally interrupted. Mass spawning occurs annually (Guest et al., 2002, 2005) and pomacentrids recruit biennially (Low et al., 1997).

Table 1. *Temporal variation in live coral cover (%) at the reef crest of selected reefs shown in Figure 1.*

Reef	1986-89	2000-03
Cyrene (Site 1)	10.2	21.1
Cyrene (Site 2)	20.1	25.7
Hantu West (Site 1)	41.5	38.8
Hantu West (Site 2)	70.4	13.6
Pulau Hantu (Site 1)	31.4	8.6
Pulau Hantu (Site 2)	49.1	46.8
Pulau Semakau (Site 1)	26.1	23.4
Pulau Semakau (Site 2)	51.8	21.4
Raffles Lighthouse (Site 1)	76.6	56.7
Raffles Lighthouse (Site 2)	76.1	48.8

2.2. Human-engineered habitats

Shorelines have been extensively modified by coastal engineering. New habitats are created, some of which support biodiversity enhancement, while the rest do not. In all cases, species associated with the original habitats are invariably affected, resulting in distribution pattern changes.

2.2.1. Sea walls and coastal structures

The development of sea walls to contain reclaimed land and provide berthing for vessels eliminated the original shore profile extensively along the southwestern coastline and many of the offshore islands (Figure 4). The seaward face of these walls offers limited opportunities for colonisation by marine communities because of their vertical or steep incline and homogeneity of substratum quality. The diversity at these walls is confined commonly to filamentous algae, oysters, barnacles, serpulid worms, bryozoans as well as gastropods and other bivalves typical of intertidal rocky shores.

Figure 4. *Reef-flats of some of the southern offshore island were transformed to sand-bottom swimming lagoons. They are protected by retaining seawalls.*

Concrete wharf pilings, which have less surface area for colonisation support a greater diversity as water currents, light penetration and other physical factors vary more around them. Growth of soft corals, sea fans and tunicates is usually profuse. Concrete jetty piles at the southern islands favour growth of corals and reef-related species. At one of these islands, coral cover on jetty piles was almost twice that of the natural reef slope, and coral species diversity more than double that of the slope (Chou and Lim, 1986). The greater colony number, size range and depth range on the piles showed that they were better for coral development. Coastal structures can promote biodiversity. The different niches and habitat complexity provided by the variety of structures at Raffles Marina on the northwest coast supported far more species than in the surrounding sea. The fish community of over sixty species represented different trophic levels and niche specialisations. Of interest was the presence of seahorse and archer fishes. The floating pontoons, sea walls and pilings supported a highly diverse epibiotic community.

2.2.2. Artificial beaches and swimming lagoons

Most of the recreational beaches on the main island are created from coastal reclamation. They have a steeper profile than the original beaches buried by the reclamation. The extent of intertidal flats is also reduced. These changes reduce the abundance of intertidal shore organisms that recolonised the new beaches. Fine-grained sand is used for some of these beaches. Below the water line, silt quickly clogs up the interstitial spaces between the sand grains resulting in deoxygenation of layers not far below the surface.

Protected swimming lagoons were constructed at a number of the southern islands, replacing existing reef flats. Reef flats were excavated and buried by sand to form the lagoon bottom, eliminating the original reef community and creating conditions for sandy bottom organisms. In many of these lagoons, the sand bottom is changing to a mud bottom with the accumulation of trapped sediment.

2.2.3. Artificial reefs

A project using tyre-pyramids and hollow concrete frames as artificial reef modules was carried out between 1989 and 1996. These structures were deployed on the seabed at a 12 m depth close to a reef system in the southern islands, facing a shipping lane. Observations revealed rapid colonisation of the structures by encrusting communities, with the concrete frames supporting a greater diversity. Fish communities established quickly at the concrete frames and increased in species richness and abundance before levelling off after the seventh year. The tyre reef was used as a nursery and colonized by larvae and juveniles, while the concrete reef favoured adults. It was concluded that the artificial reefs did not greatly affect the fish populations on the adjacent reef.

A recently concluded three-year project using fiberglass structures coated with calcium carbonate on the exterior surface showed that corals recruited on them and coral transplants attached to these structures established basal plates and grew well (Loh et al., in press). These modules (Figure 5), referred to as Reef Enhancement Units (REUs), were designed to stabilise loose substrata, shelter small fish, and provide new substrata for coral settlement and growth (Figures 6 and 7). The REU (70 cm base diameter and 50 cm height) is sufficiently lightweight and can be deployed by divers in exact locations on shallow reefs.

Figure 5. *A fiberglass reef enhancement unit secured on a reef flat. It is used as a reef restoration tool.*

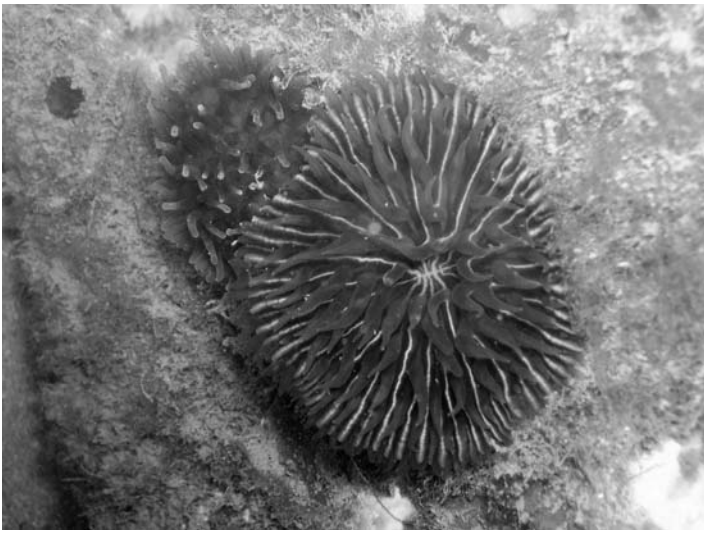

Figure 6. *Naturally recruited mushroom corals on the interior surface of a reef enhancement unit.*

Figure 7. *Transplanted Euphyllia coral fragments growing on a reef enhancement unit.*

3. MANAGEMENT CONSTRAINTS

There is no central agency responsible for the conservation of marine living resources and this weakness exposes marine biodiversity to impacts much more than is necessary. For example, the lack of mitigation against the spread of sediment by some development agencies affects habitats for great distances downstream. The country's 1992 Green Plan recognized the importance of coral reefs and called for close monitoring of future land reclamation projects to prevent excessive silting of the sea. The 2012 Green Plan did not address the sedimentation problem, but highlighted the decision to stop the intended reclamation of Chek Jawa after consultation with nature lovers and experts. On the conservation of nature, one of the targets listed was to keep nature areas for as long as possible.

The Maritime Port Authority manages almost all the marine environment to ensure navigational safety, facilitate marine traffic and prevent oil pollution. The National Environment Authority monitors coastal waters apart from inland to control marine pollution from land-based sources. Other development agencies manage selected offshore islands for industrial or recreation use. Some of the islands are designated for military training purposes. The primary focus of management is pollution prevention and economic development. Protection of natural habitats is not a high national priority and is left much to the individual agencies.

Conservation of natural resources has since its early history focused on terrestrial habitats. Little progress has been made with marine conservation until most recently. None of the coral reefs in the southern islands is under protection. The lack of an integrated coastal management mechanism has resulted in the low priority given to the protection of marine habitats. Project-based environmental impact assessment currently in practice does not take into account cumulative impacts, which lead commonly to further but otherwise avoidable reduction of biodiversity. The establishment of the Biodiversity Centre by National Parks in 2004 and strengthening of its capacity to deal with marine biodiversity issues give some indication that the present limitation may be addressed.

4. IS MARINE BIODIVERSITY COMPATIBLE WITH DEVELOPMENT?

The pace of converting marine habitats to meet development needs appears unsustainable (Hilton and Manning, 1995). Despite the extent of habitat destruction and modification, a high diversity of marine life is still present. This indicates that port waters can support marine species. Maintaining a rich biodiversity in intensively used port waters is important as it helps to regulate environmental quality. A number of factors favour the high biodiversity in Singapore's port waters. One factor is the vigorous water exchange and flushing. Another factor is its location within the Southeast Asian marine biodiversity hotspot. The naturally high biodiversity offers some degree of resilience against impacts leading to the lower than expected level of degradation, but the question remains, for how much longer? Another factor is the strong measures to control marine pollution. Although sedimentation is high, pollution from land and ship-based sources is effectively managed. Commercial fishing is limited to the few remaining fixed palisade traps, which place little pressure on pelagic or demersal fish stocks. These far-from-depleted fisheries stocks attract and maintain dolphin pods and support trophic linkages within the marine environment.

The present approach, where development takes precedence over conservation, has not resulted in drastic depletion of marine biodiversity. Newly-created and modified habitats continue to support life. Species extinction from Singapore waters is not high, as many are redistributed by the changing seascape. Ecosystem processes have not been completely overwhelmed. Seasonal mass spawning of corals, recruitment and growth patterns of other marine species all indicate that biological processes are still intact. It is therefore completely possible for port waters to be teeming with marine life and rich habitats for as long as water quality is maintained. In return, the environmental services provided by a high biodiversity will help to reduce environmental renewal costs. All that is needed is focused attention on the protection of marine living resources to minimise unnecessary damage and loss. Integrated management and strategic impact assessment are approaches relevant to Singapore's situation, with its limited but intensively used marine territory.

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CHAPTER 23

THE PHYSICAL OCEANOGRAPHY OF SINGAPORE COASTAL WATERS AND ITS IMPLICATIONS FOR OIL SPILLS

ENG SOON CHAN, PAVEL TKALICH,
KARINA YEW-HOONG GIN, AND JEFFREY P. OBBARD

1. INTRODUCTION

The island of Singapore (Figure 1) is strategically located at the crossroads of major shipping routes which link the Indian Ocean to the South China Sea. In 2004, the Port of Singapore handled 20.6 million TEUs in Singapore (www.internationalpsa.com) and approximately 140,000 vessels call at the port annually (www.mpa.gov.sg). At any one time, it is estimated that some 800 ships are distributed in Singapore port waters. From an environmental perspective, one of the challenges which port managers face is the increased risk of accidents due to the heavy shipping traffic coupled with a busy port.

In past decades, accidents have occurred occasionally, including the collision between the *Evoikos* and *Orapin Global* in 1997 which spilled 28,500 tonnes of marine fuel oil into the Singapore Strait. Singapore is also one of the largest oil refining centres of the world, with a capacity to process more than one million barrels, or approximately 137,000 tonnes of crude oil daily. The associated chemical and petrochemical industries are crucial sectors of the Singapore economy. Along with the rapid growth of the Asian economy, hence contributing to the increased shipping and port activities, the ability to minimize the risk of accidents and to track and protect the marine environment are just as critical. This would require both the development and implementation of technologies and the enhanced understanding of the physical, chemical and biological processes of the waters around Singapore.

In this chapter, an overview of the physical oceanography of Singapore coastal waters is described. Implications on transport processes with focus on the impacts of oil spills and baseline persistent organic pollutants are discussed.

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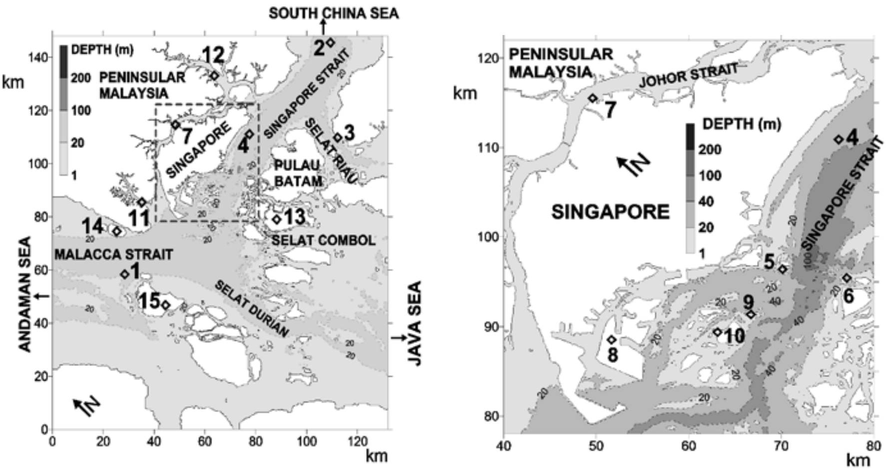


(a) (b)

Figure 1. *Bathymetry (depth in m) of Singapore Strait and ambient seas. Marked locations – 1: Pulau Iyu Kechil, 2: Horsburgh Lighthouse, 3: Tanjung Uban, 4: control point, 5: St. John's Island, 6: Pulau Lengkana, 7: Sembawang, 8: Jurong, 9: Pulau Sebarok, 10: Pulau Semakau, 11: Sungai Pulai, 12: Sungai Johor, 13: Pulau Bulan, 14: Pulau Kukup, 15: Pulau Karimun Besar.*

2. BATHYMETRY AND BOUNDARIES

The currents and circulation in the seas around Singapore are strongly influenced by the bathymetry of the Singapore and Johor Straits and the proximity of the coastal boundaries (see Figure 1). On the northern boundary of the island of Singapore is the Johor Strait, which is divided into two parts by the causeway linking Singapore and Malaysia. Water depths in the Johor Strait range from a few m along the boundaries to about 10-20 m along the center of the straights. The width of the Johor Strait varies from about 0.5 to 2 km including the tidal flats on both sides of the strait. The main rivers flowing into the Johor Strait are Sungai Johor on the east and Sungai Pulai on the west. Both rivers are on the Malaysia side of the Johor Strait.

To the south of Singapore's coast is the Singapore Strait, bounded by Singapore's and Malaysia's southern coasts on one side and the northern coasts of Indonesia's Riau Islands on the other. The Singapore Strait connects the Malacca Strait on the west to the South China Sea on the east. The Malacca and Singapore Straits are connected to the Java Sea through Selat Durian, Selat Combol, Selat Riau and several other minor straits between islands. The narrowest part of the Singapore Strait is about 5 km and between Singapore and Pulau Batam, the range is about 5 km to 15 km. From the southern end of the Malacca Strait to the South China Sea, the water depths are generally less than 50 m. However, in a small area just off the coast to the south-east of St John's Island, water depths reach more than 100 m.

The Singapore Strait is influenced by the exchange of waters between the Indian Ocean, the Malacca Strait and the South China Sea. Water depths range from thousands of meters in the basin east of Vietnam, to hundreds of meters in the shelf, and tens of meters in Singapore and Malacca Straits. In the southern half of the Malacca Straits, the water depths are in the order of a few tens of meters. North of the Malacca Strait, the water depths deepen rapidly towards the Andaman Seas, down to a thousand meters.

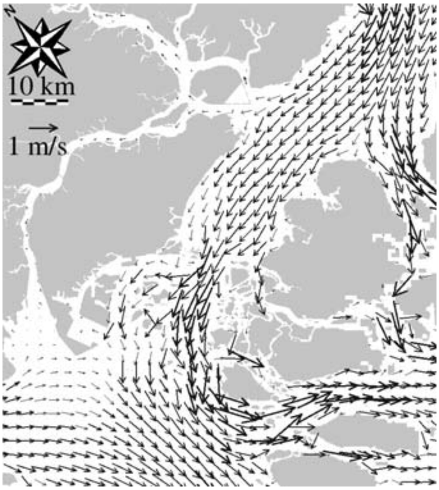
3. DRIVING FORCES

Currents in South East Asian seas, covering the Malacca and Singapore Straits, Java seas and South China Sea are driven by the monsoon winds and tides and also boundary fluxes at the coastal and open boundaries of the domain. Although wind-driven currents are dominant in the open areas, such as the South China Sea, the influence of the wind on the currents in the Singapore and Malacca Straits are less significant compared to the influence of tidal forcing. Most predictions of the hydrodynamics in the Singapore Strait based on tidal forcing at the boundaries compare well with measurements obtained in the Strait. Residual ocean currents associated with the monsoons occur, and they are an order of magnitude smaller than the peak tidal currents. River flows linked to the monsoon storms can be significant in the near field of the rivers. However, their contributions to the currents in the Singapore Strait are relatively insignificant compared to the peak tidal currents in the Strait.

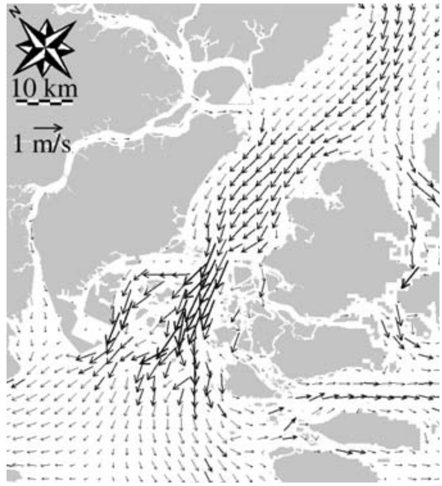
Tides in the both the Malacca and Singapore Straits are semi-diurnal. The overall circulation is driven mainly by the combination of M2, S2, K1, O1, N2 and K2 tides, of which the M2 tide is the most dominant component. Being at the node between the Malacca Strait and the South China Sea, the tidal dynamics in the Singapore Strait is complex. In a joint study on tides and currents in the Malacca and Singapore Straits (JTCS, 1979), it was noted that the dominant M2 tides generated in the South China Sea and the North Indian Ocean interacted in the Singapore Strait. The tidal range varies from about 2.8 m in the west of Singapore to about 1.5 m to the east of Singapore (MPA, 2000).

4. TIDAL CURRENTS

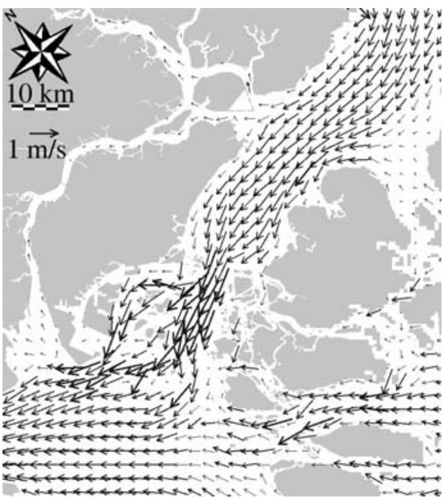
With the advent of computational capabilities in the past two decades, several attempts have been made to model the hydrodynamics in the Singapore Strait and the surrounding waters. Some of the recent efforts include boundary fitted grid models and full three dimensional models (e.g. Shankar et al., 1997; Chen et al., 1997; Zhang and Gin, 2000; Pang and Tkalich, 2003). The predictions typically compare well with the limited measurements available for validations, especially in the comparison of tidal levels. The prediction accuracy of the finer features of the tidal flows is dependent on the numerical model, the baseline information, and the



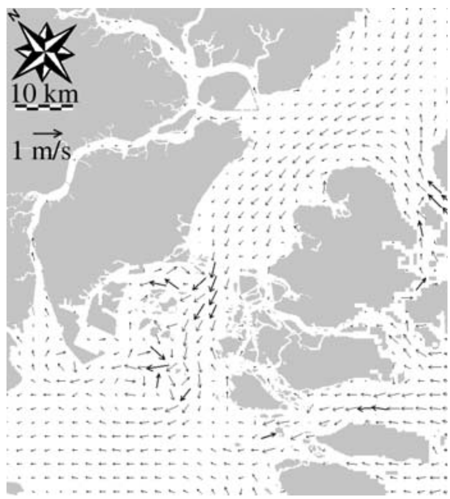
(a) 2.5 hours after the reference time



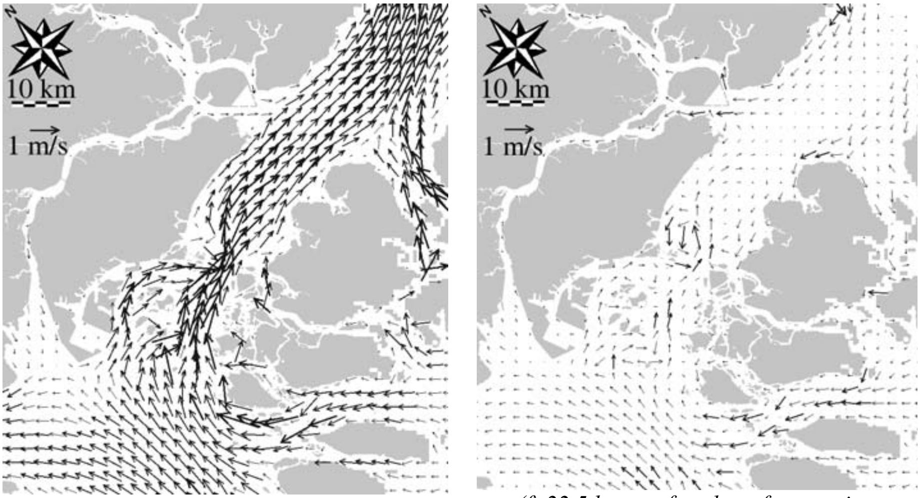
(b) 6.5 hours after the reference time



(c) 10.5 hours after the reference time



(d) 14 hours after the reference time



(e) 19.5 hours after the reference time *(f) 22.5 hours after the reference time*

Figure 2. *Snapshots of typical tidal currents in the Singapore Strait and ambient seas associated with Northeast Monsoon Spring tides (see Figure 3 for time reference).*

prescribed boundary conditions. Typically the boundary conditions are based on tidal elevations at the boundaries of the Singapore Straits, either derived from the results of the joint study of the Malacca and Singapore Straits (JTCS, 1979), or from tidal harmonics predictions Total Tide (2002). Recent modeling efforts are beginning to rely on boundary conditions prescribed further away from the Singapore Strait domain in order to minimize the influence of boundary inaccuracies.

Figures 2a to 2f depict typical snapshots of depth averaged currents associated with the Northeast Monsoon spring tidal cycles. These snapshots were derived using a semi-implicit sigma-coordinate hydrodynamic model developed at the Tropical Marine Science Institute, National University of Singapore (Pang and Tkalich, 2004).

For these simulations, tidal elevations were prescribed at the boundaries of the computational domain far away from the boundaries of the Singapore Strait. Tidal elevations at Horsburgh Lighthouse, Tanjung Uban and a control point within the Singapore Strait (Figure 1) are shown in Figure 3.

The computation was conducted in 2-D mode with a horizontal grid size of 100 by 100 m. The model was validated with available measurements at selected locations. Figure 4 shows the comparison of predicted and measured currents at the control point off the east coast of Singapore in the Singapore Strait (Figure 1).

Currents, in general, are typically less than 2 m s^{-1} in most parts of the Singapore Strait except in the narrow channel between St John's Island and Pulau Lengkana (Figure 1). Easterly flows in the Singapore Strait are usually associated with a rising tide and remain eastward flowing when the tidal elevation in the strait dips slightly before rising again (compare Figures 2a-f with Figure 3). On the other hand,

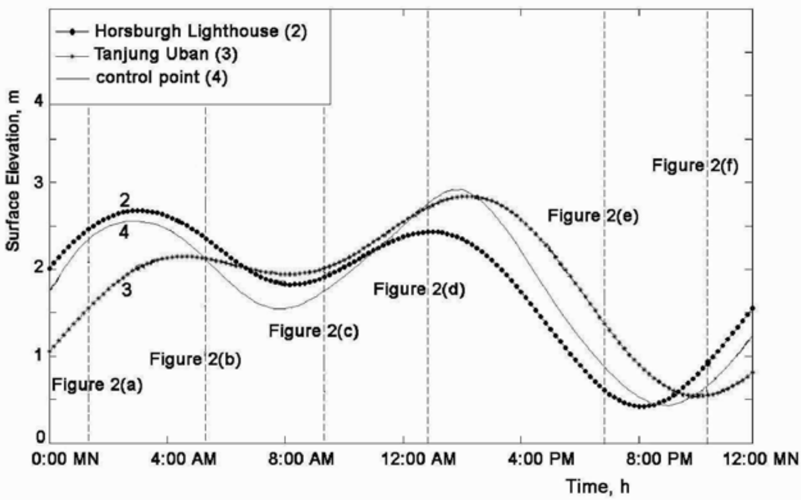


Figure 3. *Tidal elevations at Horsburgh Lighthouse, Tanjung Uban and a control point within the Singapore Strait (see Figure 1 for locations).*

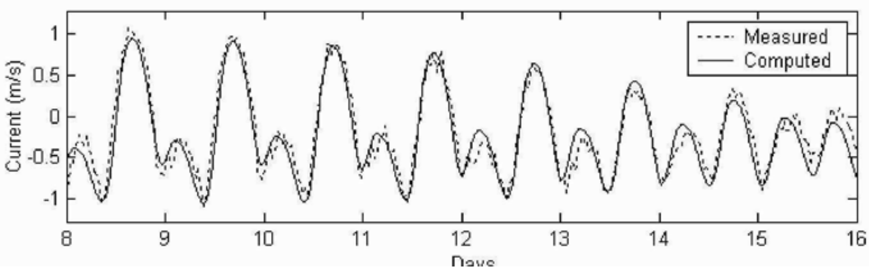


Figure 4. *Comparison of computed and measured currents at the control point in Singapore Strait (see Figure 1 for the location).*

westward flows in the Singapore Strait are associated with decreasing tides. When currents are streaming from the Horsburgh Lighthouse (Figure 1) towards the west (Figures 2a-d), the tidal streams first bend towards the south through the straits connecting the Malacca and Singapore Straits to the Java Sea (Figure 1). After the first peak flow, the currents will weaken (Figure 2b), sometimes even flowing weakly towards the east (MPA, 2003), after which the westerly flow will strengthen again but with the flows on the west heading northwest through the Malacca Strait (Figures 2c and d). During this time, the currents through Selat Durian, Selat Combol and Selat Riau also head northward to join the westward going flows in the Singapore Strait. The westward flows in the Singapore Strait will subsequently weaken and turn eastward while the flows in the Malacca Strait, Selat Durian, Selat Combol and Selat Riau Straits are still northward. During the eastward flow, a prominent eddy is also evident on the north eastern side of St John's Island. At the next turn of the flow direction in the Singapore Strait, the flows in the other straits also weaken and reverse in direction, leading back to the scenario shown in Figure 2a.

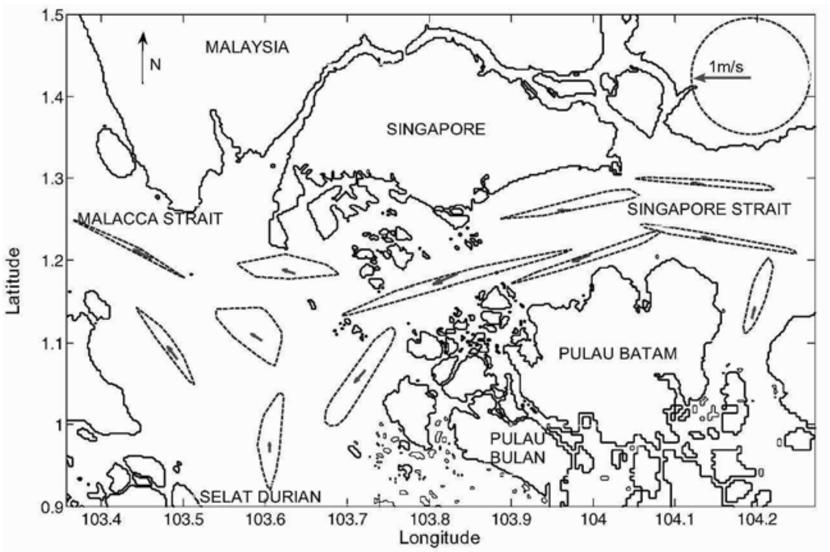


Figure 5. *Envelopes of velocity vectors over a typical 15 days Northeast monsoon period. The arrows represent the net currents.*

The above snapshots are typical of the spring tide cycles and demonstrate the complex interactions of tides in the Singapore Strait and ambient seas. While the flow directions associated with maximum flows are streamlined, those associated with the weaker flows are more complex. To illustrate the flow characteristics in the Singapore Strait, envelopes of hourly velocity vectors over a 15 days Northeast Monsoon period were obtained at selected locations and presented in Figure 5. Also included are the vectors representing the mean speed and direction over the same period.

A flatter envelope suggests a flow that switches between two dominant directions like oscillatory flows in a narrow wave flume. A more rounded envelope suggests the rotation of flow directions during the tidal excursion. The flatter envelope between the east coast of Singapore and Pulau Batam is consistent with the predominant east-west flows in this domain. A similar bi-directional scenario is evident at locations in the southern end of the Malacca Straits between Pulau Kukup (Malaysia) and Pulau Karimun Besar (Sumatra). The domain between the south west coast of Singapore, Pulau Karimun Besar and Pulau Bulan (west of Pualu Batam) yields a much more complex flow scenario due to the change in flow directions in the Malacca Strait, Selat Durian and the Singapore Strait.

The net currents during the Northeast monsoon are westward in the Singapore Strait and northwards in the Malacca Strait (Figure 5). These net directions are reversed during the Southwest monsoon.

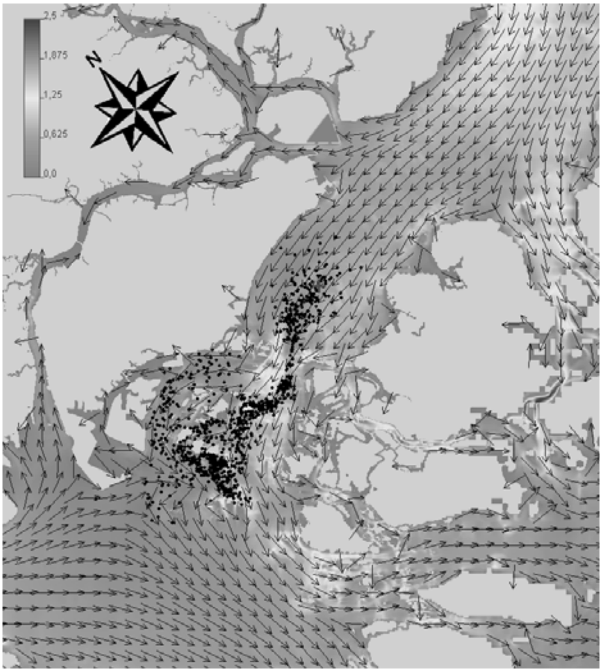
5. IMPLICATIONS ON TRANSPORT PROCESSES

In the Singapore Strait and ambient seas, particulates and dissolved matter in the water column are advected and dispersed mainly by the tidal currents. Occasional strong winds and waves would enhance the mixing in the upper layers. In terms of the suspended solids, the water column is not always well mixed. Vertical profiles of physical and biological parameters could be well structured depending on the tidal conditions, the river discharges, and also the surface winds and waves.

Considering the typical tidal flow characteristics (Figures 2a-f and Figure 5), some basic features of the transport processes could be identified. Given the dominant east-west streaming in the eastern part of the Singapore Strait, any matter discharged into the eastern Singapore Strait can be expected to oscillate east and west according to the tidal advection, with spreading mainly in the same directions.

Once the advected volume moves past the constriction of St John's Island towards the west, the advection could head southward or northward. When the tidal flow reverses, some of the transported matter would be transported back eastward following the tidal streams. During the Northeast Monsoon period, the mean drift is towards the Malacca Strait (Figure 5). During the Southwest monsoon, the mean drift is towards the South China Sea. Animation 1 depicts the transport of waterborne particles discharged at a location off the east coast of Singapore. The process of dispersion, oscillatory advection and mean drifts is evident in the animation. It should be noted that the dispersion of the particles also means a reduction in the concentration and hence a dilution of the source concentration. The streaming of the tidal flows (Figures 2a-f) also suggest that discharges at the southern end of Malacca Strait, Selat Riau, Selat Combol or Selat Durian may enter the Singapore Strait domain depending on the monsoon period and the phasing of the discharge relative to the tidal phases.

The hydrodynamic processes discussed above are important in determining the mixing and transport of anthropogenic inputs. The impact of these inputs on the environment, however, is also dependent on the coupling of these processes with the biological and chemical kinetics, especially in the cases of oil pollution and ballast water discharges.



Animation 1. *Depth-averaged velocity field and transport of matter originally discharged in Singapore Strait during the Northeast monsoon spring tide. 1000 particles were introduced at the source point. The subsequent particle locations were derived using a particle tracking model with dispersion physics incorporated. The animation depicts the transport process up to 80 hrs after release.*

6. CASE EXAMPLE: OIL SPILL

Oil spilled in the marine environment can pose a significant threat to marine life. The water-soluble components of crude oils and refined products include a variety of compounds that are toxic to a wide spectrum of marine plants and animals. Aromatic compounds are generally more toxic than aliphatics, and middle molecular weight constituents are generally more toxic than high molecular weight tars (Doerffer, 1992). A spillage of diesel fuel with a high aromatic content, is therefore much more damaging than bunker fuel oil and weathered oil, which generally have lower aromatic contents. The ecological effects of oil pollution depend on the type and amount of oil, the frequency of exposure, light and heavy oil fractions, environmental conditions (water temperature, salinity, nutrient concentrations, tide movements, wind, currents), the use of chemical dispersants and their associated toxicity and the sensitivity of specific local biological communities to the toxic effects of hydrocarbons (Price et al., 1999). Some petroleum hydrocarbons (e.g.

polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs)) have the capacity to concentrate in the tissues of marine organisms, with concentration increasing up the food chain (Baker, 1983). The result is that toxic levels could be reached for organisms at the end of the food chain, including birds and humans.

The collision between two large oil tankers, the *Evoikos* and *Orapin Global*, on 15 October, 1997, was the catalyst for recent oil spill research in Singapore, ranging from bioremediation studies, oil spill model development to environmental impact assessment. To date, this is the largest oil spill in Singapore (MPA Annual Report, 1997), with the amount discharged (28,500 tonnes) accounting for more than one third of the total oil lost for the total oil lost at sea for the world in 1997 (ITOPF, 1998). The collision occurred at night off Pulau Sebarok (Figure 1) in the Singapore Strait during spring tides. Within a few days after the collision, the oil slick had landed on the intertidal coastlines of several southern islands in the Singapore Strait. Some of the immediate questions following the spill included 1) what were the short and long-term impacts on marine life? and 2) how long would the effects of the spill remain in the marine environment?

In terms of assessing impacts on marine organisms, an understanding of baseline conditions prior to the spill was required, which, unfortunately, was unavailable at the time. Nevertheless, field monitoring studies were carried out to provide a partial biological assessment of the aftermath of the *Evoikos* oil spill and clean-up efforts conducted subsequent to the spillage (Tan et al., 1999). In general, the oil spill did not cause a major ecological disaster involving seawall molluscan communities and corals, the major organisms monitored in the study. However, both species abundance and species diversity of molluscs fluctuated considerably at both impacted and control sites. Thus, this factor needs to be taken into consideration in future studies so that a more robust dataset can be established. In addition, it is clear that biological baseline data is necessary so that impacts from future oil spills can be properly evaluated.

6.1. An oil spill-food chain model

While direct measurements of marine organisms can be used for impact assessment of oil spills, there is often a need to go beyond current situations, and to predict future scenarios. This is where models can be used to assist port managers in dealing with accidental oil spills. For example, one important application of oil spill modeling is to predict where the oil slick is likely to travel, whether it will encounter sensitive marine areas, and what time frame is involved? This will then allow port managers to allocate emergency clean-up measures to the right places and hopefully, avoid excessive ecological damage.

An oil spill-food chain interaction model for coastal waters was developed to assess the probable impacts of oil spills on several key marine organisms (Gin et al., 2001). The model consists of two parts: i) a multiphase oil spill model (MOSM) which combines sub-modules for oil slick dynamics at the water surface, oil sedimentation near the seabed and transport of the oil phases in the water column (Tkalich et al., 2003) and ii) a pelagic food chain model (Chapra, 1997) which considers the interactions between oil uptake and phytoplankton, zooplankton, small fish, large fish and benthic invertebrates. MOSM is able to compute the concentrations of specific organic compounds in their dissolved and particulate phases both in the water column and benthic layer. It provides the mass exchange between the media and oil phases, as well as losses due to evaporation, hydrolysis, photolysis, oxidation and biodegradation using a kinetic approach. Transport of the oil components and phases in the water column are simulated using the velocities and diffusivity pre-computed with a 3-dimensional hydrodynamic model.

In the food-chain model, phytoplankton, zooplankton, small fish and large fish are assumed to uptake dissolved oil from all layers of the water column, while benthic invertebrates uptake dissolved oil from the layer of water adjacent to the bed sediments as well as from within the bed sediments (Huda, 1999; Gin et al., 2001). Zooplankton and benthic invertebrates are the predators of phytoplankton; small fish are the predators of both zooplankton and benthic invertebrates; and large fish are the predators of small fish. The uptake rates for the model were estimated using the mathematical formulation suggested by Connolly (1991) while the toxicant transfer efficiencies across the organism membrane and assimilation efficiencies, which are related to the octanol-water partition coefficient (K_{ow}), were estimated from Thomann and Muller (1987). In the specific case of the *Evoikos-Orapin Global* oil spill, anthracene was chosen as the target compound of interest as it formed the major component for this particular spill. Using a hypothetical accidental spill of 28,000 tonnes, the fate and transport of the spilled was computed for typical hydrodynamic scenarios (e.g. Figure 6).

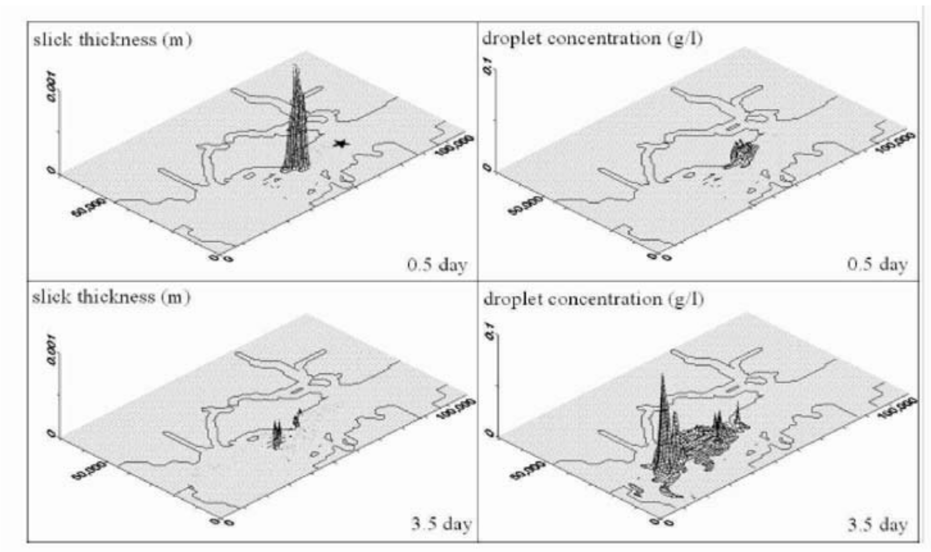


Figure 6. *Surface slick thickness and oil droplet concentrations at 3 m depth.*

The results of the simulation also showed that the concentrations of anthracene in the phytoplankton, zooplankton, small fish, large fish and benthic invertebrates one day after the spill were below the lethal toxicity levels (LC_{50}). Field measurements of polycyclic aromatic hydrocarbons (PAHs) in benthic invertebrates (i.e. the gastropod mollusc, *S. guamensis*) at different places in the Singapore Strait after the *Evoikos-Orapin Global* oil spill ranged between 9.5×10^{-5} to 1.0×10^{-3} g kg^{-1}. The results simulated from the oil spill-food chain model gave corresponding anthracene concentrations ranging from 1×10^{-4} to 5×10^{-3} g kg^{-1}, which is within the range observed for the field measurements. The model was not verified for other marine organisms due to the lack of field data.

6.2. Bioremediation studies

In the longer term, oil spilled in the marine environment will most likely undergo biodegradation by microorganisms. In the specific case of the *Evoikos* oil spill, most of the accumulated oil on the beaches and breakwaters were cleaned-up manually, with the aid of chemical dispersants. However, for the more inaccessible beaches and cleaned beaches with residual oil, biodegradation is most likely the main mechanism for oil removal. In these cases, the rate at which biodegradation of oil-based compounds can proceed is often limited by the lack of nutrients, particularly nitrogen and phosphorus. Numerous field and laboratory tests worldwide have shown that overcoming these limitations results in the successful enhancement of oil biodegradation (Bragg et al., 1994). Hence, a bioremediation study was conducted to investigate whether the indigenous microbial populations could be stimulated by the addition of nutrients (Mathew et al., 1999).

Two control plots and two treated plots measuring 1 m by 0.5 m were established on the isolated beaches of Pulau Semakau, an island 15 km south of Singapore (Figure 1). The four plots were constructed using corrugated steel sheets fixed to a depth of 0.25 m in the sand and to a height of 0.5 m and spaced 5 m apart. All four plots were located so that they would be subject to diurnal tidal flooding. 100 kg of beach sediment contaminated with oil from the *Evoikos-Orapin Global* oil spill was added to each plot. A mixture of 0.4 kg ammonium nitrate (NH_4NO_3) and 0.5 kg potassium hydrogen phosphate (K_2HPO_4) was dissolved in seawater and added to two of the experimental plots, while the remaining two plots were left untreated to act as controls. Sampling (from all four plots) and nutrient addition to the two treated plots were carried out five times over a period of 50 days. However, storm damage to the plots prevented extended sampling. The field samples were analyzed for (a) biological activity using the dehydrogenase enzyme assay; (b) total heterotrophic plate counts and (c) total petroleum hydrocarbons (TRPH) using standard methods.

The results clearly showed that the plots augmented with nutrients had significant reductions in TRPH (using the two sample *T*-test as a confidence interval of 95%) compared to the controls, especially towards the end of the experimental period (Table 1). This was supported by the enhanced activity of indigenous microbes (using the dehydrogenase enzyme assay) (Table 2) as well as increase in heterotrophic bacteria concentrations measured for the treated plots. Thus, the addition of nitrogen and phosphorus had a clear stimulatory effect on indigenous microbial populations. These microbes had an innate ability to degrade hydrocarbons under suitable environmental conditions, presumably as a result of adaptation/ acclimatization to intermittent exposure of petroleum hydrocarbons from oil spillages in the Singapore Strait over time. This study has demonstrated that nutrient addition is able to significantly accelerate the natural degradation process. Given the warm temperatures in Singapore, biodegradation rates and hence, oil contaminant removal, can be expected to be high as long as nutrient augmentation is carried out.

Table 1. *Average (n=3) Total Recoverable Petroleum Hydrocarbons (TRPH) data from reference (A&C) and nutrient amended (B&D) plots (expressed as percentages of dry weight of the sediment).*

Day	*Reference Plot A*	*Amended Plot B*	*Reference Plot C*	*Amended Plot D*
0	6.72	6.98	6.97	6.97
7	6.38	6.59	6.59	6.72
15	6.21	6.55	6.54	5.33
25	6.40	6.29	6.28	4.52
49	6.77	6.40	6.40	3.87

Table 2. *Average (n=3) dehydrogenase activity data from the reference (A&C) and nutrient amended (B&D) plots (expressed as μg INTF (iodonitrotetrazolium formazan) formed / g dry weight of the sediment h^{-1}).*

Day	*Reference Plot A*	*Amended Plot B*	*Reference Plot C*	*Amended Plot D*
0	11.47	11.46	12.02	10.26
7	5.19	11.94	5.47	11.97
15	4.08	13.05	4.00	8.99
25	4.72	11.98	3.21	7.56
49	6.34	12.28	6.00	7.66

6.3. Persistent organic pollutants in Singapore's marine environment

While most organic compounds are largely biodegradable, there are some chemicals which remain highly recalcitrant to chemical and biological degradation, and therefore persist in the environment for long periods. These Persistent Organic Pollutants (POPs) include the organochlorine pesticides (OCPs), polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs). POPs are known to adversely affect the endocrine system in both wild fauna and humans, have a propensity to bioaccumulate in the lipid fraction of biological tissues and are subject to biomagnification in both terrestrial and aquatic food webs. Humans may be chronically exposed to environmental (POPs) via the ingestion and inhalation pathways (Duarte-Davidson and Jones, 1994), and many such compounds have been detected in a range of human tissues including serum, breast milk and adipose tissue (Dewailly et al., 1993). In 2001, many countries signed the Stockholm Convention under the United Nations Environment Programme to implement measures to reduce and eliminate the release of POPs into the environment, including bans on production, import, export, and use of certain POPs (UNEP, 2001). For Singapore, with the combination of a heavy shipping traffic and numerous ship building yards, petroleum refineries, and pharmaceutical manufacturing plants located along the coastline, it is important to track and prevent both operational and fugitive discharges of POPs. Data on the prevalence of POPs in Singapore's coastal waters are needed to assess the potential threats of POPs to the marine ecosystem and human health. Since 2001, a study has been initiated to capture data on the prevalence of POPS in Singapore's coastal marine environment.

Marine water samples within 1 km of the coastline of Singapore were analysed to determine prevalent concentrations of a range of POPs by Basheer et al. (2003). Samples were collected from 0.5 m and mid-depth at twenty-two locations. The POPs analysed were classed as USEPA priority pollutants, and included: sixteen polycyclic aromatic hydrocarbons (PAHs); eight polychlorinated biphenyls (PCBs); and twelve organochlorine pesticides (OCPs) (Table 3).

Table 3*. Types of Persistent Organic Pollutants (POPs) analysed in the survey of northeastern and southwestern sectors of Singapore coastal waters.*

Polycyclic Aromatic Hydrocarbons (PAHs)	*Organochlorine Pesticides (OCPs)*	*Polychlorinated Biphenyls (PCBs)*
Naphthalene	α-BHC	2-chlorobiphenyl
Acenaphthylene	Lindane	2,3-dichlorobiphenyl
Acenaphthene	β-BHC	2,4,5-trichlorobiphenyl
Fluorene	Heptachlor	2,2',4,4'-tetrachlorobiphenyl
Phenanthrene	Aldrin	2,2',3',4,6-pentachlorbiphenyl
Anthracene	Dieldrin	2,2',4,4',5,6'-hexachlorobiphenyl
Fluoranthene	Endrin	2,2',3,3',4,4',6-heptachlorobiphenyl
Pyrene	Endosulfan II	2,2',3,3',4,5',6,6'-octachlorobiphenyl
Benz[a] anthracene	p,p'-DDD	
Chrysene	p,p'-DDT	
Benzo[a] fluoranthene	Endrin aldehyde	
Benzo[k] fluoranthene	Methoxychlor	
Benzo[a]pyrene		
Indeno[1,2,3-cd]pyrene		
Dibenz[a,h] anthacene		
Benzo[ghi]perylene		

6.3.1 Polycyclic aromatic hydrocarbons

All sixteen PAHs were detected in all water samples from both depths at every sample location. Total PAH concentrations in seawater ranged from 93.0 to 1419.6 ng l^{-1} and from 88.4 to 1472.8 ng l^{-1} in the northeastern and southwestern region, respectively. The overall mean total PAH concentrations for seawater depth levels were as follows: surface, 235.1±46.2 ng l^{-1}; and mid-depth, 343.1 ±61.5 ng l^{-1}. The highest total PAH concentrations measured were obtained at locations off Sembawang (in the northeastern sector) and off Jurong (Figure 1) (in the southwestern sector), which are both in the vicinity of shipyards and industrialized coastal areas. The lowest concentrations of total PAHs for these regions were obtained at locations remote from industrial areas and where the water column was well mixed by strong currents. Among the sixteen PAHs measured, the highest individual PAH concentrations measured were for six ring indeno[1,2,3-cd]pyrene i.e. 712.9 ng l^{-1} and 218.8 ng l^{-1} at the northeastern and southwestern regions, respectively. The lowest concentrations of acenapthylene detected were 1.3 ng l^{-1} and 1.9 ng l^{-1}, respectively. This distribution profile may reflect the different properties of low and high molecular weight PAHs, where low molecular weight compounds have higher vapour pressure and water solubility, and are therefore more readily volatilized and degraded by microbial activity. In contrast, higher molecular weight PAHs are more likely to be associated with the particulate phase within the water column and undergo sedimentation, thereby accounting for their higher concentration at mid-depth. Similar vertical distributions have been previously noted in a study of Baltic coastal waters by Broman et al. (1991). At mid-depth, PAHs were dominated by indeno[1,2,3-cd]pyrene, but other abundant compounds included dibenzo[ah]anthracene, benzo[ghi]perylene and anthracene.

Overall, higher molecular PAH compounds were more prevalent in Singapore coastal waters than lower molecular weight compounds. The prevailing ocean currents in the region most likely govern the fate of PAHs, and the presence of localized high levels of PAHs may be a function of petroleum discharges from shipping. The highest total PAH concentration detected in Singapore's coastal waters (i.e. 1472.8 ng l^{-1}) is greater than that reported for Xiamen Harbour, China i.e. up to 945 ng l^{-1} (Zhou et al., 2000); the German-Baltic sea i.e. 6.7 ng l^{1} (Witt, 1995); the Coral Sea, Australia i.e. 240 ng l^{-1} (Smith et al., 1987); Chesapeake Bay, USA i.e. 14.05 ng l^{-1} (Ko and Baker, 1995); the Northwestern Black Sea, Ukraine i.e. 0.7 ng l^{-1} (Maldonado et al., 1999); and Admiralty Bay, Antarctica i.e. 80 ng l^{-1} (Bícego et al., 1996), but lower than concentrations reported for Rhode Island, USA i.e. 115,000 ng l^{-1} during an oil spill event (Reddy and Quinn, 1999).

6.3.2. Organochlorine pesticides (OCPs)

In Singapore, extensive agricultural activities have been phased out for more than two decades. Although some minimal agricultural activities remain, they do not generally involve extensive use of the types of pesticides discussed in the present work. However they are used in neighbouring countries. Nevertheless, OCPs were detected in samples taken at both depths from all locations from both the northeastern and southwestern region.

Total OCP concentrations ranged from 4.0 to 22.0 ng l^{-1} and 3.0 to 21.9 ng l^{-1} at the northeastern and southwestern regions, respectively. Overall, higher concentrations were detected in the northeastern region, which is most likely due to the confined waters in the northeast, which limits the hydrodynamic dispersion of contaminants. This river runs across agricultural, commercial and industrial land in Malaysia and into the Straits of Johor, adjacent to Singapore. BHC and Dieldrin were the most abundant pesticides present and their levels ranged from 0.13 to 17.87 ng l^{-1}, and 0.34 to 6.91 ng l^{-1} and in northeastern and southwestern regions respectively. In both regions, the highest concentration Lindane, 0.34 ng l^{-1}; Endrin, 1.97 ng l^{-1}; p,p'-DDT, 1.14 ng l^{-1}; and p,p'-DDD, 1.17 ng l^{-1} at northeastern and southwestern locations respectively.

OCPs are, to a variable extent, insoluble in seawater (< 1 ppb), but are readily soluble in fat and adsorb strongly onto suspended particulates in the water column. The surface layer of the sea comprises a film of about 1 mm of thickness, which is known to contain fatty acids. Due to the lipophilic and persistent nature of OCP, accumulation in this surface layer is known to occur (Zhou and Rowland, 1997). OCP enrichment of the surface film may be of considerable importance to surface living organisms or to birds that skim food off the sea surface. Surface plankton and other organic particulates are readily associated with OCPs and undergo subsequent sedimentation. In general, higher amounts of OCPs were detected at mid-depth locations close to industries and shipping anchorages, where hydrodynamic dispersion is confined.

The land area under agriculture use in Singapore is negligible and there is no direct application of organochlorine pesticides in the country. However, pesticides may be easily transported through the ambient environment by different mechanisms including volatilization from soil and spray drift during application to crops (Dörfler and Scheunert, 1997). The presence of OCPs in Singapore's marine waters is probably a function of their use in neighbouring countries.

Concentrations of OCPs measured in Singapore seawater are comparatively lower than those detected in water from the Selangor River in Malaysia; i.e. Aldrin, up to 884.00 ng l^{-1}; Dieldrin, up to 850.00 ng l^{-1}; Endrin, up to 10970.0 ng l^{-1}; α-Endosulfan, up to 8.90; β-Endosulfan, up to 12270 ng l^{-1}; Heptachlor, up to 13710.00 ng l^{-1}; Lindane, up to 40950.0 ng l^{-1}; p,p'-DDT, up to 44770.00 ng l^{-1}; p,p'-DDE up to 2310.00 ng l^{-1} (Mustafa et al., 2000); as well as the Surabaya river, Indonesia p,p'-DDT up to 49.63 ng l^{-1} (Dewi, 2000); Philippine coastal waters, α-BHC up to 21 ng l^{-1} and aldrin at 7 ng l^{-1} of (Santiago, 2000); the Dampha and Balat estuaries in Vietnam, DDT i.e. 30.00 ng l^{-1} (Viet et al., 2000); Bohai Sea, China, DDE, DDD and DDT up to 50 ng l^{-1} (Yeru and Hao, 2000). However, OCP levels in Singapore's coastal waters are higher than those found in the Coral Sea, Australia where total OCP concentrations have been measured at 1.21 ng l^{-1} (Tanabe et al., 1984), and 5.5 ng l^{-1} (Kurtz and Atlas, 1990).

6.3.3. Polychlorinated biphenyls (PCBs)

PCBs represent a group of compounds that have been widely used in a range of industrial applications. All eight PCBs analysed in the study were detected in both the northeastern and southwestern sectors for the majority of sample stations.

Total PCB concentrations in seawater from both regions varied from 0.04 to 61.7 ng l^{-1} and 0.22 to 20.1 ng l^{-1} in northeastern and southwestern regions, respectively. The highest measured concentration of an individual PCB congener i.e. 2,2,'3,3',4,5',6,6'-octachlorobiphenyl was 40.71 ng l^{-1} and 15.42 ng l^{-1} at the northeastern and southwestern regions, respectively.

Long-range atmospheric transport is likely to be a source of PCBs detected in remote waters and results in low-level concentrations in nearly all environmental matrices (Bidleman et al., 1989). However, higher levels can be associated with proximity to industry, as well as waste discharges from shipbuilding yards and municipal sewage plants located in coastal regions. PCBs are hydrophobic compounds with an octanol-water partition coefficient (K_{ow}) ranging from 4.5 to 8.2. The aqueous solubility is less than 5 mg l^{-1} for the more chlorinated congeners (i.e. >2 chloro group) (Patil, 1991). The distribution of PCBs in coastal waters contrasts with that of PAHs, where the surface was more contaminated than at mid-depth for the majority of sample locations. The measurement of PCBs in seawater samples shows that the northeastern coastal region of Singapore has much higher concentrations than the southwestern region. These variations are most likely due to historic and episodic inputs from industrial sources as well as hydrodynamic factors.

The maximum level of PCB contamination detected in Singapore coastal waters (i.e. 61.7 ng l^{-1}) is lower than that recorded from Xiamen, China and Victoria Harbour, Hong Kong i.e. 151 ng l^{-1} (Hong et al., 1995); Jamaica-Kingston Harbour, i.e.3,500 ng l^{-1} (Mansing et al., 1995); Doñana National Park, Spain i.e. 237 ng l^{-1} (Fernàndez et al., 1992); and higher than levels measured in the Gulf of Mexico and Atlantic Ocean (North) i.e. <0.003 ng l^{-1}, (Sauer et al., 1989), the Northern Pacific Ocean i.e. 0.59 ng l^{-1} (Tanabe et al., 1984) and the Dutch Wadden Sea i.e. 0.62 ng l^{-1} (Duinker and Hillerbrand, 1983).

7. CONCLUSIONS

The physical oceanography of the Singapore Strait and ambient seas is governed mainly by the tides together with a seasonal net circulation. The interaction of tidal streams from the Malacca Strait, South China Sea and straits linking the Malacca and Singapore Straits to the Java Sea makes the overall flow in the Singapore Strait rather complex. Being a node at the confluence of these interacting water bodies, any discharges in the domain, such as oil spills, would affect the surrounding seas, although the concentration would be diluted through mixing and transport. In addition to physical processes, the ultimate fate of a pollution event would depend on the coupled interactions with biology and chemistry of the water body, including food-chain interactions and biodegradation. For the more persistent organics in the marine environment, baseline studies showed that concentrations of PAHs measured in Singapore's coastal waters were generally higher than levels reported elsewhere, whereas OCPs and PCBS were generally lower than reported levels for other Asian countries, but higher than some levels reported elsewhere in the world. Overall, the prevalence of POPs in Singapore's coastal waters suggest the need for continued monitoring and evaluation of their transport and biological impact in the marine environment.

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CHAPTER 24

MANAGING THE PORT OF JAKARTA BAY: OVERCOMING THE LEGACY OF 400 YEARS OF ADHOC DEVELOPMENT

DIETRIECH G. BENGEN, MAURICE KNIGHT, AND IAN DUTTON

1. INTRODUCTION

In the first comprehensive review of Indonesia's marine resources, Tomascik et al. (1997) noted that… "One of the many challenges facing Indonesia today is the reconciliation of development objectives and conservation aims in the marine and coastal sector". Nowhere are these challenges more evident than in Jakarta Bay. Since the decision by the Dutch East India Company to relocate the capital to Batavia (now Jakarta[1]) in the 1620s, Jakarta Bay (Figure 1) has served as the principal gateway to Indonesia and the lifeline for development of the Jakarta region. It continues play a pre-eminent role in the development western Java.

Over the subsequent 400 years of colonial rule and the nearly 60 years since Indonesia's independence, the bay and watershed have served as the hub for rapid national development. Jakarta is now the primary economic centre in Indonesia. Some 11.5 million people live in greater Jakarta, and an estimated 10 million more live in the exurban and rural watersheds that drain to the bay.

On a global scale, the bay is not a particularly notable harbour due to its shallow depth, lack of clear navigation channels and exposure to prevailing winds during the monsoon season. However, the combination of welcoming local leaders, adequate nearby water supplies and undeveloped land, strategic trade location at the centre of the Indonesian archipelago and ready military defensibility made Jakarta Bay an ideal focus for colonial rule. Jakarta Bay quickly rose to international prominence as a leading harbour in Southeast Asia. This strategic significance was reinforced during the rapid economic growth experienced during the l980s. Double-digit growth in GDP during that decade led to the rapid expansion of port facilities in the

[1] Also known as Djakarta

E. Wolanski (ed.), The Environment in Asia Pacific Harbours, 413–431

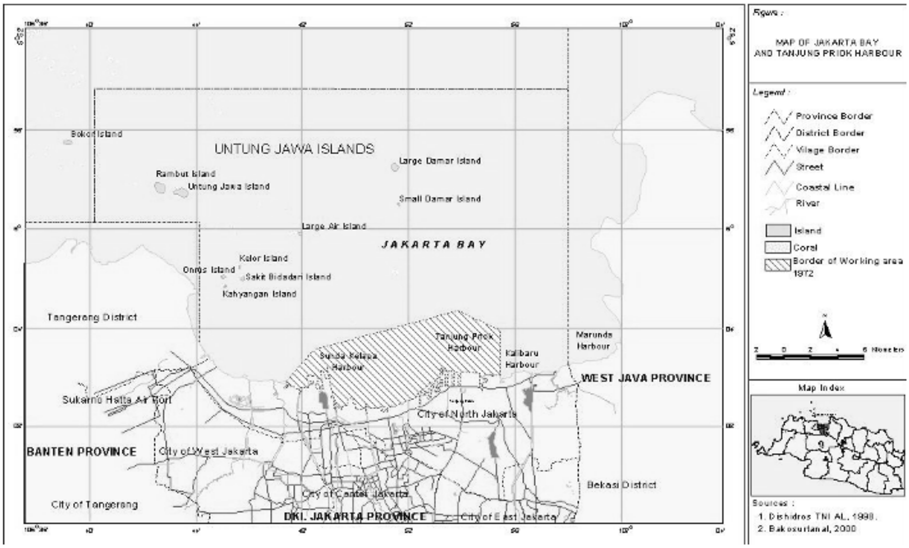

Figure 1. *Map of Jakarta Bay, DKI Jakarta Indonesia.*

suburb area of Tanjung Priok and the reclamation of large areas of the bay foreshore and wetlands to accommodate industrial and urban development.

This development occurred at significant ecological and social cost. Jakarta Bay is perhaps the most polluted harbour in Asia and has very little intact fisheries. What fisheries still exist have a non-original structure and are comprised of more opportunistic species that can exist in the now heavily polluted waters. The combination of inappropriate past development and continuing lack of coordinated bay governance has created a legacy of severe environmental damage – several of the former “thousand islands” that comprised a small archipelago of coral cays in the outer arc of the bay have disappeared due to sand mining and those that remain are among the most threatened coral reefs in Asia.

This chapter traces the development of the bay over the past 50 years and how, through a combination of neglect and ignorance, Indonesian society now faces major challenges to sustain the economic, social and ecological values of the bay. There are growing calls to reduce pollution, protect traditional fishing communities, restore coral reefs and mangrove systems and optimize bay use for industrial, urban, tourism and conservation uses. However, efforts to address bay management on a more integrated and long-term basis are only now emerging and will require a far greater level of political commitment if current degrading influences are to be reversed and an effective and comprehensive management regime introduced.

2. NATIONAL SETTING

2.1. Significance of coastal and marine resources

Coastal and marine resources obviously are of paramount importance in an archipelagic nation like Indonesia where more than 75% of the national area is sea and the 24% that is land is fragmented amongst more than 17,000 islands. The 81,000 kilometre shoreline is the world's second longest after Canada (Tomascik et al., 1997), but is the world's most usable in terms of overall accessibility. The Indonesian archipelago contains some of the world's largest remaining mangrove forests and has the largest area of coral reefs of any country (Hopley and Suharsono, 2000). Indonesia's waters are among the most productive of all tropical seas. They cover the epicenter of global marine biodiversity and provide globally significant corridors for migratory species (Rais et al., 1998; Kahn, 2003).

Coastal and marine industries such as oil and gas production, transportation, fisheries and tourism account for a quarter of Gross Domestic Product and employ more than 15% of Indonesia's workforce. Some 140 million Indonesians live within 60 kilometres of the coast; many of these within the large coastal cities that occupy a predominant position in the national economy (Dahuri and Dutton, 2000).

2.2. Development experience from 1969-1998

During the first 25 year development plan (PJP I – 1969-1993) and related five year development plans (Repelita), national planning policy placed considerable emphasis on terrestrial development, particularly in Java and Sumatra (Sloan and Sughandy, 1994; MOSE, 1996). The first marine protected area (Kepulauan Seribu in Jakarta Bay) was only formally gazetted in 1982 and it was not until the late 1980s that strategic attention was given to management at broader scales (Alder, et al. 1994; Sloan and Sughandy, 1994).

In the initial phase of PJP II (1993-1998), four goals for coastal and marine resources development were established:

- support expanded coastal and marine enterprises throughout Indonesia, especially in Eastern regions
- support offshore industries, especially oil and gas production
- strengthen national sovereignty and jurisdiction by mapping of continental shelves and the EEZ
- establish a coastal and marine geographic information network.

However, achievement of these objectives was limited by the lack of integrated management of coastal and marine areas. Under the "Wawasan Nusantara" (or archipelagic outlook) concept that Indonesia successfully championed in the formulation of the international Law of the Sea, decision-makers argued that all marine areas were deemed to be part of the national estate and thus indivisible for management purposes. Management in this case was interpreted as synonymous with "control", which also conflicted with the Indonesia constitutional view of the sea as common property and therefore equally open to every citizen user. These interpretations result in unclear responsibility for regulating access to marine resources, for resolving conflicts between uses and for ensuring that marine resources are managed on a sustainable basis. No one agency was specifically responsible for coordinating between sectors or between the different levels of government.

There are some 22 statutes and literally hundreds of regulations and ministerial decrees that relate to coastal resources (Patlis et al., 2003). The consequences of this multiplicity of legislation were increasingly unclear (and often overlapping) jurisdictions and lack of coordinated policies. Through the mid 1990s, there were various efforts to stimulate increased local governance efforts in some provinces (Hunt et al., 1998; Crawford et al., 1998), including a model bay management program initiated in Balikpapan bay in east Kalimantan (Dutton et al., 2000). However, for the most part it was "business as usual" because most capacity building projects came into being through the central government. Local government remained dependent on central agencies to define their agendas.

Sofa (1998) undertook an evaluation of coastal management and related projects between 1987 and 1998. In that study she estimated that some US$ 400 million has been spent on coastal and marine resources management projects (excluding fisheries) during that period. Relatively few of these initiatives continued once direct funding via central government agencies ceased and very few of these projects directly impacted the quality of life of coastal communities or quality of marine ecosystems (Dutton, 2005).

Thus, despite a long history of investment in water resources development and more recently watershed, coastal and marine management programs, integrated management of watersheds with the coastal areas they drain to remain uncommon. Programs that address land-based impacts on marine ecosystems; a threat that Edinger et al. (1999) describe as the most critical facing Indonesia's coastal ecosystems, was non-existent. In greater Jakarta, with its rapid economic expansion and attendant population growth, the impact of this lack of integrated programming has been devastating to the Jakarta Bay ecosystem.

2.3. The reform era (1998 to Present)

In May, 1999, the passage of the landmark National Law No. 22 on Regional Autonomy (UU22/1999), moved responsibility for planning and management to local governments in all but a few keys areas such as religion, national defense, etc. Law No. 22 specifically made provision for Provincial governments to exercise authority over the territorial seas out to 12 nautical miles[2] and for District/City administrations to have authority for the first third of that area (up to a maximum of four nautical miles). Law No. 22 decentralizing management authority was made meaningful by passage of the National Law 25 on Fiscal Equalization (UU 25/1999) that transferred budgeting responsibility primarily to districts and city administrations. Suddenly, decision-making, management responsibility and financial resources was pushed to lower levels of government, closer to the users of coastal and marine resources.

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[2] one nautical mile equals 1.9 kilometers

A further boost for the marine sector occurred in October 1999, when then newly elected President Abdurrahman Wahid established a Ministry specifically concerned with the identification and development of marine and coastal resources, particularly fisheries. He appointed a veteran and respected politician to lead the Ministry. This historic decision marked a true watershed in marine resource management and symbolised an increased level of political recognition of the significance of Indonesia's seas.

Improved fisheries and coastal management were two key priorities of the new Minister (Kusumaatmadja, 2000). A comprehensive review of fisheries development policy was quickly undertaken and the subsequent white paper proposed a series of sweeping reforms for fisheries management (PCI, 2001). Those reforms stimulated the Ministry to restructure and improve fisheries management in concert with marine and coastal resources management at large. For example, initial reforms redirected fisheries development policy towards two key objectives (MMAF, 2002)

- utilize fisheries resources to improve the welfare and prosperity for Indonesian people
- conserve resources for sustainable utilization.

Since 2000, the Ministry has been engaged in developing a draft National Coastal Management Act that seeks to empower local governments in implementing integrated land and marine management. This reform has long been proposed (see Rais et al. 1998) and development of the new law has employed an exemplary process of global learning and public engagement in legal drafting (Patlis et al., 2003). Several districts have now enacted regulations relating to integrated coastal management.

Despite these advances, many challenges remain (Dutton, 2005), including:

- a lingering overestimation of, and undue emphasis on, maximum sustainable yield (MSY) as a fisheries management tool;
- ineffective surveillance and regulation of offshore fisheries and illegal long-lining and trawl fishing efforts;
- little understanding of regulations at provincial and local levels and little capacity for implementation;
- inadequate data from which to make management decisions at all scales;
- inadequate integration of land and water management; and
- ongoing lack of inter-departmental coordination on marine conservation and resource management.

3. IAKARTA BAY SETTING AND SIGNIFICANCE

3.1. Overview

Since the earliest day of Dutch colonial settlement, Jakarta Bay has played a critical role as Indonesia's main gateway. The principal port in Jakarta Bay, Tanjung Priok, acts as the direct conduit for goods and services to an estimated 25% of Indonesia's population. Exports and imports through the port represent almost 30% of total

GDP (Batubara, 2005). Jakarta Bay is thus a key engine of national economic development.

Jakarta Bay is bordered by three provincial administrations i.e. DKI Jakarta Province (Indonesia's capital city), Banten Province, and West Java Province – the combined population of these areas is around 35 million and it is estimated that 21 million of these live in watersheds that drain directly to the bay.

3.2. Geography of Jakarta Bay

Geographically Jakarta Bay is located between 106°40'45" - 107 ° 01'19" East longitude and 05°54'40" - 06°00'40" South latitude, and spreads from Tanjung Kait (Banten Province) at the west, to Tanjung Karawang (West Java Province) at the east. Jakarta Bay has an area of 514 km^2. There are 105 small islands in the Jakarta Bay area, known as Kepuluan Seribu (literally Thousand Islands) group. This group of islands recently was set aside as its own regency administration. Although currently still assisted by the DKI Jakarta government, the Thousand Islands Regency is rapidly developing its own governance capacity. Among the most important islands in the Thousand Islands are Nyamuk Besar, Nyamuk Kecil, Damar Besar, Damar Kecil, Anyer Besar, Kelor, Untung Jawa, Rambut, and Ubi Besar islands (UNESCO, 2003).

3.4. Topography and hydrology

The Jakarta Bay coastal area topography is low and flat. In general, the ground surface elevation is less than 5 metres. The shoreline and adjacent sea floor are very much affected by the tides and seasonal flooding, and experience a high level of sedimentation from the big rivers that flow into the Jakarta Bay. The average Jakarta Bay coastline surface slope is less than 2%.

There are thirteen rivers that flow into Jakarta Bay. Some are relatively big rivers such as the Cisadane at the Western part, Citarum and Bekasi at the Eastern part and Ciliwung at the Central part. These rivers have large watershed areas (Indonesian 'daerah aliran sungai) that start from the volcanic mountain ranges south of Jakarta. Several smaller rivers, including Kali Angke, Sungai Grogol, Sungai Mookervart, Kali Krukut, Kali Sunter, Kali Ancol, Sungai Blencong, and Kali Cakung also have large affects on the overall hydrology of Jakarta Bay (Batubara, 2005). All these rivers provide both the initial means of access and transport in the lower floodplain and more recently have become the principal pathways for aquaculture pond development and industrial and urban waste disposal.

3.5. Hydro-oceanographic characteristics of Jakarta Bay

Tides in the bay are diurnal. The surface current in Jakarta Bay is mostly affected by the wind and tides. Basically, the affect of tidal currents is not significant as the maximum spring tide is only 110 cm. During the west monsoon season from November to March, the sea current in north Jakarta Bay flows to the Southeast and

East with the average velocity of 0.4-0.6 m s^{-1} (Figure 2A). In the east monsoon season from May to September the sea water current turns to run generally the opposite way to the northwest, with average velocity at 0.5 m s^{-1} (Figure 2B). Near the shoreline, Jakarta Bay waters reveal a relatively complex pattern as water movement is influence by seashore constructions such as jetties, breakwaters and reclamation areas in the Cengkareng, Muara Karang, Pantai Mutiara, Muara Baru, East Ancol, Tanjung Priok and Muara Cakung drainage areas. The influence of tides is more important, and potentially destructive and dangerous, in areas where interaction with rivers can cause rapid flooding in coastal areas. The current velocity in the internal area of Jakarta Bay ranges between 0.25-0.5 m s^{-1}.

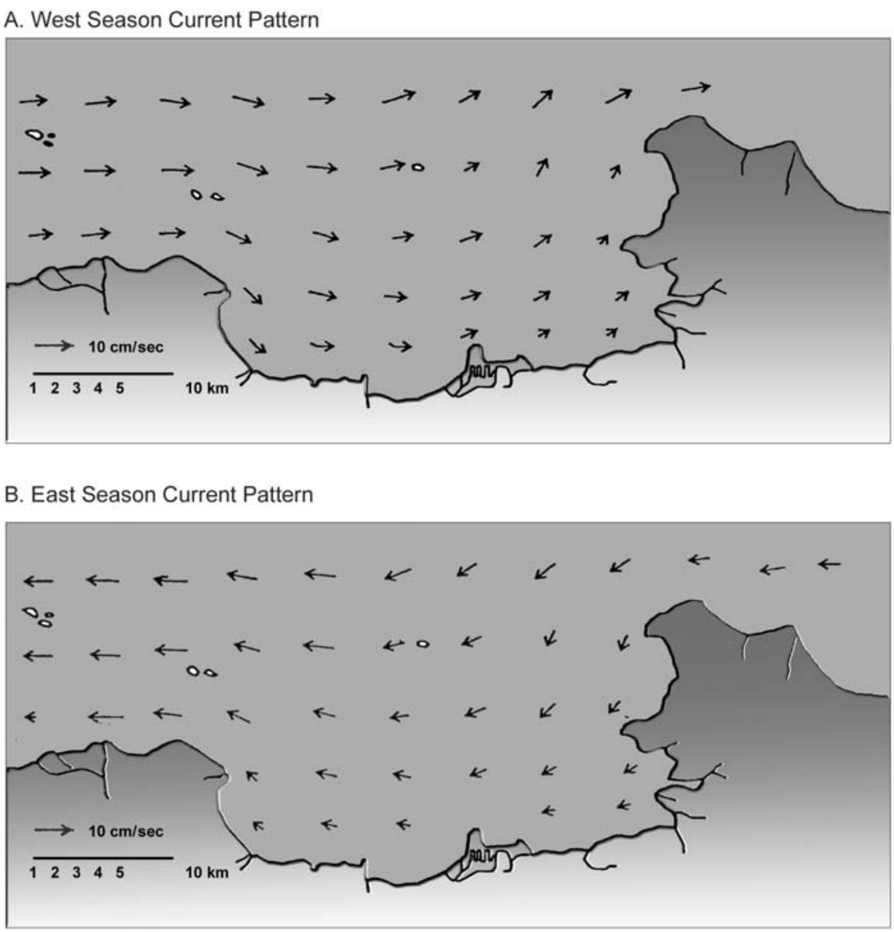

Figure 2. *Jakarta Bay current pattern in the West Season (A) and East Season (B).*

3.6. Water quality of Jakarta Bay

With no effective sewage treatment capacity in any of the large cities in the watershed and largely unregulated industrial waste disposal into rivers directly draining to the bay pollution loads in near shore waters are very high – a Ministry of Environment study of the Jakarta Bay watershed estimated that some 370 kg of mercury enters the bay every hour (MOSE, 1996). The pollution load has increased in line with the increased population in the watershed and largely unregulated industrial development in many urban and suburban areas. Research conducted by Anna (1997) concluded that some specific parameters i.e. COD, phosphate, nitrate, Zn all exceed the regulatory thresholds value.

The sanitation sector is one of the weakest in Indonesia (World Bank, 2004) and nowhere is this more evident than in Jakarta Bay. In Jakarta coverage is negligible with only approximately 2.8% of the population connected to any type of sewerage system and even that infrastructure is of low quality and reliability. Septic tanks are used by about one million people in Jakarta but with a bay watershed population estimated at 21 million, most of which drain household wastes directly into canal and rivers draining to Jakarta Bay, the daily bay waste loading is enormous.

Obviously, port activity at Tanjung Priok also contributes to the decrease of Jakarta Bay water quality. The result of the research carried out by the DKI Jakarta in 1998 at Tanjung Priok Port indicated that Oxygen, Chemical Oxygen Demand (COD), phenol, ammonia, nitrate, sulfide, cadmium, copper, tin, lead and chromium have far exceeded global seawater threshold values for marine biota.

3.7. Ecosystem and natural resources uses of Jakarta Bay

3.7.1. Mangrove ecosystem

Figure 3. *Mangrove ecosystem at Kamal Muara, coast of West Jakarta.*

The Jakarta Bay foreshore and some of the offshore islands still have remnant mangrove ecosystems. The largest and most intact areas are found at Kamal Recreation Forest, Muara Angke Fauna Reservation, Angke Kapuk, Kemayoran Protected Forest and in the Cilincing – Marunda vicinity (DKI Jakarta Forestry Services, 1996) (Figure 3). Within the Kepulauan Seribu Islands group, islands such as such as at Rambut, Bokor, Untung Jawa, Lancang, Lancang Besar, Peteloran Barat, Penjaliran Barat, and Penjaliran Timur have the most intact forests. A study carried out by the DKI Jakarta Forestry Services in year 1999 indicated that mangroves growing in the western part of Jakarta Bay are still in relatively good condition, while in the Cilincing-Marunda vicinity, growth is relatively poor, with large tracts of forest now fragmented.

Mangroves in the Kamal Recreational Forest, Muara Angke Fauna Reservation, and Angke-Kapuk Protected Forest are relatively homogenous and dominated by mangrove of Avicennia spp, while Rhizophora spp is found in relatively smaller numbers and in sporadic growth patterns. Tree vegetation is Avicennia marina, A. officinalis, and A. alba (BAPEDALDA DKI, 2001). The mangrove vegetation at Kepulauan Seribu Islands group has been damaged from abrasion, pollution, solid waste and illegal logging. Mangrove species and areas found in Kepulauan Seribu Islands group are presented in Table 1.

Table 1. *Mangrove vegetation at Seribu islands group reservation area. (Source: BAPEDALDA DKI, 2001).*

No	Location	Area (ha)	Number of Species	Species
1	Fauna Reservation Rambut Island	27.00	9	Rhizophora stylosa, R. mucronata, Sonneratia alba, Bruguiera gymnorrhiza, Avicennia marina, A. alba, Ceriops tagal, Excoecaria agallocha, Xylocarpus granatum
2	Nature Reserve Bokor Island	25.23	2	Rhizophora mucronata, Sonneratia alba
3	Untung Jawa Island	31.00	2	Rhizophora mucronata, Avicennia alba
4	Lancang Besar Island	16.50	3	Rhizophora mucronata, Sonneratia alba, Avicennia marina
5	Nature Reserve Peteloran Barat Island	11.30	3	Rhizophora mucronata, Ceriops tagal, Avicennia marina
6	Nature Reserve Penjaliran Barat Island	8.30	4	Rhizophora stylosa, Ceriops tagal, Sonneratia alba, Avicennia marina
7	Nature Reserve Penjaliran Timur	6.80	4	Rhizophora stylosa, Ceriops tagal, Sonneratia alba, Avicennia marina

3.7.2. Coral reefs

In the "Reefs at Risk" study, Burke et al. (2002) noted that when naturalist J. Umbrove arrived in Jakarta Bay in 1928 he was struck by the beauty of the coral reefs in the bay and yet, even in those days, development impacts in the reefs were already very evident. Today, due to pollution, mining, destructive fishing, shipping, tourism, private residential development, oil and gas development and coral bleaching, average coral cover in the area is just five percent, making these some of the most degraded reefs in Indonesia (Tomascik et al., 1997) (Figure 4). The coral reefs closest to Jakarta are most affected, but over fishing and destructive fishing practices have also devastated the outermost reefs. The El Nino episodes of 1982 and 1998 further stressed the reefs, causing 90-95 percent mortality.

The most rapid decline in reef condition occurred since 1970. UNESCO (2003) notes that on Pari Island and Air Island reefs, where good benchmarks of coral condition existed, coral cover declines from 80% to 15% and from 70% to 30%, respectively, were recorded between 1970 and 1996.

Figure 4. *Coral reef at Pramuka Island, Kepulauan Seribu Islands Group, Jakarta Bay.*

3.7.3. Seagrass Ecosystem

Another key ecosystem found in Jakarta Bay waters is seagrass. Usually this ecosystem is found close to coral reefs and supports the coral reef ecosystem as nursery, feeding and spawning areas for fish species and other marine biota. The most common types of seagrass found in Jakarta Bay comprise Thallasia, Syringodium, Thalasodendrum and Cymodocea genera. There are also various genera of seaweed such as Halimeda, Padina, Caulerpa, Sargassum, Turbinaria and Euchema.

3.7.4. Conservation areas

The Kepulauan Seribu National Park (Thousand Islands National Park) is one of the pioneer conservation areas in Indonesia. In 1931, Bokor Island was declared a wildlife reserve through Letter of Decree No. 6 Year 1931, by the Dutch Indies Governor General. This island was set aside as a botanical sanctuary to protect mangroves (i.e. Rhizophora mucronata and Sonneratia alba), rare birds (e.g. Ducular bicolor, Fregata ariel, and Oriolus chinensis) and the long tailed monkey (Macaca fascicularis).

In 1970, in response to concerns about sand mining, the Governor of Jakarta issued a protection order for coral reefs and some islands in Jakarta Bay that eventually led to the formal declaration of a National Park in 1982. As Alder et al. (1994) note, the subsequent process of establishing a management plan for the park that accommodated the disparate interests of key stakeholders proved a very challenging task. Even today enforcement of park regulations is low and there is clear evidence that the park has not achieved intended conservation objectives. Thus while the park covers some 108,000 hectares, it has almost no areas that are "self-protected" (by access restrictions) or effectively managed (to limit destructive uses), with the exception of small islands and adjacent areas that are under private ownership.

On some of the small islands and foreshore areas where more intensive conservation management has been implemented, results are mixed. For example in the Muara Angke Fauna Reservation, the mangrove ecosystem has degraded into swampy forest with significant canopy cover loss. The fauna reservation was intended to protect the birds that depend on similar species at Rambut Island for feeding grounds. Populations of the long tailed monkey (Macaca fascicularis) in the reservation have now declined to unviable levels due to poaching and urban encroachment.

4. COASTAL AND MARINE BASED ECONOMIC ACTIVITIES OF JAKARTA BAY

4.1. Urban and Industrial Development

Approximately 60% of the bay shoreline has been modified for urban, industrial or infrastructure development, with a further 30% developed for agricultural or aquaculture. In recent years there has also been an extensive reclamation program to create new land for port, urban and recreational (including golf and hotel) facilities.

Some 17,000 people live on the islands within the bay and most of these depend directly on fisheries, tourism and industrial (including shipbuilding and port labor) activities.

4.2. Fisheries Activities

Indonesia's fisheries produced an estimated 4-5 million tonnes per annum, although reliable data are hard to obtain on actual landings and trade. Marine fisheries comprised the majority of total fish production and the major types of fisheries are (in order) purse seine, lift net, trammel net and pole and line tuna fisheries. Some 94% of this estimated production is reported to be captured by small scale fishers (Dutton, 2005). However, because overall catch is significantly under-reported that figure is misleading. There was been a sharp increase in subsistence and export fisheries in most areas as a result of the Asian monetary crisis due to both the high value of capture fisheries and the increased costs of alternative protein sources.

In Jakarta Bay, once strong commercial fisheries are in serious decline (Yowono, 1998), with most fishers struggling to cover fuel, equipment and labor costs. There mainly are two kinds of operations still continuing in the Jakarta Bay capture fishery sector, namely

(1) fixed fishing activity generally is dominated by fish traps (bagans). There are some 850 of these structures in the shallow waters of Jakarta Bay, but less than 30% of these are maintained and used regularly (Figure 5); and,

(2) mobile fishing activity generally dominated by net and fishhook/fishing rod fishing. It is estimated that there are around 2600 fishers in the various Jakarta Bay coastal communities who derive their principal source of income from fishing.

Figure 5. *Fish trap ('bagan') distribution in Jakarta Bay*

Increasing numbers of Jakarta area residents also engage in fishing for occasional subsistence and recreational purposes. This additional activity is more significant than previously thought, with some 2500-5000 fishers engaged in "recreational fishing" activity in the wetlands around Jakarta bay on holidays.

In addition to marine capture fisheries, aquaculture is an increasingly important component of Indonesia's fisheries. The most commonly cultured fish species are the ubiquitous milkfish (73%), tilapia (12%) and mullet (6%). Some 2.05 million fish farmers were employed in 1998, with the majority (65%) in Java.

In the Jakarta Bay area, the estimated 1500 ha of aquaculture ponds are mostly "sour" and unproductive due to mismanagement of ponds, pollution from nearby development, flooding and lack of technical skills (Animation 1). Many ponds, developed during the peak of aquaculture development in the bay area in the 1970s, have now been reclaimed for housing, industrial, leisure (golf course) and infrastructure development (Figure 6).

Animation 1. *Green mussels (Mytilus edulis) raft spreading at Jakarta Bay.*

Figure 7. *Fish pond area at Kamal Muara, coast of West Jakarta.*

5. TANJUNG PRIOK HARBOUR ENVIRONMENTAL SETTING

5.1. Indonesian harbour and navigation characteristics

According to National Law No. 21 of 1992 (UU21/1992) and Government Regulation (PP) No. 69 Year 2001, a harbour is defined as a place that consists of land and surrounding specified border waters, as a place for ships to anchor and harbour, passengers to embark and disembark and/or goods loading and unloading, is equipped with navigation safety facilities and port activities support, and also a place for various intermodal transportation activities. “Harbour affairs” covers everything related to port operational and other activities in the framework of carrying out its function.

Ports are differentiated into two main categories, i.e. public ports and special ports. A public port serves public/community interests while a special port is only to support specific activities. Based on its role and function, ports are further differentiated into five hierarchies: (1) hub international ports are main primary ports, (2) international ports are main secondary ports, (3) national ports are main tertiary Port, (4) regional ports are primary feeder ports, and (5) local ports are secondary feeder ports. Jakarta Bay includes all these types of ports when including the small port facilities on the islands and along the bay coastline.

Navigation is defined as every activity related to navigation including supporting facilities, communication, hydrographic characteristics, channels, ship draft/construction management, salvage and underwater works to secure navigation safety. Government Regulation No. 81, 2000 (PP81/2000) on navigation states that navigation channels, natural or man-made, are defined in terms of depth, width, navigation obstacles and other aspects that define and relate to navigation safety. It further states that to maintain navigation safety, clear safe and secure zones within and in the vicinity of navigation channels need to be established. For Jakarta Bay, navigation requirements are well established and regulations enforced by the Indonesia Navy, as well as other sector specific enforcement authority such as the Ministry of Marine Affairs and Fisheries patrol boats.

5.2. Harbour hydro-oceanographic factors

As described above, wind is one of the hydro-oceanographic factors that influence port activities in Jakarta Bay. Besides as a wave stimulant, wind plays an important role in dispersal of pollutant and waste loading in the bay. Current is also a hydro-oceanographic factor that influences port water quality conditions. Waters depth is also an important factor affecting Jakarta Bay water quality. The overall depth of Jakarta Bay is shallow and combined with the rather slow speed of currents, and the often prevailing landward wind direction, sediment deposition is high.

5.2.1. Harbour development needs

At present Tanjung Priok Port has an area of 1028 ha consisting of 424 ha of deep-water harbour or waters that is bordered by breakwaters and 604 ha of land. (Figure 1). Tanjung Priok deep-water harbours are divided into five (5) navigation channels and seven (7) harbour basins of 9,507m breakwater. Tanjung Priok Port has 10,597m wharf facility with seventy (70) berths for various ship tonnage and draft. In addition, there are other available facilities such as 45 warehouses, open piling areas, stock tanks and container areas.

The number of ships calling at Tanjung Priok Port has increased from year to year. Calls reached 17,086 in 2001 or an average of 47 calls per day, an increase of 41% compared to ship calls in 1991. The tonnage of ships tends to be bigger, as well. In 1991, average ship tonnage was 3,487 GRT. In 2001, average ship tonnage was 5,226 GRT showing an average increase of 1.5% over the preceding ten year period. The loading and unloading activities at Tanjung Priok Port showed a corresponding increase, with the yearly loading and unloading average increases at 8.7% for general cargo, 2.2 % bag cargo, 2.6 liquid bag cargo, 7.9 % dry bulk cargo and 11.9 % for containers (Batubara, 2005).

Capacity estimates indicate that Tanjung Priok Port can serve cargo ships at the maximum rate of 16,000 – 16,500 calls per year, or at the average of 45.2 ships per day – a level already exceeded. To mitigate and manage expected continuing increases in ship calls, and the resultant impacts on Jakarta Bay, expansion and improvement of Tanjung Priok Port will be compulsory. The present channel and harbour basins are very narrow and can only serve one-way traffic with limited area within the turning basins. This is already restricting ship calls due to safety concerns. To increase the capacity and efficiency of the port, and increase safety two-way channel traffic must be developed.

5.2.2. Environmental impact of harbour development

There are significant conflicts in bay utilization between port and fishery activities. In general, environmental impacts from Tanjung Priok port falls largely on fisheries and conservation areas through increased water pollution. Port and navigation activities negative impact fisheries through pollution from ships liquid waste such as ballast discharges, fuel and petroleum product spills, and other waste discharges. The range of discharges affecting Jakarta Bay includes ship movement and resulting turbidity increases, liquid waste such as wastewater such as ballast water, fuel spills, accidents related to transportation and storage of fuels and oils, solid waste produced during ship loading and unloading, and other storage and transportation activities. Additional impacts come from ship building, repair and maintenance activities and dredging and dumping of dredged material.

This range of impacts has resulted in decreased overall fish stock because of damage to spawning and nursery grounds, and loss or damage to mangrove, coral reef, and sea grass ecosystems. Pollution and contamination of habitat has led to replacement of original species with species able to withstand more polluted habitats. In addition, financial losses are significant as a result of decreased quality and aesthetic values in recreational areas and water sports such as scuba diving and swimming. Particularly worrying are potential health impacts to populations relying on fish caught in Jakarta Bay, although little research has been done in this area.

Table 2. *Port and other activities interaction. (Source: Batubara, 2005)*

Activity Component	Description	Impact on the Navigation Activity	Sensitivity towards Navigation activity
Cultivation	Fish and green mussel culture, catching green mussel fingerling, rearing and harvesting.	The presence of raft disturbs the navigation channel and harboring area.	Sensitive toward pollution due to the port and ship activities.
Catching	Set catching device and catch fish within a certain cycle	This activity could prevent the ship's movement, while catching device such as net and long line could endanger a ship's safety	Navigation activity could damage the fishing gear.
Fishery Harbour	In and out of fishery vessels to the fishery harbour.	Fishing vessels increase the crowded traffic	This activity is not sensitive towards Port activities
Tourism	Tourism object could be waters of natural resource with aesthetics value.	There is no impact towards port activities	Sensitive to pollution due to the port and ship activities
Conservation Area	Natural Conservation such as natural reservation, national park, protected area etc.	Have no impact on the port and navigation activity	Sensitive to pollution due the port and ship activity

5.2.3. Environment Management Implication of harbour development

Given the impending need to upgrade Tanjung Priok Port navigation channels, turning basins and harbour facilities, there is an opportunity to minimize environmental impacts while securing optimal harbour functions. Management implications for Tanjung Priok Port development can be differentiated into three categories: (1) adjusted management of port activities, (2) management related to legal requirements, and (3) management related to spatial utilization conflicts.

Adjusted management of port activities

Socialize new management regimes among stakeholders utilizing the port and Jakarta Bay to modern management principles and establish Jakarta Bay as a Port Activities Zone governed by minimal performance standards.

Control ship activities in areas that provide maximum security, safety and protection against environment pollution.

Maintain navigation channels, turning basins and harbour waters using environmentally based systems to maintain bay functions, including dredging and disposal of dredged materials in environmentally safe disposal. Prevent pollution from ship waste through establishment of appropriate regulations and an effective monitoring and control program. Establish and require use of solid and liquid waste reception facilities for port calls.

Improved management of legal aspects
Declare within Jakarta Bay a new Tanjung Priok Port Working Area (DLKR) and Jakarta Bay Environmental Necessity Area (DLKP) to establish clear guidelines for activities in various areas within the port, i.e., permit only designated activities in specific use areas. Implement existing laws and regulations governing discharges to river systems by industries and municipalities.

Resolve conflicts through improved marine and land spatial zoning
Allow fishing in DLKP using only mobile fishing gear and zone the bay to establish an appropriate spatial balance of protection and use areas Allow use of fish traps only outside of DLKR. Work with private landowners and the tourism sector to better define property rights, obligations and opportunities for co-management of areas to be protected. Work with provincial and municipal administrations within the Jakarta Bay watershed to develop integrated coastal and marine management plans that reduce pollutant loads entering Jakarta Bay and establish a more coordinated framework for bay management.

6. CONCLUSION

Recognition of the need for better management of the economic, social and ecological assets of Jakarta Bay, and for improved integration of management of land and water generally, is increasing amongst resource users and resource managers. On February 3, 2000, at the first seminar on integrated management of Jakarta Bay, the Governor of Jakarta acknowledged that the traditional sectoral approach to management of the city and Jakarta Bay was the root cause of such diverse problems as flooding, pollution, poverty and health problems, and natural resource depletion.

Surprisingly, it was in the Segara-Anakan region (of south Java) that integrated watershed and coastal area management was first pioneered in Southeast Asia. However, those efforts were mostly academic and until recent efforts in the Balikpapan Bay area of East Kalimantan there have been few working models of effective integration of bay and watershed management (Proyek Pesisir Kalimantan Timur, 2002).

The lack of an integrated approach to pollution management has had profound, but inadequately documented, effects on the health of coastal communities and ecosystems in the Jakarta Bay area and has resulted in significant economic disruption and inefficiency. The net economic loss to Indonesia from the types of coral reef degradation that has occurred in Jakarta Bay amounts to a staggering $30 billion over the next 25 years (Cesar et al., 1997). If one extrapolates the total amount of restoration and rehabilitation needed to recreate healthy and functional ecosystems in Jakarta Bay, and the costs of protecting human health from the impacts of current and future pollution, that figure may well under represent the true cost to the country of the legacy of past inappropriate development.

All stakeholders in Jakarta Bay managers urgently need to review how the bay is valued and used. A fundamentally new approach to bay management is needed. Central to that approach are three governance requirements:

(1) Establish a long-term plan for bay and port development that identifies optimal areas for port and transport facilities, as well as other land and sea uses, and that protects key areas that need to remain off limits to development in order to sustain natural systems and ecosystem processes that a healthy bay requires;

(2) Establish a financial management system that enables a "user pays" system to generate appropriate levels of funding for Jakarta Bay management. Currently, most users of Jakarta Bay resources pay only a modest proportion of total costs of operations and there is no "operational fund" for bay management; and

(3) Establish a seamless intersectoral and hierarchical structure for bay governance that involves all levels of government. Given the importance of the bay to the national economy, we recommend the establishment of an independent port and bay management authority including representatives of government, industry and the community comprising an executive body and reporting directly to the President.

7. ACKNOWLEDGEMENTS

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The views expressed in this paper are those of the authors and do not reflect any institutional affiliation, past or present.

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CHAPTER 25

DARWIN HARBOUR: WATER QUALITY AND ECOSYSTEM STRUCTURE IN A TROPICAL HARBOUR IN THE EARLY STAGES OF URBAN DEVELOPMENT

A. D. MCKINNON, N. SMIT, S. TOWNSEND,
AND S. DUGGAN

1. INTRODUCTION

The city of Darwin (Figure 1) is Australia's northernmost city (latitude 12º 27'S) and closest port to SE Asia, with a natural deep-water harbour. Aboriginal people lived in the region for at least 50,000 years prior to 1869, when the British established the city of Darwin and began a period of more intense use of the harbour and modification of the surrounding environment. Early use of the harbour by Europeans included a substantial pearl shell fishery – in 1884 alone, 50 tonnes of shell were removed from the harbour. The population of the Darwin region was modest in the first 80 years of the city's history – in 1947 it amounted to only 2,500 people. When Cyclone Tracy destroyed Darwin in 1974 the population was 47,000, and in the 2001 census it was 110,000, accounting for 55% of the total population of the Northern Territory (NT). Current projections of regional population growth amount to a population of 228,000-309,000 by 2021 (Australian Bureau of Statistics). However, despite the population growth since the establishment of the city, only since the reconstruction after Cyclone Tracy has there been significant economic development in the catchment and along the foreshore of the harbour. The primary shoreline developments have been ports, marinas and residential living.

The NT economy currently has an annual gross state product (GSP) in excess of \$AUS 9 billion. Mining is the largest industry, accounting for 22% of GSP in 2001 – 2002 (NT Govt, 2002a), followed by tourism, defence, construction and government services. Darwin already exports more live cattle than any other Australian port, and is now expanding its role as a supply base for both offshore and onshore gas projects. In 2003 – 2004 export cargo comprised mainly livestock (26%), metal products (31%) and petroleum products (18%) – the latter comprised 68% of import

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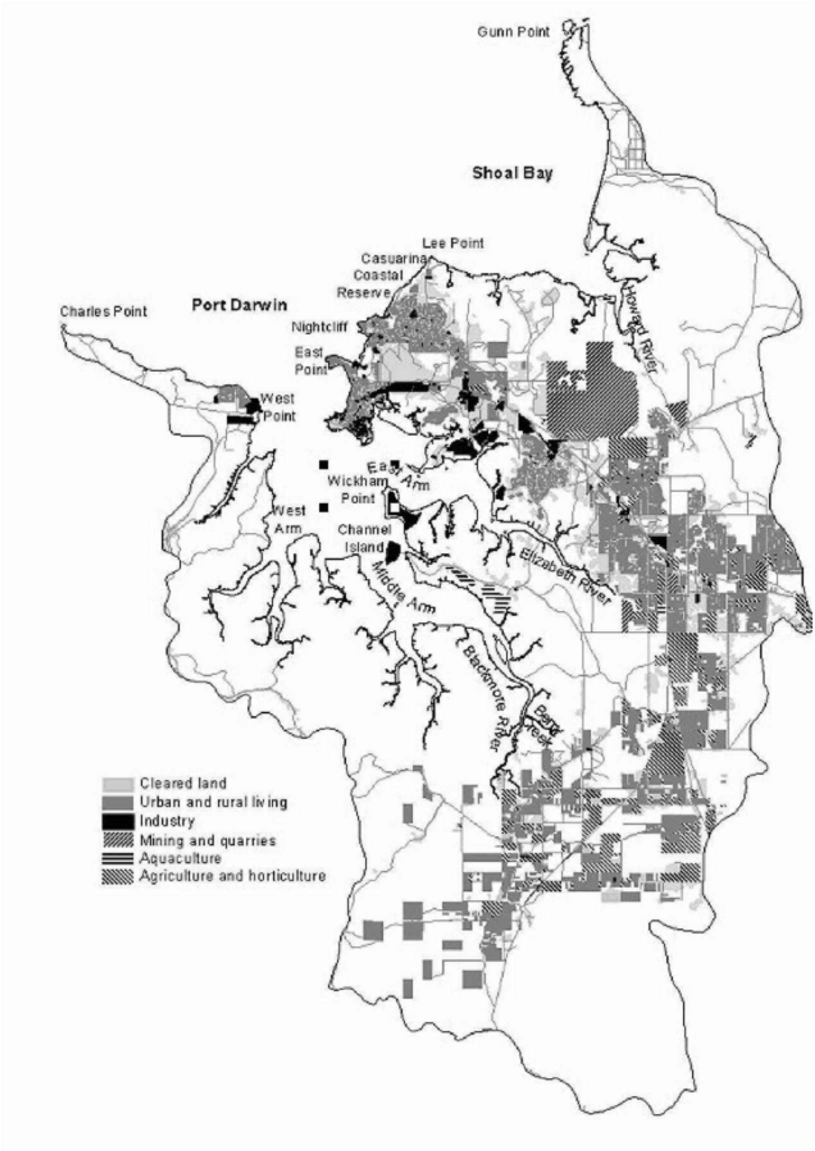

Figure 1. *Darwin Harbour catchment showing urban areas (indicated by road network), cleared native vegetation and mangroves.*

cargo (http://www.nt.gov.au/dpa). During this time the port had 4,400 vessel visits, and handled 1.3 million tonnes of cargo. Recent improvements in infrastructure, such as the Adelaide-Darwin rail link and an expansion of port facilities in the East

Arm of Darwin Harbour, will see the expansion of the harbour as an entrepôt for trade. The NT government has ambitious plans for the expansion of the liquid natural gas industry with a vision to make Darwin Australia's fourth gas hub. The initial phase of this expansion is the construction of a pipeline from production platforms in the Timor Sea to a processing plant at Wickham Point, within Darwin Harbour. Growth in the regional economy, and particularly in the natural gas sector, will inevitably increase human impacts on the coastal ecosystems of the Darwin area.

Darwin Harbour is rich in natural heritage and is relatively pristine considering that it is the backyard of a capital city. There is no other capital city in Australia that can proudly say, for example, that marine turtles nest on its beaches or that 25 of the 48 species of waterbirds found along its shoreline are listed under international migratory bird agreements. Coastal and marine habitats in Darwin Harbour are diverse and contain considerable biodiversity.

The combination of a near-pristine ecological state and an ambitious prospectus for economic growth requires a pro-active approach by environmental scientists and managers. This chapter summarises the current state of knowledge of the Darwin Harbour area based upon the work of NT government agencies and on measurements of Darwin Harbour water quality and ecosystem processes made by the Australian Institute of Marine Science from 2002 – 2004. It attempts to capture the essence of a tropical harbour likely to undergo significant change in the next few decades but as yet apparently not greatly impacted by human activities.

2. RESULTS AND DISCUSSION.

2.1. The Darwin Harbour catchment

The spatial extent of Darwin Harbour adopted in this chapter is defined as the waters inshore of a line drawn between Gunn and Charles Points (Figure 1), which includes Port Darwin and Shoal Bay. The catchment area of Darwin Harbour is small, being 2417 km^2 compared to a harbour surface area of 1220 km^2 at the highest astronomical tide (c.f. 660 km^2 at lowest astronomical tide); a terrestrial to water ratio that approximates 2:1. Darwin Harbour is macro-tidal, with a maximum tidal range of 7.8 m, and mean spring and neap tidal ranges of 5.5 m and 1.9 m (Byrne, 1988). Such a large tidal movement produces strong currents of up to 2 m s^{-1} (Byrne, 1988) that transport sediment within and across the harbour's boundaries.

Darwin has a wet/dry tropical climate, with the wet season dominated by monsoonal and cyclonic activity, as well as convective thunderstorms. Darwin's annual rainfall averages 1714 mm, with 64% of rain falling between January and March, 97% between October and April, and negligible amounts between May and September. River flow reflects this highly seasonal rainfall pattern, commencing in December or January, with maximum flows during periods of heavy monsoonal and cyclonic rains between January and March. At the end of the wet season flows decline and most rivers and streams cease to flow between May and July. Consequently, riverine inflow into the harbour over the dry season is negligible.

Several rivers flow into Darwin Harbour, the largest being the Blackmore and Elizabeth rivers that flow into the middle and east arms respectively of Port Darwin. In addition, the Howard River flows into Shoal Bay. Collectively these three rivers drain half of the harbour's catchment. A spring-fed creek (Berry Creek) feeds into the Blackmore River throughout the year, but this input is negligible compared to the flow subsequent to periods of heavy rain. Run-off from rural and savannah woodland catchments into the harbour averages 33% of annual rainfall and in wetter years can be as high as 48% (Hatton et al., 1997). In the highly urbanised catchments of Darwin however, runoff into the harbour can be as high as 78% of annual rainfall due to the large area of impervious surfaces (Haig and Townsend, 2003). The only substantial river impoundment in the catchment is Darwin River Reservoir in the upper reaches of the Blackmore River catchment. The reservoir's catchment constitutes 8% and 23% respectively of the total harbour and Blackmore River catchment areas.

The Darwin Harbour catchment is ancient and highly weathered. Tertiary sediments, that have been repeatedly weathered and redeposited, dominate the land surface (Wood et al., 1985). As a result, the catchment's soils are relatively infertile and the terrain low lying with most of the catchment being less than 50 m above sea level, and having a maximum elevation of only 140 m. Annual fluvial export coefficients of sediment, phosphorus and nitrogen from rural and undisturbed catchments are low (2 -9 kg N ha^{-1} yr^{-1}; 0.08 - 2 kg P ha^{-1} yr^{-1}; 80 - 600 kg sediment ha^{-1} yr^{-1}; Water Monitoring Branch, 2005) which is typical of the region, and lower than that reported for other tropical regions where the coefficients may be two orders of magnitude higher (Townsend and Douglas, 2000). This equates to flow-weighted mean concentrations of suspended sediment, total phosphorus and total nitrogen from rural and largely undisturbed catchments of 10 - 20 mg l^{-1}, 0.010 - 0.015 mg l^{-1} and 0.30 - 0.35 mg l^{-1} respectively (Padovan, 2002).

The catchment's dominant vegetation communities are savannah woodlands and forests. Approximately 19% (46,000 ha) of native vegetation has been cleared and replaced by housing developments, small-scale agriculture, horticulture and grazing land-uses (Fig 1). The ecological condition of riparian land along the catchment's rivers and streams, which is important in reducing diffuse water pollution, remains intact with only ~11% cleared of native vegetation (Water Monitoring Branch, 2005). Urban and light industrial land-use, which represents ~3% of the catchment, has the greatest impact on the quantity and quality of runoff into the harbour, whereas mixed land-uses have not been shown to differ from undisturbed areas (Water Monitoring Branch, 2005).

Urbanisation increases annual export loads of nutrients, metals and sediment to the harbour by between 3 and 90 times compared to pre-urbanisation loads (Table 1). Additional nutrients are discharged to the harbour from treated wastewater and aquaculture point sources, with the most significant impact being phosphorus from wastewater sources (Table 1). The amount of pesticide entering the harbour is negligible (Waugh and Padovan, 2004). Dieldrin is the most frequently detected pesticide and is no longer used, but has a long lifetime in the environment. The water quality of runoff into Darwin Harbour from most of the catchment is not significantly impacted by human activities in the catchment, with the exception of urban land-use and point source discharges that have a localised impact on harbour water quality (Padovan, 2003).

Table 1. *Total annual catchment loads to Darwin Harbour.*

Contaminant	Pre-urbanisation load* (t)	Current Catchment load* (t)	Ratio of load increase with only urbanisation	Treated Sewage Load (t)	Aqua-culture load (t)	Current Total Load (t)	Ratio of load increase with urbanisation and wastewaters
Nitrogen	480	530	1.1	250	17	797	1.7
Phosphorus	17	29	1.7	70	2.0	101	5.9
Aluminium	370	440	1.2	-	-	440	1.2
Arsenic	0.38	0.48	1.3	-	-	0.48	1.3
Cadmium	0.093	0.11	1.2	-	-	0.11	1.2
Chromium	0.77	1.6	2.1	-	-	1.6	2.1
Copper	1.3	2.0	1.5	0.2	-	2.2	1.6
Iron	660	700	1.1	-	-	700	1.1
Manganese	15	17	1.1	-	-	17	1.1
Nickel	0.63	0.78	1.2	-	-	0.78	1.2
Lead	0.58	2.2	3.8	-	-	2.2	3.8
Zinc	5.4	17	3.1	-	-	17	3.1
Total suspended sediment	16,000	20,000	1.3	1,400	-	21,400	1.3
Mineral suspended sediment	12,000	15,200	1.3	200	-	15,400	1.3
Organic suspended sediment	4,000	4,800	1.2	1,200	-	6,000	1.5

*Median values (Water Monitoring Branch, 2005)

2.2. Geomorphology

Post-glacial flooding of a plateau dissected by the Blackmore and Elizabeth rivers created Darwin Harbour's present coastline (Seminuik, 1985). These river systems drain 590 km^2 and 290 km^2 respectively, and have resulted in a gradual deposition of terrigenous sediments since sea level stabilised at its present height some 6000 – 7000 years ago. Sediments in the catchment are predominately fine-grained, mainly clay and silt. Creeks and rivers transport coarser material (e.g. sand) into the estuary during the wet season, though much is trapped by coastal vegetation, both riparian and mangrove. Soft sediments are also derived from settlement of water column detritus and marine geochemical processes such as the precipitation of calcium carbonate.

Where the marine and terrigenous sediments meet, soft sediments are reworked and sorted by wave, storm, wind and tide induced currents. The strength of tidal currents determines which sediments remain in suspension and which are deposited, resulting in gradual sorting. As a result, hard substrates generally occur in high-energy areas. Soft substrates form by wave action moving coarser sediments (sand) landwards and finer sediments (mud and clay) to deeper waters. Strong currents in river and estuary channels erode fine sediments (mud, silt and fine sand) from the substrate, leaving coarser material (coarse sand, gravel and rubble) behind, and redepositing the finer suspended sediments in low current areas such as subtidal and intertidal mud flats and mangroves. Coastal cliffs and cliff screes are scattered along the northern coastline. Weathered and laterised sandstones and conglomerates form the majority of intertidal and subtidal rocky platforms and outcrops, and the rock bars in the mouths of tributaries. Seminuik (1985) and Mitchie (1988) classified the harbour's floor into a range of soft-substrate units on the basis of geomorphological processes. The main channel within the harbour and its arms is composed of coarse sand and gravel, which is fringed successively by fine sand and extensive intertidal and subtidal mud flats in the more sheltered parts of the harbour. Mangroves and salt flats form the transition between intertidal and subtidal mud flats and terrestrial environments.

2.3. Water quality in Darwin Harbour

Water quality within Darwin Harbour varies according to three factors: stage of the tide, location within the harbour and season (Padovan, 1997).

2.3.1. Tidal effects

The large tidal range and severe tidal currents found in Darwin Harbour result in tidal energy being a major determinant of water quality. Concentrations of suspended sediment and total phosphorus, as well as turbidity vary over the tidal cycle (Padovan, 1997; Wilson et al., 2004), and are most pronounced in the upper reaches of the harbour.

High turbidity is characteristic of Darwin Harbour: secchi depth typically varies from ~1 m in the Blackmore River to >10 m in the outer harbour. This turbidity originates from a number of sources: catchment inflow, resuspension and transport of mangrove mud and bottom sediments as a result of high tidal energy, flocculation of organic material within the water column, and from the standing stocks of phytoplankton. Mangrove material is likely to be a major component of the particulate material within the harbour.

Tidal effects on water quality differ markedly during neap and spring tidal cycles (Figure 2). In this example, taken from Station 6 (~27 m deep near the mouth of Port Darwin; Figure 3) in February 2004, the tidal range during the neap cycle was 3.6 m and during the spring cycle was 6.9 m. During the neap cycle the water column was stratified with respect to turbidity, with near-bottom water having higher turbidity, particularly during the turn of the tide. Chlorophyll fluorescence showed near-surface maxima during low tide periods, possibly because of the seaward advection of inner-harbour water containing more plankton. During spring tides both turbidity and chlorophyll showed maxima at low tide, again because of the advection of more turbid inner harbour water. There were no clear patterns in dissolved nutrients over either tidal cycle.

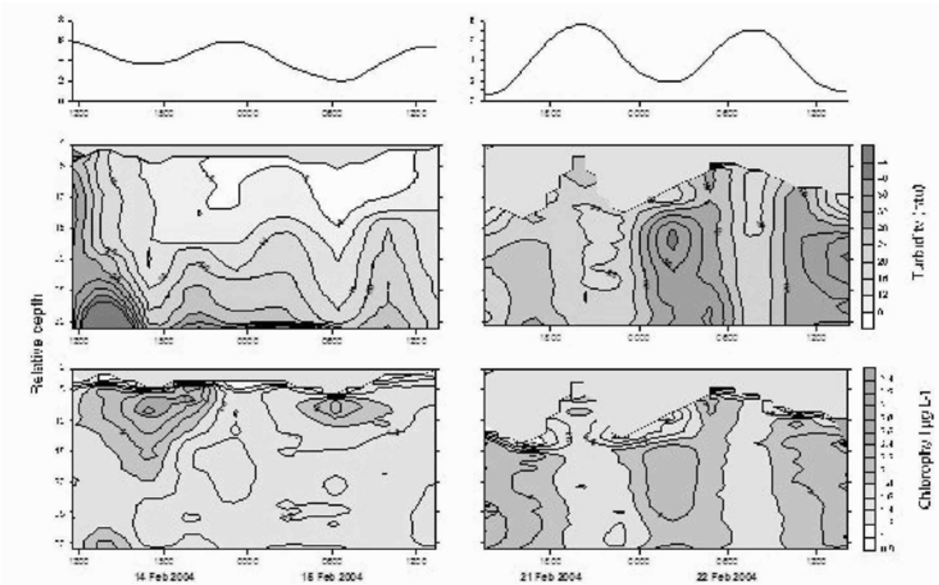

Figure 2. *Changes in turbidity and chlorophyll concentration occurring over neap (left panel) and spring (right panel) tidal cycles in Darwin Harbour. The data originates from surface-to-near bottom profiles with a conductivity-temperature-depth (CTD) profiler (Seabird SBE 19plus). This instrument was fitted with a fluorometer (Wetlabs ECO-FL), and optical backscatter (D&A Instruments OBS-3) sensor. Note that turbidity is derived from OBS data, and that the nephelometric turbidity units (NTU's) are not necessarily comparable with those derived using other instruments, since these are instrument-specific.*

2.3.2. Seasonal and spatial differences

Water temperature in Darwin Harbour varies seasonally, and during 2003 – 2004 ranged between 24.4 °C in the dry season (June 2003) to 33.4 °C in the "build-up"to the wet season (December 2003). There is little spatial variation in temperature within the harbour. Salinity depends on the extent of rainfall during the wet season. In 2003 – 2004 the Blackmore River contributed more fresh water to the harbour than the Elizabeth River (Table 3). For most of the year, Darwin Harbour is well mixed with respect to salinity. Sections through the harbour in the wet (February) and dry (June) seasons of 2003 (Figure 3) illustrate the transformation of the harbour from a relatively homogeneous (in terms of salinity and temperature) embayment into an estuary. For most of the year (as exemplified by the June section) there is little vertical stratification in terms of either turbidity or salinity, and only a very gradual pattern of decreased turbidity and increased salinity with distance from the upper Blackmore River to the outer harbour. During February 2003 a distinct salt wedge formed, with highly turbid low salinity water overlying marine water from the Timor Sea. Turbidity decreased on the seaward end of the harbour as a result of the dilution of inner harbour water typical of mangrove environments with the more oligotrophic, low turbidity waters of the Timor Sea.

Dissolved nutrient concentrations vary spatially and seasonally. Figure 4 contrasts the concentrations of major dissolved nutrients (NH_4, NO_2, NO_3, PO_4) in the Blackmore River estuary with those at the mouth of Darwin Harbour (Station 6). Interestingly, nutrient concentrations within the Blackmore River estuary sampled during the wet season period of peak flow are lower than those recorded during the dry season. It is likely that water coming into the harbour from early rainfall events are high in nutrients as they strip nutrients from the catchment, but that after this period the riverine water is little more than rainwater, with correspondingly lower nutrient concentrations. Highest concentrations of nitrogen species within the Blackmore River estuary were found during the dry season of 2003, and may originate from remineralization of organic material originating from relatively high concentrations plankton (Figure 5) occurring during a period of long residence time (see below). In contrast, at Station 6 concentrations of nitrogen species were highest during the wet season. This most probably indicates the importance of sheet runoff as a source of new nitrogen in Darwin Harbour, as rainfall washes nutrient from inside the mangrove ecosystems and from built environments into the harbour. Runoff from built environments in the area can have NO_3 concentrations as high as 90 $\mu M\ l^{-1}$ (Schult, 2004). Alternatively, the higher concentrations of NO_2 in the wet season may be evidence of bacterially mediated denitrification processes occurring in low light (heavily overcast) conditions, but these processes are more typical of de-oxygenated waters, rather than the well-oxygenated conditions observed. There was little seasonal variation in the concentration of PO_4 in either the Blackmore River estuary or the harbour itself, but the concentrations were higher in the former.

2.3.3. Wet and dry season contrasts

Table 2 contrasts water quality parameters in the wet and dry seasons of 2003, during which our sampling periods coincided with extremes of salinity regime within Darwin Harbour.

Average daily input into the Middle Arm from the Blackmore River during February 2003 was 2,356 ML (Table 2) and resulted in upper reaches of the Middle Arm becoming fully fresh (Table 2, Figure 3). In the estuary, concentrations of suspended sediment were as high as 73 mg l^{-1} in the wet season (Table 2), due probably to the reduced flocculation of fine material in the fresh waters and consequent persistence of suspended sediment. On a harbour-wide basis, the tidal resuspension of mangrove mud is likely to be the largest contributor to turbidity.

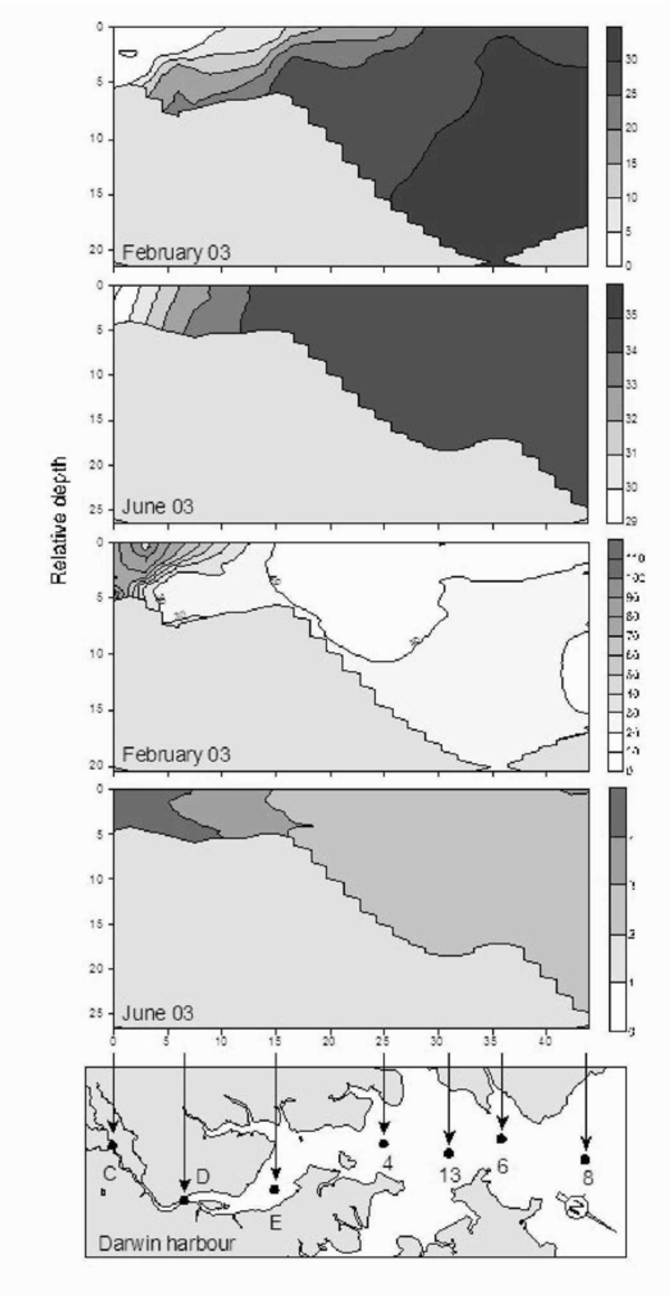

Figure 3. *Sections through Darwin Harbour in wet and dry seasons. Data derived from CTD (Seabird SBE 19plus) casts, at the stations indicated in the lower panel.*

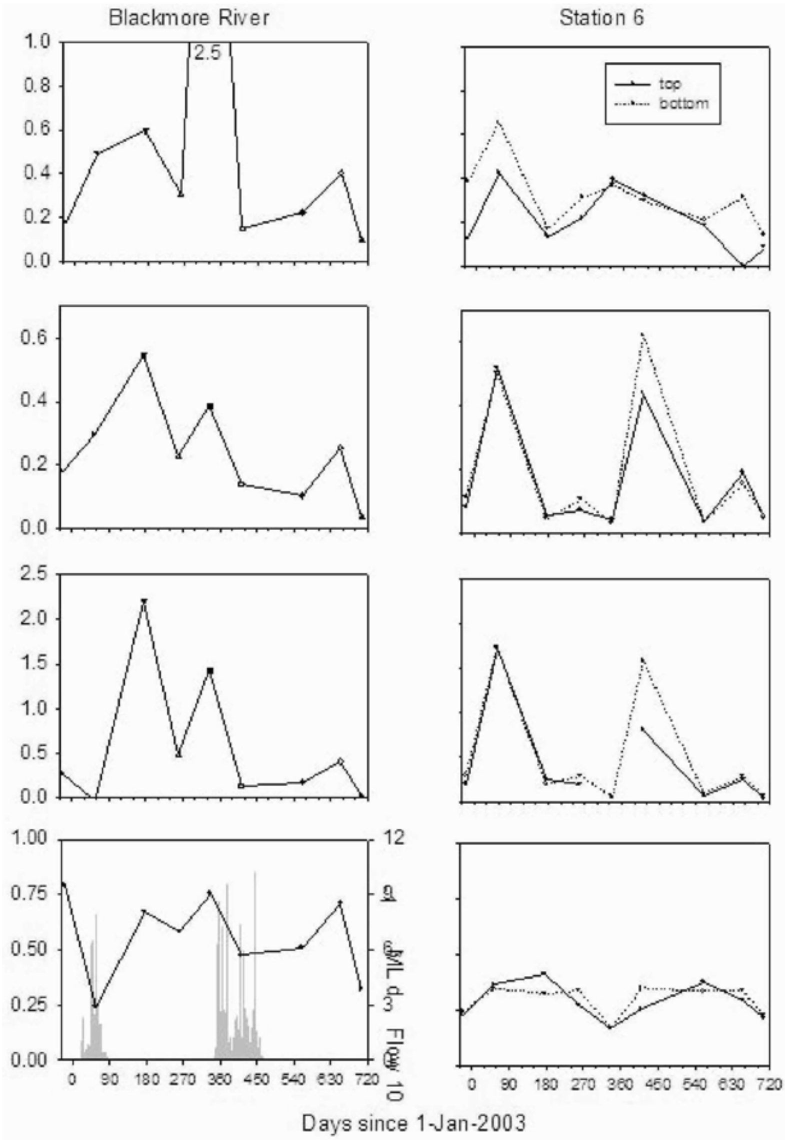

Figure 4. *Dissolved nutrient concentrations at two stations in Darwin Harbour during 2003 and 2004. The left panels refer to Station C, at the tidal limit in the Blackmore River (see Figure 3), and the right panel refers to Station 6, in the mouth of the harbour. Blackmore River flow data (lower left panel) is derived from a DIPE gauging station at Tumbling Waters.*

Table 2. *Means and ranges of water column variables in wet and dry seasons. The data represent means ± standard deviation of near surface and near bottom water samples at 9 stations throughout Darwin Harbour. Analytical methods and further description of sites are available at http://www.aims.gov.au/pages/research/darwin/dhwqmp/dhwq-003-01.html*

	Feb 2003 (Wet)	**June 2003 (Dry)**
Elizabeth River flow, February (ML d^{-1})	1693 ± 1823 163-8901	0 0
Blackmore River flow, February (ML d^{-1})	2356 ± 2319 189-7883	0 0
Temperature (°C)	28.82 ± 0.60 26.81 – 29.71	25.11 ± 0.31 24.38 – 25.62
Salinity	23.47 ± 9.71 0.09-35.61	34.29 ± 1.67 28.52-35.61
Suspended solids (mg l^{-1})	19.93 ± 18.94	5.62 ± 2.30
pH	8.12 ± 0.40 7.36-8.49	8.05 ± 0.16 7.78-8.24
Dissolved Oxygen (mg l^{-1})	5.85 ± 0.42 5.52-6.68	6.03 ± 0.35 5.35-6.55
NH_4 (µM)	0.90 ± 0.79 0.26-3.46	0.27 ± 0.20 0.07-0.77
NO_2 (µM)	0.53 ± 0.16 0.18-0.74	0.08 ± 0.13 0.01-0.56
NO_3 (µM)	1.54 ± 0.63 0.07-2.46	0.29 ± 0.53 0.04-2.30
Dissolved organic nitrogen (DON) (µM)	11.43 ± 4.10 6.14-24.19	10.41 ± 2.81 5.43-19.31
Total Dissolved Nitrogen (µM)	14.41 ± 4.09 9.03-26.93	11.06 ± 3.06 5.78-19.63
Particulate nitrogen (PN) (µM)	2.35 ± 1.34 1.14-6.22	1.31 ± 0.53 0.72-2.9
PO_4 (µM)	0.38 ± 0.10 0.15-0.58	0.35 ± 0.11 0.22-0.68
Dissolved organic phosphorus (DOP) (µM)	2.00 ± 1.26 0.37-5.0	0.51 ± 0.30 0.18-1.86
Total Dissolved Phosphorus (µM)	2.38 ± 1.25 0.66-5.27	0.86 ± 0.32 0.47-2.08
Particulate phosphorus (PP) (µM)	0.24 ± 6.54 0.15-0.35	0.13 ± 7.16 0.08-0.35
Silicate (µM)	16.42 ± 10.70 5.79-36.05	12.02 ± 13.28 4.89-56.49
Chlorophyll *a* (chl) (µg L^{-1})	0.77 ± 0.42 0.12-1.99	0.89 ± 0.5666 0.41-2.55
>10 Chlorophyll *a* (µg L^{-1})	0.17 ± 0.13 0.01-0.47	0.35 ± 0.25 0.15-0.97
Heterotrophic Bacteria (*1000)	153.71 ± 98.45 48.85-355.1	106.49 ± 57.76 42.65-236.07
Zooplankton biomass (mg m^{-3})	96.35 ± 158.90 27.14-674.61	62.13 ± 30.07 23.2-106.02
Zooplankton abundance (*1000)	33.49 ± 17.24 20.42 – 72.16	41.08 ± 37.71 13.09 – 136.20

There was little seasonal variation in pH – but this was lower by 0.4 than that recorded by Padovan (1997). Dissolved oxygen was slightly higher in the dry season, probably because of the lower temperature, and was in excess of 80% of saturation at all times. Overall, the largest nitrogen pool is dissolved organic nitrogen (Table 2), which comprises 67% of the total water column nitrogen in the wet season and 83% in the dry season. Total dissolved nitrogen was highest in the wet season (14.4 μM) with most dissolved fixed nitrogen in organic form (79 – 94%). Nitrate is the principal form of dissolved inorganic nitrogen in both wet and dry seasons. Ammonia and oxidized forms of nitrogen were higher in the wet season by a factor of 3-5, though these concentrations were still low (NH_4 < 3.5 μM, NO_2 < 0.8 μM, NO_3 <2.5 μM). Consequently, the concentration of dissolved inorganic nitrogen exceeded that of particulate nitrogen (PN) in the wet season, but was less than half of PN during the dry season. Dissolved inorganic phosphorus concentrations were invariant, but dissolved organic phosphorus was on average 4-fold higher in the wet season than in the dry season (2μM vs 0.5μM: Table 2). Only 9-13% of the water column phosphorus pool was in particulate form. Particulate forms of nitrogen and phosphorus were both higher in the wet season, as might be expected from the higher catchment loads of suspended sediments and their persistence in estuarine waters. Model II regression (Riggs et al., 1978) indicates that chlorophyll *a* (hereafter chl) concentration accounted for 62% of the variation in PN, and 41% of the variation in particulate phosphorus (PP). Similarly, suspended solids accounted for 55% of the variation in PN, and 69% of the variation in PP. Together, these results indicate that most of the PN is derived from organic material, but that much of the PP is bound to detrital particles in the water column.

Our measurements are comparable to those measured in 1990 – 91 (Padovan, 1997), indicating there has been little change in the intervening 12 years. Wet season values of NOx -N recorded during the 1990 – 91 study at Station 6 were 0.027 mg l^{-1} (2.25 μM), which is comparable to our harbour-wide mean of 2.07. Similarly, chl was 1μg l^{-1} compared to our value of 0.80 μg l^{-1}. Comparison of other variables is confounded by differences in the analytical methods used. Furthermore, Padovan (1997) compared his measurements to the ANZECC guidelines (1992) and found that recommended guideline concentrations developed principally for temperate waters were inapplicable to Darwin Harbour.

Using the sum of the inorganic nitrogen species to calculate nitrogen availability, (i.e. assuming that most DON is not available to primary producers), our data (Table 2) indicates that the dissolved N: P ratio averages 8:1 during the wet season, approximately half of the expected Redfield N: P ratio of 16:1 (Redfield et al., 1963). This ratio is considerably lower in the dry season, when N: P = 1.8. It therefore appears that nitrogen is the limiting nutrient for phytoplankton growth in Darwin Harbour.

2.4. The pelagic ecosystem of Darwin Harbour

2.4.1. Phytoplankton

Phytoplankton standing stocks within Darwin Harbour, as indicated by the concentration of chl, do not vary greatly during the year: harbour-wide mean chl concentrations during the wet season were 0.77 μg l^{-1} and 0.89 μg l^{-1} during the dry season (Table 2). These values are not significantly different (t-test, $T_{(54)} = -0.94$, $p = 0.35$). A comparison of the spatial distribution of chl through the harbour (Figure 5) demonstrates that during the wet season, in high river flow, all the phytoplankton is washed out of the Blackmore River but that a bloom develops in the dry season. This may arise as a result of the higher residence time within the river (i.e. poor flushing) together with the greater availability of nutrients at that time of the year within the Blackmore River (Figure 4). Paradoxically, a greater proportion of this chl is in cells >10 μm during the dry season (39%) than during the wet season (22% - Table 2, Figure 5). This fraction of the chl is of interest because it normally comprises larger, more quickly growing cells such as diatoms, which because of their size are available for grazing by mesozooplankton such as copepods, which in turn are important prey times for juvenile fish. There is little difference in chl concentration at different depths in the water column, though in the wet season near-surface values were marginally higher (Figure 5) possibly because of light limitation of phytoplankton growth below the shallow euphotic zone.

Autotrophic picoplankton (cells < 2 μm) dominate primary production in the tropics (e.g. Furnas and Mitchell, 1987). Most autotrophic picoplankton was removed from the Blackmore River by inflow during the wet season, but there was a continuous increase in the numbers of both the prochlorophyte *Synechococcus* and pico-eukaryotic cells closer to the mouth of the harbour (Figure 5). Heterotrophic bacteria showed no such pattern, and were similar in abundance both spatially and seasonally.

During the dry season there was little difference in *Synechococcus* abundances spatially in the harbour, but pico-eukaryotes were more abundant within the Blackmore River.

2.4.2. Zooplankton

Mesozooplankton (>73 μm) showed a peak of abundance in the Blackmore River in the dry season. Abundances within the body of the harbour were similar throughout the year. In the Upper Blackmore River in February 2003, a zero abundance of zooplankton was recorded as a result of high riverine flow (Figure 5). Most (94%) of the mesozooplankton in Darwin Harbour were copepods, which were dominated by members of the calanoid family Paracalanidae (e.g. *Parvocalanus crassirostris*) and the cyclopoid family Oithonidae (e.g. *Oithona attenuata*, *O. nana*, *O. simplex*). These groups dominate inshore plankton in tropical Australia, and are common in mangrove systems (McKinnon and Klumpp, 1998). It is therefore not surprising to find the plankton composition within the Blackmore River in the dry season to be

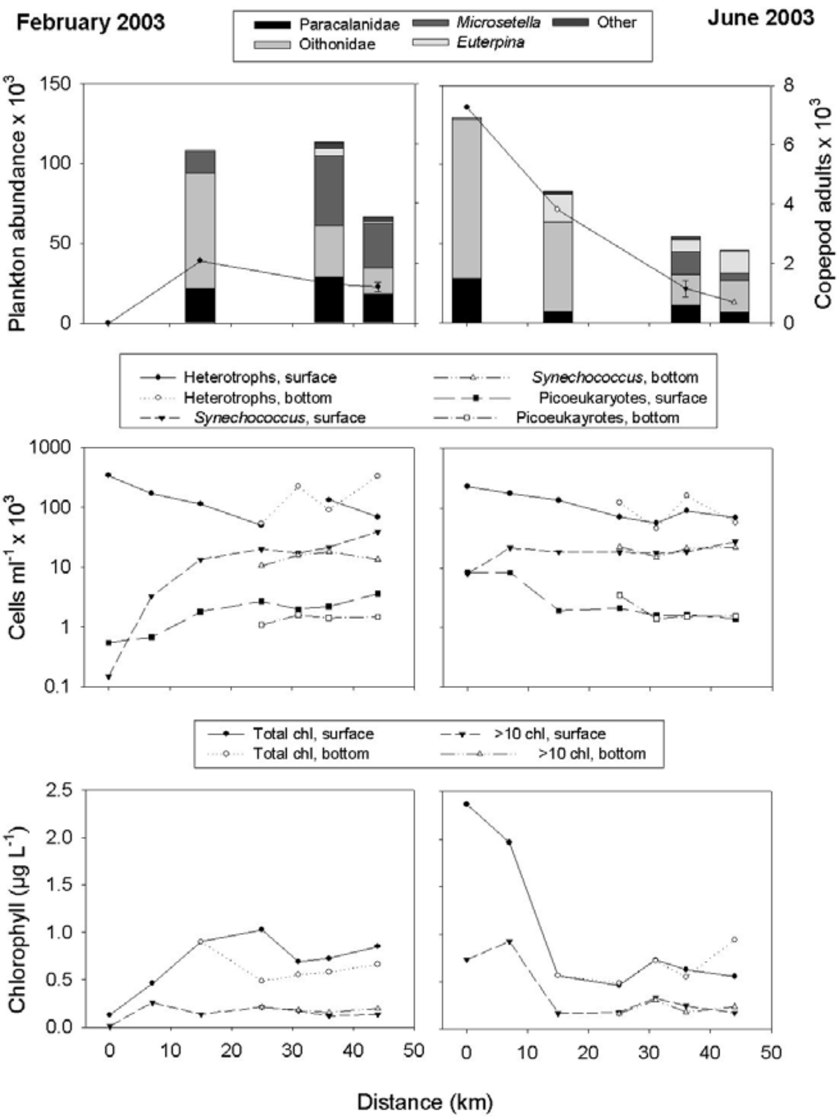

Figure 5. *Plankton distributions in Darwin Harbour along the Middle Arm transect, as in Figure 3. Field and analytical methods are given at http://www.aims.gov.au/pages/research/darwin/dhwqmp/dhwq-003-01.html*

wholly comprised of these organisms. In the main body of the harbour, particularly in the wet season, the harpacticoid copepod *Microsetella* becomes important. This genus is adapted to feeding on the cyanobacterium *Trichodesmium*, which commonly forms large surface slicks in the Timor Sea. *Trichodesmium* has the ability to fix atmospheric nitrogen, and may be an important additional nitrogen source to Darwin Harbour.

2.4.3. The mangrove ecosystem of Darwin Harbour

Mangroves fringe at least two-thirds of Darwin Harbour's foreshore, and occupy 27,350 ha. Ten vegetation associations have been identified and mapped (Brocklehurst and Edmeades 1996; Table 3). A total of 36 mangrove plant species occur in the harbour, constituting about half the world's mangrove species. The most common are *Rhizophora stylosa*, *Ceriops tagal, Sonneratia alba, Bruguiera exaristata, Avicennia marina* and *Camptostemon schultzii*. Research conducted on the mangroves in Darwin Harbour has recently been reviewed by McGuinness (2003).

The distribution of mangrove vegetation associations is primarily a function of elevation and foreshore geomorphology (Figure 6), though disturbances by cyclones and clearing affects a small area (<0.01%). Approximately 2% of the area of mangroves of Darwin Harbour has been reclaimed, with nearly 96% of the remainder zoned "conservation", which requires NT Government approval to change.

Two vegetation communities, *Rhizophora stylosa*/*Camptostemon schultzii* and *Ceriops tagal* forests (Table 3), comprise almost two-thirds of the mangrove area surveyed by Brocklehurst and Edmeades (1996). The litter fall of mangrove vegetation associations ranges from 4.2 to 11.2 t ha^{-1} yr^{-1}, with an area weighted mean of 6.5 t ha^{-1} yr^{-1} (Table 3). Litter fall is highest during the wet season and lowest during the dry season (Metcalfe, 1999; Comley, 2002).

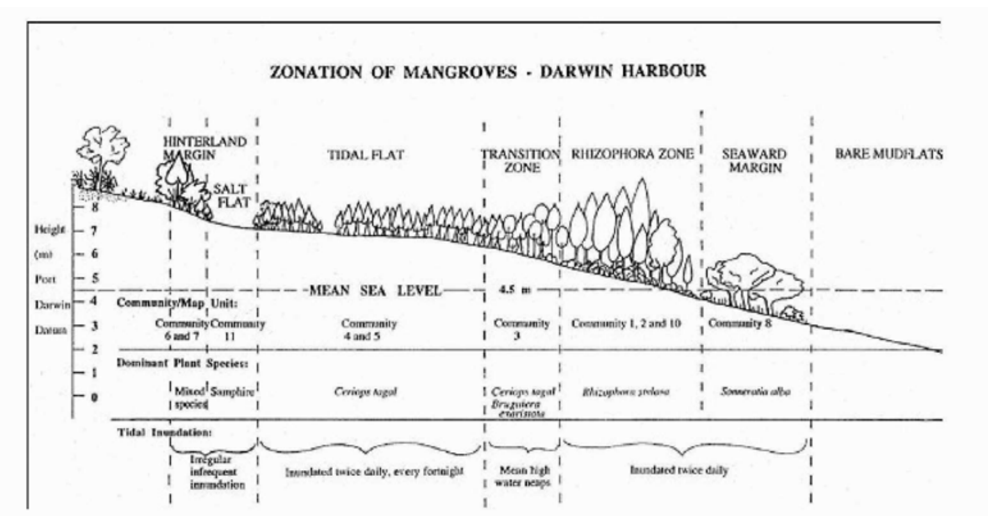

Figure 6. *Stylised zonation of mangroves in the main body of Darwin Harbour (Source: Brocklehurst and Edmeades, 1996; Metcalfe, 1999).*

Table 3. *The area of mangrove vegetation associations mapped by Brocklehurst and Edmeades (1996) and leaf litter fall (Metcalfe, 1999), between Charles Point and Sadgroves Creek. This area constitutes 75% of the total mangrove area of Darwin Harbour. Community codes refer to Figure 6.*

Community code	Description	Area (%)	Litter fall ($t\ ha^{-1}\ y^{-1}$)
	Mangrove Closed Forest		
1	*Rhizophora stylosa* closed forest/low closed forest (shoreline forest)	3.5	9.6
2	*Rhizophora stylosa/Camptostemon schultzii* closed forest (tidal creek)	31	8.4
3	*Rhizophora /Bruguiera/Ceriops* closed forest/open forest (transition)	3.9	5.5
4	*Ceriops tagal* low closed –forest (mid tidal flat)	42	4.2
5	*Ceriops tagal/Avicennia marina* low closed forest/open forest (high tidal flat)	4.7	5.4
6	Mixed species low closed forest/open forest (hinterland)	8.0	8.5
	Mangrove Woodlands/Open Woodlands		
7	Mixed species low woodland	1.5	6.1
8	*Sonneratia alba* woodland	5.1	11.2
9	*Rhizophora stylosa* low woodland (islands, rocky shores)	0.01	-
10	Low open woodland (low tidal mudflat)	0.13	-
Area weighted mean annual mangrove litter fall			6.5

The mangrove fauna of Darwin Harbour includes at least 60 fish species, 36 crustacean species and 31 species of molluscs (Northern Territory Government, 2002b). The dominant crabs in the mangroves are fiddler crabs (family Ocypodidae) and members of the family Grapsidae (McGuinness, 2003). A trapping program removes about 241 saltwater crocodiles (*Crocodilus porosus*) from the harbour each year (pers. com. Tom Nichols, DIPE). Homalopsine marine snakes capture fish and crustaceans either in the water or on exposed mud banks (Whiting, 2003), with the most common being the bockadam (*Cereberus rynchops*) and the white bellied mangrove snake (*Fordonia leucobalia*).

The mangrove avifauna of the NT is one of the richest in the world, with 10 mangal dependent taxa (Noske, 2003) and many species that visit this biome. Birds are important pollinators for several mangrove plants, notably species of *Bruguiera*, and seed dispersers of mangrove mistletoes. Small insectivores dominate the avifauna of the harbour mangroves, with many species nesting predominately on or adjacent to salt flats (Noske, 2003). Bats, especially the black flying fox (*Pteropus alecto*) and less frequently the northern blossom bat (*Macroglossus minimus*), are also common visitors to the mangroves.

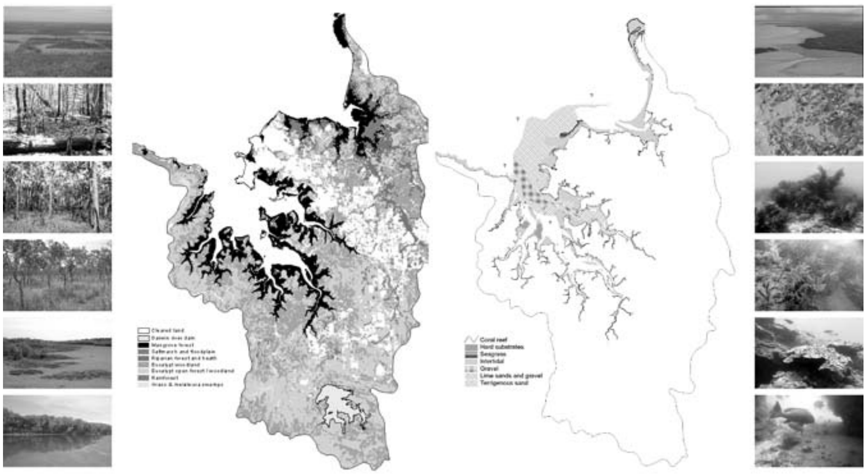

Figure 7. *Terrestrial (left) and marine (right) habitats in the Darwin Harbour region. Photos: left top to bottom:* ***Habitat fragmentation and clearing, Riparian forest, Eucalypt woodland, Eucalypt open woodland/forest, Saltmarshes, Mangroves; right, top to bottom: large intertidal mudflats, seagrass (*Halophila decipiens*), Algal beds (wet season), Sponge garden, coral reef, transition zone between hard and soft substrates.*

2.5. Benthic ecosystems of Darwin Harbour

2.5.1. Hard substrates

Hard substrates provide a three-dimensional structure for a wide range of flora and fauna and are characterised by high species diversity, especially in the case of coral reefs. Hard substrates occur mainly in the upper reaches of Darwin Harbour, but are poorly mapped (Figure 7). Community structure in these habitats follows a distinct pattern of zonation (Smit, 2003 and references therein). Intertidal zonation is mostly determined by the period of exposure during low tides. In general, the upper intertidal areas are bare with a few hardy species of barnacles and oysters. The intertidal/subtidal areas are a continuation of the subtidal flora and fauna, but with a less diverse biota comprising hard and soft corals (e.g. *Singularia* spp, *Sacrophytum* spp. and *Lobophytum* spp.), sponges (e.g. *Dysidea* spp. *Sigmadocia symbiotica*, *Adocia* spp., *Mycale* spp.), molluscs (e.g. chitons), crustaceans, anemones, and many species of macro-algae (e.g. *Padina* spp.). From the intertidal/subtidal interface to the subtidal zone, hard substrate zonation is mostly determined by the combined effect of exposure rate, light intensity, current strength, wave action and turbidity.

These habitats can be classified according to their dominant community assemblages (e.g. macro-algae, coral and non-coral dominated reefs). Macro-algal dominated reefs are located often on reef crests, and a few metres either side of the low water mark, often in association with coral or sponge dominated reefs. Algal composition is highly seasonal (Ferns, 1996) and apparently regulated by the amount of time the algae is exposed during spring low tides. During the “build up” season (October – December) turf algae dominates, whereas during the dry season the larger macro algae are more prolific. The most dominant species are *Padina australis, Laurancia* cf. *papilosa, Splatoglossum aperum,Dictyopteris sp.,Saragassum sp.* and *Chondrococcus hornemanni*.

Corals are surprisingly well represented in Darwin Harbour (123 species are known) given the low light availability, high sedimentation rates, and lack of suitable substrate. Darwin Harbour does not have true coral reefs, but coral communities dominated by the families Favidae, Acroporidae and Poritidae grow on a substrate of sedimentary rock at e.g. Channel Island, Platters Rock, East point and Nightcliff reefs (Figure 7). Four additional species (*Leptastrea transversa, Montipora aequituberculata, Acropora tenuis* and *Acropora valida*) are found on the extensive coral reefs at Gunn Point (Wolstenholme et al., 1997). These communities are sparsely distributed in lower intertidal to high subtidal zones to a depth of 5 - 10 m, in areas characterised by strong currents. Non-coral reefs dominated by sponges (56 spp.), bryozoan, ascidians, and hydroids (63 spp.) occur in areas where hard substrate is available and corals cannot establish because of poor light quality or high sediment loads. These habitats occur primarily within the lower-intertidal and subtidal areas and are patchy by nature, often forming a transition zone between hard substrates and the subtidal mud-dominated substrates. The local popular dive site East Point Sponge Gardens is a good example of such a transition zone.

2.5.2. Soft substrates

Soft substrates cover at least 80% of the marine environment of Darwin Harbour. The spatial extent of muddy, sandy and coarse sediments is often difficult to determine because of gradual transitions between sediment types. There has been little research effort towards describing floral and faunal community structure in relationship to sediment characteristics in these habitats.

Intertidal soft substrates have significant conservation value because they support wader-birds, Flatback turtles and dugongs. These areas mainly consist of muddy to sandy-mud sediments dominated by infauna such as crustaceans and polychaete worms, in particular capitellids and nereidids (Hanley, 1988; Smit et al., 2000). These areas generally have low species diversity, but high population numbers. Seagrass meadows are not well developed in Darwin Harbour, due to the large tidal range and high turbidity, with the only known one at the Casuarina Coastal Reserve. Only three of seven species of seagrass occurring in the NT are known from this area: *Halodule uninervis*, *Halophila ovata* and *Halophila decipiens*. Sightings of dugong in Fannie Bay and in the upper arms and tributaries of Darwin Harbour indicate that seagrass is likely to be more widespread than is currently recognised. Casuarina Coastal Reserve is also an important nesting beach for the endangered Flatback turtles.

Coastal salt marshes or samphires are found amongst elevated areas within the mangroves or between mangrove and terrestrial habitats. These comprise approximately 450 ha of the Darwin Harbour coastal vegetation types, but are small and isolated in comparison with the large extensive salt flats in Shoal Bay, just east of Lee Point. These habitats are characterised by large sparse/bare areas with small pockets of vegetation (7-10 plant species). Microphytobenthos and some macro-algae drive productivity in these areas and are likely to play a major role in the carbon balance of the harbour (see below), in addition to playing an integral role as a nursery and feeding habitat for many invertebrates and fish.

2.6. Fish and Fisheries

The fish community of Darwin Harbour is faunistically similar to that of the southern Great Barrier Reef, but is comparatively depauperate in coral-associated species. There are 415 species of fish, from 95 families in Darwin Harbour (Larson and Williams, 1997). The dominant four families, the Gobiidae (66 spp.), Apogonidae (21 spp.), Syngnathidae (18 spp.) and Pomacentridae (12 spp.) are all coral reef families.

In the mangrove forests that form a large proportion of the harbour margins, the most abundant families are the Ariidae, Mugilidae, Clupeidae and Engraulidae. Mangroves provide habitat for juveniles of most of the fish species commonly harvested by recreational and indigenous fishers, such as trevallies (*Caranx* spp.), mackerel (*Scomberomorus semifasciatus*), salmon (*Eleutheronema tetradactylum* and *Polydactylus macrochir)*, grunter (*Pomadasys kaakan*) and barramundi (*Lates calcarifer)* (Martin, 2005).

Most fish sampled within Darwin Harbour mangrove systems had full guts (Martin, 2005), suggesting that they were actively feeding in these habitats and implying that mangroves provide an important source of food for the fish community of the harbour as a whole. Mangroves provide a broad dietary spectrum, with shrimps, fish and crab the important prey categories (Martin, 2005). In particular, sesarmid crabs appear to be an important trophic intermediary between mangrove trees and fish communities, as they are prey items for many fish species and are important in decomposing mangrove litter fall (Robertson et al., 1992). Crab zoea are also an important component of the diet of juvenile fishes in mangrove systems (Robertson et al., 1988).

Darwin Harbour provides a venue for commercial, recreation and subsistence fishing. Commercial mud crabbing and gillnetting in Darwin Harbour were closed in 1991 and 1998, respectively. The remaining commercial fisheries (coastal net, coastal line and bait net fisheries) are able to fish Darwin Harbour, but there is currently very little commercial activity by commercial operators in order to avoid conflict with the recreational fishing sector.

The recreational fishery in Darwin Harbour is substantial (Coleman, 1998, 2004). In 2000, 37% of Darwin residents spent some of their time fishing and one in every five resident households owned a pleasure boat used at least partly for recreational fishing (Coleman, 2004). As a result, the NT government is actively engaged in furthering facilities and opportunities for the recreational fishing sector, including increasing the number of boat ramps, improving boat ramps so they can be used during all tides, and increasing the number of artificial reefs in the Harbour.

The total number of hours fished annually in Darwin Harbour in 2000 was estimated to be 540,481 hours, which equates to 30% of all recreational fishing effort in the NT. In 2000 over a half million aquatic organisms were caught by fishers in Darwin Harbour, which primarily consisted of 374,842 finfish and 103,415 mud crabs. Most of the finfish catch were tropical snappers (Lutjanidae, 25%), small baitfish (9%), mullet (Muglidae, 4%) and barramundi (Centropomidae, 5%). The most prized fish species are snappers, mackerel (Scombridae), barramundi, jewfish (Sciaenidae) and mudcrabs. Not all of the fish which are caught are harvested, and harvest proportions differ from species to species. For example, small baitfish, mullet, and trevally have harvest rates of over 80%, in stark contrast with the harvest rates for sharks/rays (13%) and catfish (3%). In 1995, the recreational fishery caught 64,211 snappers; 15,804 jewfish and 13,356 mackerel (Coleman, 1998), conservatively estimated to be ~210 t of fish (Smit, 2003). The ecosystem impact of the harvesting of these organisms from Darwin Harbour is unknown and requires investigation (Fishery Status Reports, 2003).

Besides recreational fishing for local residents, recreational fishing plays an important role for tourism operators. Coleman (2004) estimates that 21% of the recreational fishing effort in Darwin Harbour is by visitors. The annual expenditure directly attributable to recreational fishing in the Darwin region is almost $AUS 22 million (Henry and Lyle, 2003). Although there are no data specific to indigenous harvest in Darwin Harbour, a recent survey of indigenous harvest across northern Australia in 2000 indicates that the total NT indigenous harvest was over half of the total recreational harvest for the whole of the Northern Territory (Coleman et al., 2003). The most abundant species of fish harvested were mullet, catfish, snappers, bream, barramundi and mud crabs. However, based on the overall catch composition, indigenous fishers had a greater reliance on non-fish species. Species targeted by indigenous fishers also included species protected from harvest by non-indigenous Australians, which include crocodiles, turtles and dugong.

2.7. Biodiversity of Darwin Harbour

Darwin Harbour is one of the few regions in NT waters that are reasonably well described in terms of number of species. The Museum and Art Galleries of the Northern Territory has an extensive database and collection for invertebrate and vertebrate marine fauna, but the marine flora has limited representation in the NT Herbarium's database. Although taxonomists believe that only a very small proportion (e.g. 10%) of the fauna has been described, the total number of species in Darwin Harbour most likely exceeds 3,000 species (Hanley, 1988), summarised in Table 4.

Biodiversity hot spots can be identified using surrogates, such as substrate type and physical parameters. In general terms, subtidal environments are more diverse than intertidal substrates. High species diversity occurs in hard substrate habitats highly three-dimensional and high-energy environments such as that of Channel Island. Conversely, areas of low current and low relief (e.g. intertidal and subtidal mud flats) have low diversity, but these habitats play an important role in the trophic structure of the ecosystem. Consequently, the identification of priority areas for protection cannot be based solely on biodiversity issues but needs to take account of trophic and biogeochemical processes within the harbour.

Table 4. *Species diversity: number of species per taxa in Darwin Harbour region (Adapted from Hanley, 1988; Hanley et al., 1997; Russell and Hewitt, 2000; Smit et al., 2000; Smit, 2003).*

Taxa	no. of species	Comments
Algae	110	These numbers only reflect macro algae, poorly researched taxa.
Seagrasses	3	No species surveys have been conducted, however surveys in Bynoe Harbour and Fog Bay have resulted in the collection of 5-6 species.
Sponges	56	Only approx. 10% of the sponge fauna has been described (56 species). Sponge fauna in NW West Australia, including Darwin Harbour, is unique for Australia.
Cnidarians		Low diversity, poorly represented.
soft coral & seawhips	50-65	Surprisingly rich considering the high turbidity and little substrate available to colonise, but diversity is lower than E and W coasts of Australia.
corals	123	
hydroids	63	
Marine worms	600+	Highest diversity on subtidal reefs.
Crustaceans	1,000+	Estimated number of species.
decapods	360	Includes crabs, prawns, shrimps and lobsters.
other	24	Isopods.
Molluscs	924	Diversity is considered low when compared with similar areas e.g. Port Essington, Cobourg Peninsula.
Echinoderms	60 - 117	Echinoderm fauna is poor with the exception of Brittlestars which are associated with muddy habitats; soft substrates, considered species rich.
Fish	415	High diversity for Syngnathid species, otherwise the species composition reflects the available habitats in the harbour and is comparable with neighbouring areas.

3. DARWIN HARBOUR IN CONTEXT

3.1. Primary production, nutrient demand and assimilative capacity

Primary production in Darwin Harbour has three main contributors: mangroves, phytoplankton, and the microphytobenthos associated with the mud flats. Unfortunately no data is yet to hand on the production of the latter. A first order estimate of mangrove production can be made on the basis of leaf litter fall from the study of Metcalfe (1999). Assuming an annual production of mangroves on the basis of litter fall of 6.5 tonnes ha^{-1} yr^{-1}, and that mangroves cover a total area of 27,350 ha in the harbour (Table 4), annual litter fall in the harbour is 177,775 t DW. If litter fall represents 30% of total mangrove production and the carbon content of litter is 48% (Alongi et al., in press), the production of Darwin Harbour mangroves amounts to ~284,000 t carbon. Taking the N and P content of green leaves from arid-zone environments in North Western Australia (1.36% and 1.37×10 % of DW respectively; Alongi et al., in press), this translates to an annual N demand of 2,418 t and P demand of 244 t from leaf production alone.

The high turbidity of Darwin Harbour results in the water column euphotic zone often being restricted to <10 m. Pelagic primary production, therefore, is likely to be light limited. In spite of this, the harbour is strongly autotrophic (i.e. Photosynthesis: Respiration (P: R) > 1 - Table 5) with high rates of photosynthesis occurring in the

Table 5. *Water column production and respiration rate in Darwin Harbour. CR, community respiration; NPP, net community production; GPP, gross community production (=CR+NPP); P: R; the ratio of GPP: CR. Production and respiration of natural plankton communities were calculated from measurements of oxygen flux in 125ml flasks incubated under simulated ambient conditions at 4 depths through the water column (production) and in the dark (respiration). Dissolved oxygen concentration was determined with an automated precision Winkler titration system developed at the Oceanographic Data Facility, Scripps Institution of Oceanography, University of California, San Diego. Oxygen data was converted to carbon-specific rates assuming a photosynthetic quotient of 1.2 and a respiratory quotient of 0.8 (Laws 1991).*

Location	Chl	CR	NPP	GPP	P:R
	$\mu g\ l^{-1}$	$mg\ C\ m^{-2}\ d^{-1}$			
February 2004					
Stn 6	1.24	1159	1700	2860	2.47
Stn 6	0.57	1546	1586	3132	2.03
Stn 4	0.72	869	957	1826	2.10
Wickham Pt	0.81	775	4416	5191	6.70
MEAN	**0.83**	**1087**	**2165**	**3252**	**3.32**
October 2004					
Stn 6	0.85	795	719	1514	1.90
Stn 13	0.91	443	940	1383	3.12
West Arm	1.80	593	570	1163	1.96
East Arm	1.51	504	1400	1904	3.78
Middle Arm	1.22	605	1264	1869	3.09
MEAN	**1.26**	**588**	**979**	**1580**	**2.77**

surface layers of the water column and compensating for the community respiration occurring throughout the water column. Pelagic primary production in the wet season is at times very high: gross production can be in excess of 3 g C m^{-2} d^{-1}. These high rates do not necessarily accord with the standing stocks of phytoplankton in the water column – for example the highest rate of production we observed (4.4 g C m^2 d^{-1} net production) occurred near Wickham Point in February 2004, despite the total chl concentration being moderately low (0.8 μg l^{-1}). Mean net production in

February 2004 was 2.2 ± 0.8 S.E. g C m^2 d^{-1}. During the dry season primary production is lower, with a net mean production of 1.0 ± 0.2 S.E. g C m^2 d^{-1}.

Our data indicate that net pelagic primary production in Darwin Harbour is ~183 mmol C m^{-2} d^{-1} in the wet season, and ~83 mmol C m^{-2} d^{-1} in the dry season. Based on Redfield ratios of C: N: P of 106:16:1 (Redfield et al., 1963), this corresponds to nitrogen demands of 28 and 13 mmol N m^{-2} d^{-1} respectively, and phosphorus demands of 1.7 and 0.8 mmol P m^{-2} d^{-1}. Assuming a harbour-wide average water depth of 15 m, the calculated depletion time for depth integrated DIN stocks is 1.6 d in the wet season, and 0.8 d in the dry season. Similar calculations based upon PO_4 stocks results in depletion times of 3.3 d in the wet season and 6.8 d in the dry season. The difference in depletion times based upon N and P is further evidence of nitrogen limitation of phytoplankton production in Darwin Harbour.

Assuming a 3 month wet season and the area of the harbour (at low tide) to be 660 km^2, total phytoplankton primary production is at least 312,000 t C. Annual demand fluxes on the basis of phytoplankton production alone are 56,000 t N and 7,700 t P. Consequently, the estimated total annual anthropogenic loadings of 797 Tonnes of N and 101 Tonnes of P (Table 1) represent a tiny proportion (<1.5 %) of the annual nutrient demand to fuel primary production within the harbour.

The waters of Darwin Harbour are by nature highly turbid, leading to a public perception of poor ecosystem health – clear blue waters usually being held in greater esteem. Nixon (1995) defined a trophic classification system (i.e. degrees of "eutrophication") for marine systems in terms of organic carbon supply. Our measurements of net primary production within the body of Darwin Harbour are of the order of 2 g C m^{-2} d^{-1} during the wet season, and 0.8 g C m^{-2} d^{-1} during the dry season. Extrapolating these data on the basis of a 3 month wet season, the supply of carbon to Darwin Harbour from primary production alone should be in the order of 400 g m^{-2} y^{-1}, which would make Darwin Harbour "eutrophic" according to this classification scheme. However, mangrove ecosystems such as Darwin Harbour have naturally high carbon loadings, and the unpalatable term "eutrophic" does not necessarily carry any connotations of pollution. Nutrient concentrations in Darwin Harbour are low, and are similar to those found in the "oceanic" eastern part of Moreton Bay (c.f. Dennison and Abal, 1999). In fact, NO_3 concentrations in the dry season are similar to those of inshore waters of the Great Barrier Reef and PO_4 concentrations are only about double (c.f. Table 3 of Furnas, 2003), a small difference in view of the disparate oceanic regimes. N: P ratios in Darwin Harbour are typically ~8, well below the Redfield ratio of 16, indicating nitrogen limitation of primary production and a system with unsaturated assimilative capacity for absorbing nitrogenous wastes. By way of comparison, in heavily polluted environments these ratios are much higher. For instance, the N: P ratios in the Pearl River estuary, servicing the harbours of Shanghai and adjacent to Hong Kong are as high as 300 (Huang et al., 2003), and in Tokyo Bay can approach 800 (Yu et al., 1995).

3.2. Managing the future of Darwin Harbour

A paper entitled "Management Issues for the Darwin Harbour Region" prepared by the Darwin Harbour Advisory Committee convened by the NT government, identified a number of threats to the ecosystems of Darwin Harbour. Recurrent themes throughout were overfishing, habitat loss, increased turbidity and deterioration of water quality because of the introduction of nutrients or contaminants. The introduction of pests, primarily from foreign vessels visiting the port, was also a concern and one for which there is some precedent in an outbreak of black-striped mussels in Darwin marinas in 1999 (Marshall et al., 2003). The large number of fish removed from the harbour each year is of concern not only because of the potential changes in community structure caused by removal of apex predators, but also because of cascading trophic ecosystem effects. Dredging and consequent spoil disposal are also appreciable, through short-term threats, given the scale of developments currently in progress – for example the LNG pipeline to Wickham Point, and the substantial marina and harbour expansion projects. Primary production is already apparently light-limited; increased turbidity originating from dredging activities will further limit primary producers within the harbour.

In 2004 the NT Government implemented a Darwin Harbour Plan of Management to provide an integrated framework for management of the region's natural resources and to protect community uses and values. However, resource managers and scientists currently have a poor understanding of the biogeochemical, physical and oceanographic processes that drive the Darwin Harbour ecosystem, and are therefore limited in their ability to make informed decisions regarding the sustainable management of the Darwin Harbour ecosystem. One of the key challenges for the near future is to develop a coordinated and focussed approach to the design of monitoring and research programs to alleviate this data shortfall. For Darwin Harbour to avoid the degradation observed in other regional harbours, it is to be hoped that these plans are successful.

4. ACKNOWLEDGMENTS

We wish to thank Dr. Eric Wolanski for the invitation to contribute to this volume. Armando Padovan, Rod Metcalfe and Lyle Thompson (DIPE) helped with the collection and interpretation of water quality data, and the crew of the R.V. Cape Ferguson facilitated our work in Darwin Harbour in 2004. The manuscript benefited from discussions with Anne Coleman (Department of the Chief Minster, NT Government) and Drs. Dan Alongi and Miles Furnas (AIMS).

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CHAPTER 26

HYDRODYNAMICS OF DARWIN HARBOUR

DAVID WILLIAMS, ERIC WOLANSKI, AND SIMON SPAGNOL

1. INTRODUCTION

Historically, mining, agricultural and urban developments in Australia have proceeded without fully understanding their environmental impact on estuaries and coastal waters. The management focus was on measuring environmental degradation and assessing the need for remedial measures. These remedial measures were often economically and socially expensive, and not the optimum way to achieve sustainable resource development. Science-based solutions need to be developed for assisting resource development and urban expansion without compromising the sustainability of living resources and the quality of urban and traditional life. Northern Australia provides this opportunity because while large-scale developments are planned it currently has a low urban population density, it is resource rich and still has a relatively pristine marine environment.

Darwin (12^{o} 35' South and 131^{o} 31' East; see Figure 1) is the capital of the Northern Territory, Australia, and is located around the eastern shores of Darwin Harbour. Annual average rainfall is 1500 mm and this rain falls mainly between the months of December to April. The remainder of the year is dry. From May to October the dry, south easterly trade winds prevail, and in the wet season the wind blows moisture-laden air from the northwest. The Northern Territory waters may experience several tropical cyclones per year. Occasionally, a cyclone hits Darwin; in 1974 tropical cyclone Tracy caused devastation in Darwin when most of the cities housing and infrastructure was destroyed or badly damaged.

To date little has been done to understand the link between hydrodynamics, sediment and nutrient dynamics of the harbour in order to assist with the management of infrastructure developments. Some modeling of dredge disposal has been done to track the fate of dredge spoil dumping plumes. Salinity may play an important role in resuspending sediment due to dispersion when freshwater runs over the top of salinised muds.

After extensive community consultation the NT Government has declared beneficial uses for Darwin Harbour as a basis for the protection of its aquatic habitat and its cultural, aesthetic and recreational values. The NT Government has also stated that the Harbour will be managed to ensure that the majority of the mangrove

E. Wolanski (ed.), The Environment in Asia Pacific Harbours, 461–476.

communities will be conserved. The aim is therefore to achieve environmental sustainability whilst allowing responsible economic development. However, the scientific data are missing to facilitate the system of review and regulation to maintain beneficial uses and the ecological services provided by the harbour (eg fishing and a healthy estuary). This chapter provides the first peer-reviewed publication on key physical processes controlling the hydrodynamics, the flushing rate, and the sedimentation of Darwin Harbour, and, as demonstrated in the previous chapter by McKinnon et al. in this book, these physical processes play a major role in controlling the health of Darwin Harbour waters.

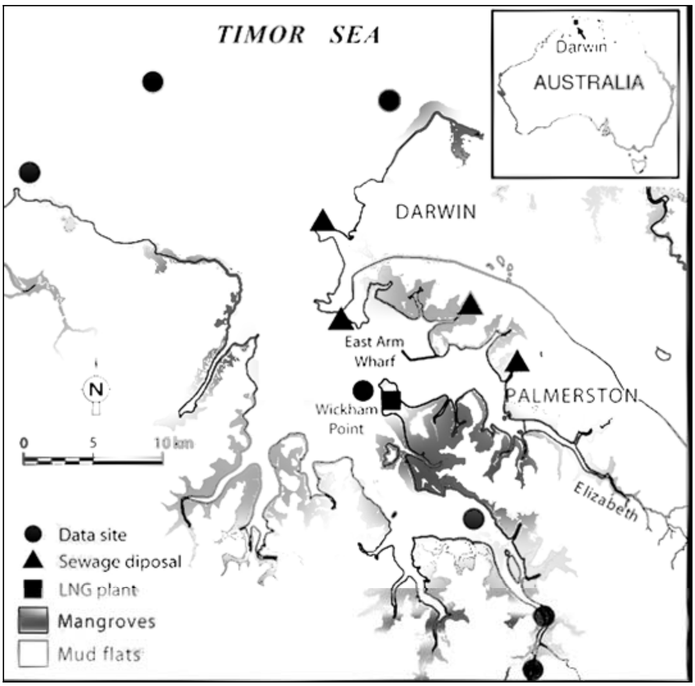


Figure 1. *General location map and a map of Darwin Harbour showing the mangroves, intertidal mud flats, oceanographic data sites, sewage discharge outfalls, and major harbour infrastructures. East Arm is the estuary of the Elizabeth River. Middle Arm is the southernmost estuary (marked by two data sites in this figure) that drains the Blackmore Estuary. West Arm is the estuary to the west.*

2. INFRASTRUCTURE

Darwin Harbour has always been a trading port albeit one of low traffic. The infrastructure has grown steadily especially since the mid 1990's (see Table 1).

In 1942 Darwin Harbour was bombed by the Japanese Imperial Forces and many shipwrecks still lie on the bottom of the harbour. Until the mid 1990's, the main shipping traffic was limited to live cattle exports and the export of manganese and some other minerals. Since then the major wharf facility has been moved to the new East Arm Wharf (construction began in 1994). East Arm Wharf is currently being expanded to provide increased berthing facilities and storage for bulk oil supplies. Previously the harbour had one marina facility. Another commercial marina was constructed in 1985. Four marinas have been constructed in recent years for recreational vessels and a further one is under planning and construction will begin in mid 2005. In 2004 the Darwin to Alice Springs railway was completed and the East Arm Port expanded giving emphasis for Darwin Harbour to continue as a main trading port in the Australasia region. Construction also began on a Liquid Natural Gas processing plant and pipeline for gas delivered from the Timor Sea.

Table 1. *Infrastructure developments.*

Fort Hill – Stokes Hill Wharves	1880
Sadgrove Creek barge facilities	1970
Naval Base Marina	1977
Sadgroves Creek Basin	1985
Cullen Bay Marina	1992
East Arm Wharf	1994
Bayview Marina	1997
Tipperary Waters	1999
Alice Spring – Darwin Railway	2004
Wickham Point LNG Facility	2004
Waterfront development	2005

In addition to the infrastructure developments Darwin Harbour has several sewage treatment plants servicing about 100,000 people and discharging into the main harbour at a number of points shown in Figure 1. Most of the population lives around the coastal zone. Therefore the majority of runoff from the residential and industrial areas will enter the harbour waters. Planned expansion of residential areas is to occur in the upper parts of the harbour.

There have been several aquaculture projects initiated in Darwin Harbour. The first was a pearl oyster farm near Wickham Point (see Figure 1) and since then several prawn farms have been established in the middle arm (i.e. the Blackmore Estuary; for a detailed location map see Figure 1 of the previous chapter by McKinnon et al. in this book). It should be noted that the majority of the recent infrastructure developments, aquaculture and sewage disposal have been sited in the upper arms of the harbour which may the least flushed, hence the most susceptible to environmental degradation from human, land-based activities.

3. HARBOUR OCEANOGRAPHY

The harbour has 3 main zones, namely the outer harbour, the inner harbour and the 3 arms. East Arm has mainly mud beds with a large calcareous sand deposit upstream of the East Arm Wharf. Middle Arm has mainly mud beds but with some significant shoals of siliceous sands that fine seaward indicating a terrestrial origin. West Arm is mainly mud.

The harbour is 50 km in length measured from the outer boundary to the uppermost estuarine limit in Middle Arm. Infrastructure developments are significantly modifying the shape of the harbour domain through construction of wharves, marinas, jetties and the associated dredging. For example the width of the East Arm at high tide prior to the construction of the East Arm Wharf was 3.5 km, it is now 2.2 km.

Darwin Harbour has semidiurnal macro-tides, with a strong diurnal inequality; the highest astronomical tide is 8 m (Table 2). There is a major spring-neap fluctuation, the smallest neap tide is 0.3 m.

The tides propagate into the harbour as a progressive wave, with a 1.5 hr lag between the mouth and the upper reaches (Figure 2). The tides also become asymmetric as they propagate into the upper reaches where the ebb tide lasts about 1 hr longer than the flood tide. This is reflected in peak tidal currents being about 25% larger at flood tide than at ebb tides.

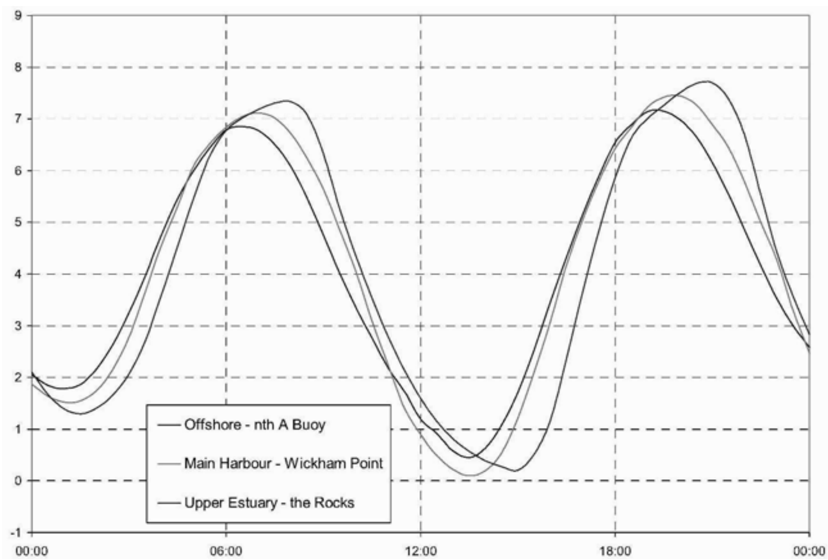


Figure 2. *Time series plot of the tides at three locations in Darwin Harbour during spring tides and in the dry season when runoff is negligible. Nth buoy=offshore, Darwin=Darwin city, Rocks= upper Blackmore Estuary. Note the increasing range and asymmetry of the tide as it propagates up-estuary.*

At spring tides, the peak tidal flux through the heads of the harbour is ~1.2 x 10^5 $m^3\ s^{-1}$. Runoff occurs mainly in the wet season through the three rivers draining into each of the arms (Figure 1). The largest measured cumulative runoff during an exceptional flood, was 10^3 $m^3\ s^{-1}$ or about 1% of the peak tidal discharge at the mouth.

A program of occasional current monitoring has been carried out since 1998, and in 2004 an extensive data set was collected on currents and sediment transport. The

Table 2. *Tidal planes (in metres above local gauge values) for Darwin Harbour.*

Highest Astronomical Tide (HAT)	8.0
Mean High Water Springs	6.9
Mean High Water	5.9
Mean High Water Neaps	4.9
Mean Sea Level (MSL)	4.0
Mean Low Water Neaps	3.1
Mean Low Water	2.2
Mean Low water Springs	1.2
Lowest Astronomical Tide (LAT)	0.0

location of the oceanographic mooring sites is shown in Figure 1. These data are used to calibrate a numerical model of the water circulation. The present numerical hydrodynamic model covers a 620 km^2 domain ending at the shoals offshore from the heads of Darwin Harbour. The models used are RMA2, a finite element depth averaged model, and RMA11, a cohesive sediment transport model (King, 2004). The models use tidal elevations at the sea as a boundary condition. In the upper arms, the models assume no freshwater inflow during the dry season and they use the flows recorded at gauging stations in the rivers draining into the arms during the wet season.

The model output, verified against field data (Figure 3), shows a complex circulation near headlands and embayments, which includes jets, eddies, separation points, and stagnation zones (Figure 4). These currents are different at flood and ebb tides. It results that dispersion of water-born particles is asymmetric and that trapping occurs near headlands and in embayments (Signell and Geyer, 1990; Wolanski, 1994). The importance of the tidal asymmetry on the fate of fine sediment is described later.

4. SALINITY

The salinity varies both spatially (horizontally and vertically) and temporally (both at tidal frequency and at seasonal frequencies). Oceanic salinity at the mouth of the harbour remains almost constant (variation less than 1) throughout the year.This implies that the freshwater runoff is strongly diluted by the time it reaches the mouth of the harbour. There is no marked river plume exiting Darwin Harbour so most of the mixing between freshwater runoff and the ocean occurs within the harbour and this is attributed to the large tides (eg Wolanski and Spagnol, 2003).

Most of the freshwater runoff occurs in the period January to March in the form of a few discrete flood events. Each event lasts from a few days to a few weeks, reflecting the small watershed and the patchy rainfall. During these events the horizontal salinity gradient is at a maximum (Figure 5) and a sharp front visibly separates water in each arm from that in the main body of the harbour and the water is stratified on both sides of the front (Figure 6). Mixing is inhibited between the arms and the harbour. As a result of the strong tidal currents, the vertical stratification persists for up to two weeks in the arms, and for less than one week in the main body of the harbour. In the wet season the harbour is a stratified estuary.

In the dry season the system is vertically well-mixed in salinity. Due to evapotranspiration an along-channel salinity gradient is created. This salinity gradient reaches a maximum value late in the dry season. For instance, on October 10, 2004, the salinity was 35.9 at the mouth of the West Arm and 38.9 in the upper

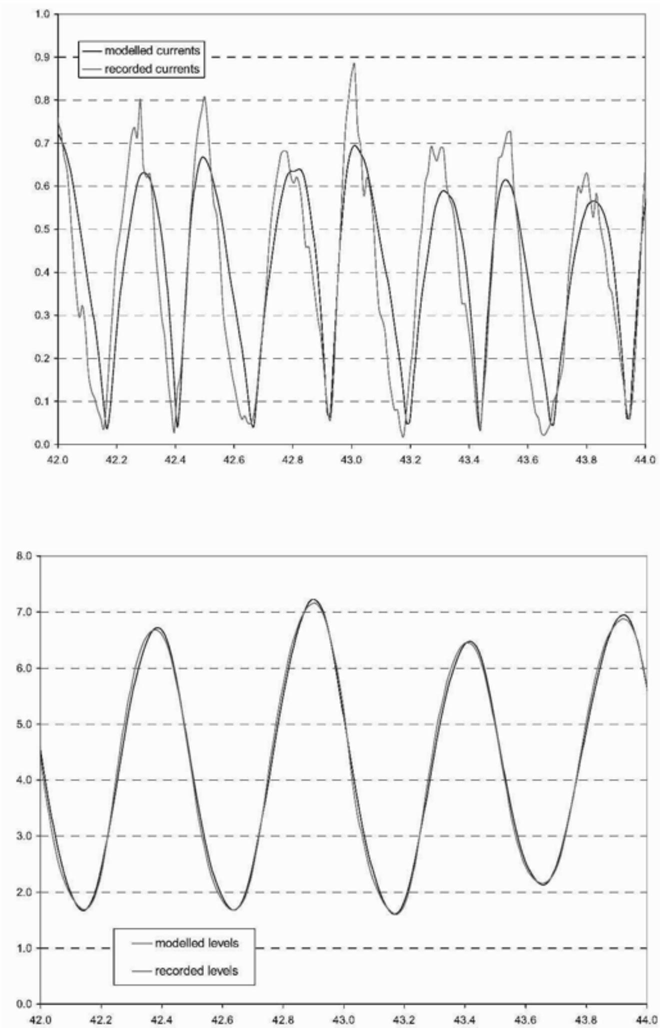


Figure 3. *Comparison of depth-averaged, predicted versus recorded tidal currents (m s^{-1}) and water levels (m) near Wickham Point.*

reaches (not shown). The vertical stratification was less than 0.1 across the depth. In the dry Season the West Arm is an inverse estuary. The loss of water by evaporation is compensated by an inflow of oceanic water. The water is then trapped in the upper reaches of the arm. A similar process was found in the East Arm. In the Middle Arm, a small freshwater spring inflow persists in the upper reaches. This results in the formation of a salinity maximum zone located in the mid-region of the arm (Wolanski, 1986), and the water is still trapped upstream of the salinity maximum zone.

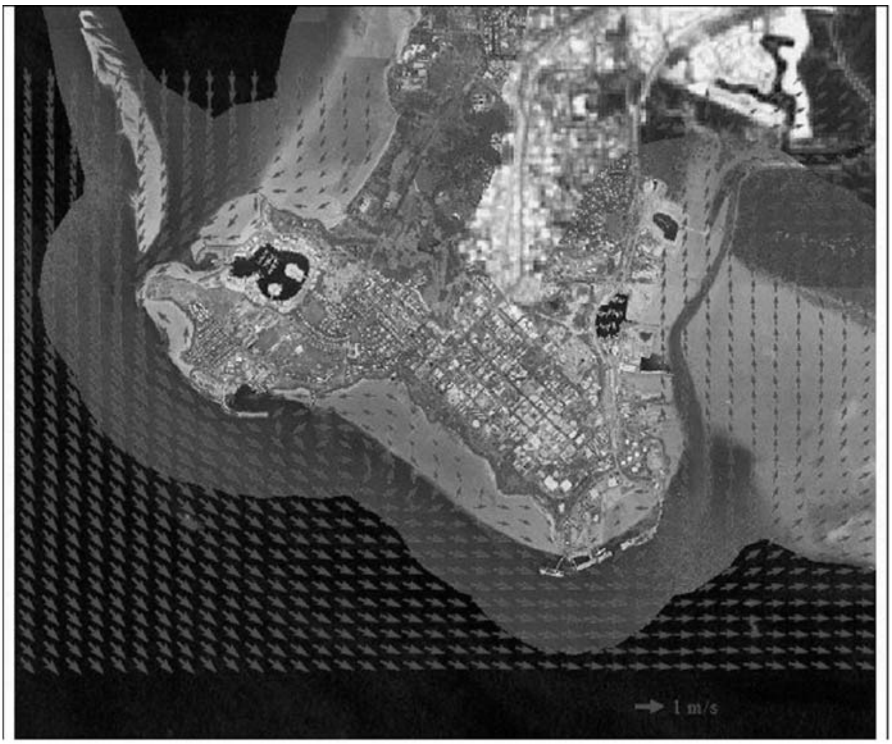


Figure 4. *A snapshot of the predicted distribution of the horizontal currents at flood tide near Darwin City during a spring tide. The blue shading indicates the intertidal area.*

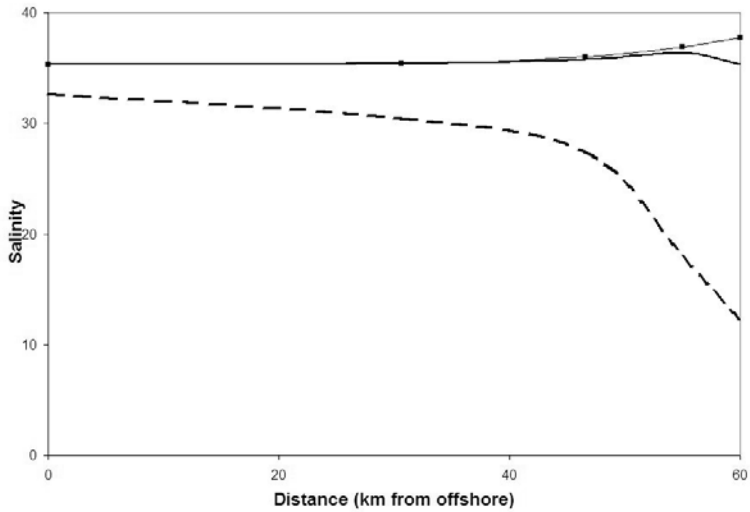


Figure 5. *Horizontal profiles of salinity to the tidal limit of Darwin Harbour in the wet (---) and dry (line) seasons of Darwin Harbour for the Middle Arm (circles) and the West Arm (squares).*

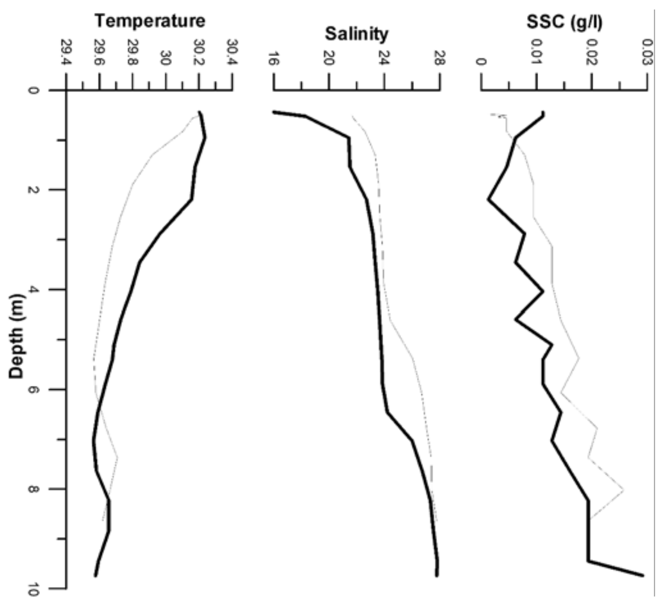


Figure 6. *Vertical profiles of salinity, temperature, and suspended sediment concentration (SSC) in the harbor (thin line) and in the West Arm (thick line) on either side of a front spanning the width of the arm near high tide at 1310 h on March 22, 2004.*

5. RESIDENCE TIME

In the dry season Darwin Harbour is an inverse estuary, similar to many other Australian tropical estuaries (Wolanski, 1986). A salinity maximum zone is observed in the upper reaches. The loss of water from evaporation and evapotranspiration is compensated by an inflow of oceanic water. Excess salt is exported by tidal dispersion. A steady state a salt balance is established, so that the tidally-averaged outflow of salt is equal to the inflow of seawater. This balance implies that (Wolanski, 1986):

$$E_t\, S\, A_h \sim E\, A\, dS/dx \qquad (1)$$

where S is the salinity, E_t is the evaporation/evapotranspiration rate, x is the distance along channel, E is the along-channel eddy diffusion coefficient, A is the cross-sectional area, and A_h is the surface area of the harbour including the intertidal (both vegetated and unvegetated) area. In each arm, $E \sim 0.007$ m d^{-1}, $A_h \sim 10$ km^2, $A \sim 400$ m^2, thus $E \sim 100$ m^2 s^{-1}.

Therefore in the dry season the flushing time T_f of water from the arms of Darwin Harbour is

$$T_f \sim L^2/E \sim 20 \text{ days} \qquad (2)$$

where L is the length of the estuary.

This long residence time is also supported by the output from the numerical model (see Animation 1 discussed later).

6. COARSE SEDIMENT

The predicted, tidally-averaged, residual circulation (Figure 7) shows a number of eddies centered near sand shoals, suggesting that the location of these shoals is determined by the tidal circulation (Bastos et al., 2004).

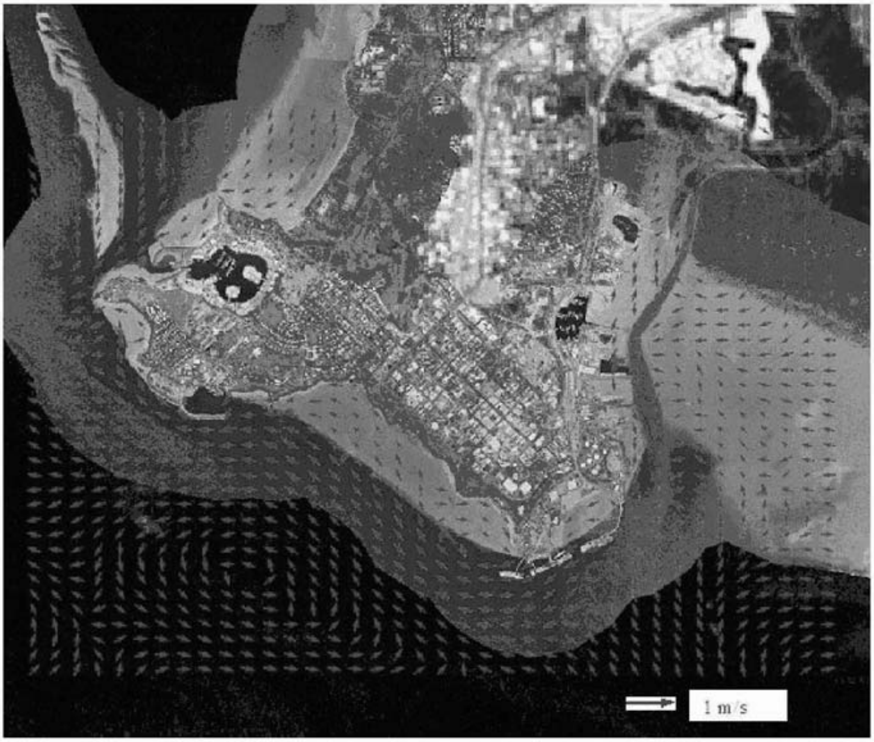


Figure 7. *Synoptic distribution of the predicted, residual, tidally-averaged circulation near Darwin City, superimposed on an aerial photo taken at low tide.*

7. SUSPENDED SEDIMENT

The waters are moderately turbid; with a small vertical stratification is suspended sediment concentration (SSC; see Figure 6). SSC values are usually less than 50 mg l^{-1}. A turbidity maximum zone exists inside the mouth of each arm in both the dry and wet seasons. In that turbidity maximum SSC values can reach 250 mg l^{-1} (see Figure 8). This figure also demonstrates that the SSC values fluctuate markedly at tidal frequency and also with the spring-neap tide cycle. This behaviour is similar to other tropical, shallow, macro-tidal estuaries such as the Fly Estuary, the Ord Estuary and King Sound where, like in Darwin Harbour, these processes lead to the formation of turbidity maximum zone and trapping of fine sediment (Wolanski and Spagnol, 2003; Wolanski et al., 1995 and 2004).

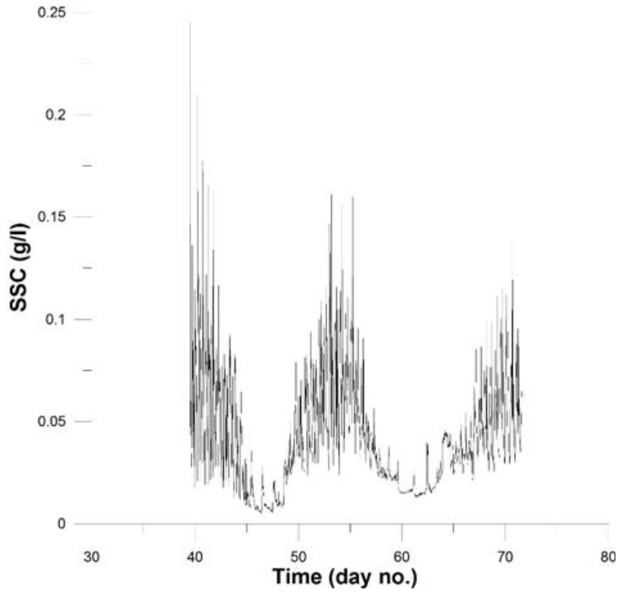
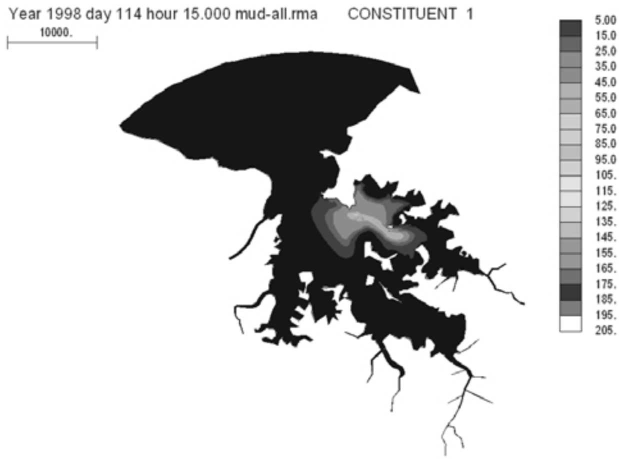


Figure 8. *Time series plot of the suspended sediment concentration (SSC) in the main body of the harbor near Wickham Point. Time starts on January 1, 2004.*

A small amount of fine sediment is delivered to the upper arms by runoff; the majority of the fine sediment in the upper arms is dispersed from the channel banks and mangrove zones by the freshwater runoff. Most of this sediment then drops out of suspension from the surface plumes because the SSC is smaller in the plume than near the bottom (Figure 6). This fine sediment is then eroded and deposited at tidal frequencies, especially at spring tides. The combination of the complex circulation near headlands and embayments, and of the asymmetry of the tidal currents controls the fate of fine sediment (Animation 1). This animation shows the predicted spread of mud discharged continuously near Darwin City within the harbour. The mud is rapidly dispersed and preferentially transported up-estuary towards mud banks and mangroves where it settles. The remaining small fraction is exported to the ocean.

The net fluxes of sediment over one month were also calculated from the field data on currents and suspended sediment concentration collected at 15 min intervals, over one month, in both the dry and wet seasons of 2004, at four sites in the inner harbour, using the method of Wolanski et al. (2004). These net fluxes, shown in Figure 9, are directed up-estuary, also indicating trapping of fine sediment.



Animation 1. *The predicted movement of a muddy plume arising from a continuous mud discharge from a dredging operation.*

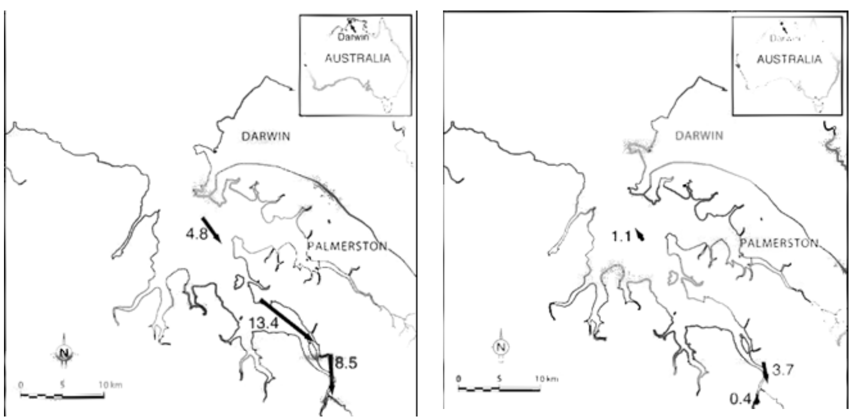


Figure 9. *Net suspended sediment fluxes (tons $m^{-1}d^{-1}$) at the oceanographic mooring sites in (left) February-March 2004 (wet season) and (right) October-November 2004 (dry season).*

Futher evidence of tidal trapping in the upper arms of the harbour was collected during the 2004 – 2005 wet season when an outbreak of a noxious aquatic weed was discovered in Darwin River which then flows into Berry Creek (see a location map in Figure 10), a small tributary of the Blackmore Estuary. The river was blocked and the weed was treated with the herbicide 2,4D (Padovan, unpublished data). 2,4D can be used as a conservative tracer as it has a half life of ~ 30 days and in acidic form does not adsorb to sediments. Monitoring of 2,4D concentrations in the Blackmore Estuary showed that the concentrations increased in an upstream direction showing that the chemicals were trapped and not able to advect or diffuse downstream (Figure 10).

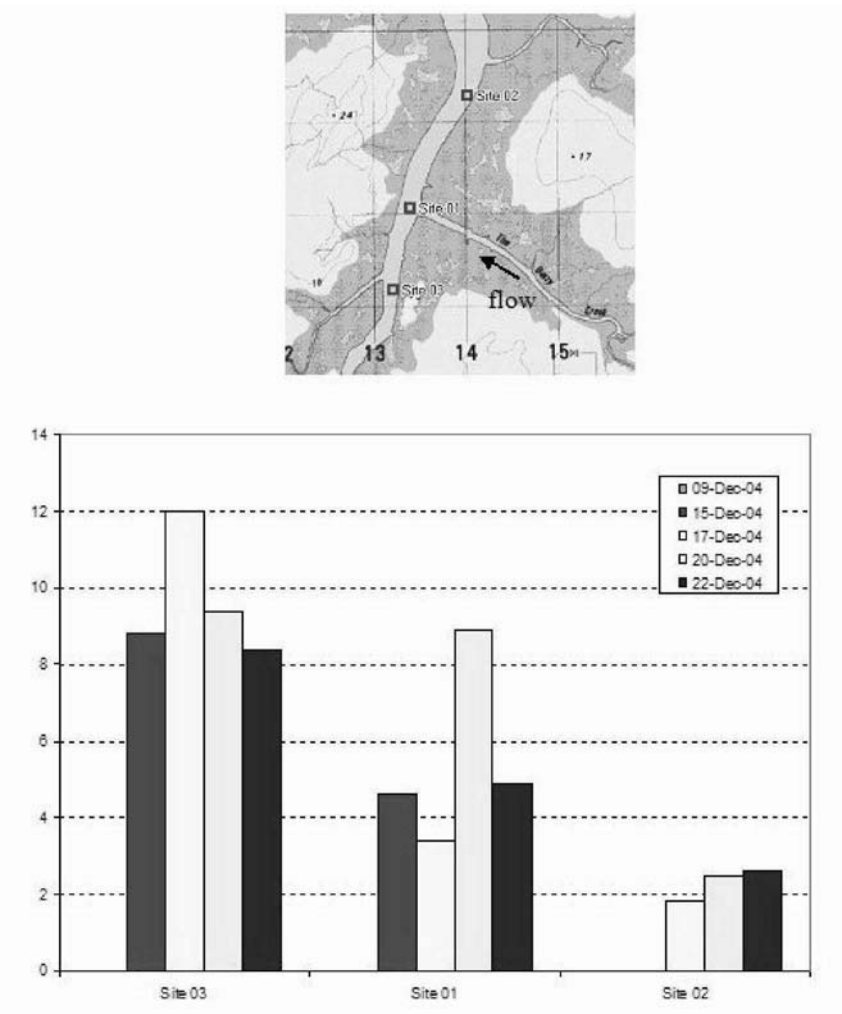


Figure 10. *2,4D monitoring sites on the Blackmore Estuary showing how the 2,4D concentrations increased upstream of the river confluence, indicating that particles are trapped and their advection and diffusion downstream is limited.*

Microscope pictures of the suspended sediment in suspension show the fine sediment is flocculated, with floc size typically 50-200 μm. Near the mouth of each arm, in waters with SSC values less than 40 mg l^{-1}, there exists a marine snow zone where the floc size is in the range 200-1000 μm (Figure 11). In that zone, the influence of plankton and bacteria on the flocculation of suspended mud is dominant, similarly to that experienced in other macro-tidal tropical estuaries (Wolanski and Gibbs, 1995; Ayukai and Wolanski, 1997; Wolanski and Spagnol, 2003). This occurs because muddy marine snow offshore is formed as a result of the small mud flocs adhering to sticky transparent exopolymeric particles (TEP; see Passow and Alldredge, 1994; Alldredge and Gotschalk, 1988). The flocs are teeming with very small plankton that we could just detect with our microscope (<5 μm) and these attracted plankton, presumably to feed on, while other plankton seem to be smeared and stressed by the sticky flocs (see animations 2 and 3), appearing to try to remove the coating of mud, behaving in a similar way that benthic coral organisms behave at the bottom (Fabricius and Wolanski, 2000).

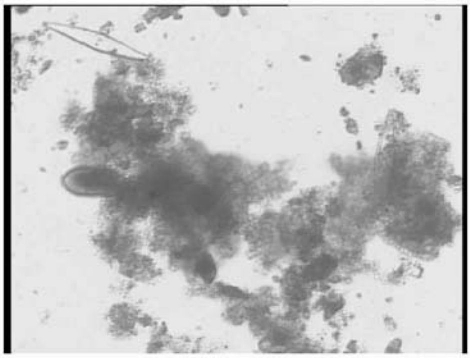
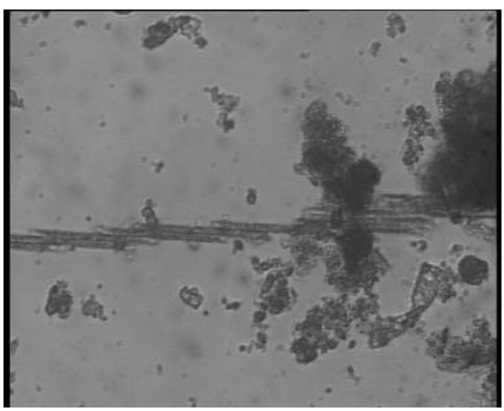
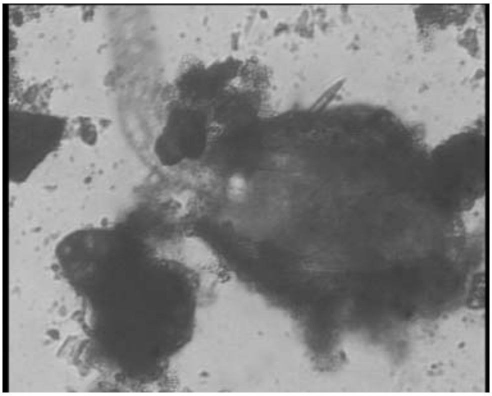


Figure 11. *Microphotograph spanning 500 μm of a marine snow floc at the mouth of the Middle Arm.*



Animation 2. *Micro-movie spanning 500 μm of a muddy marine snow floc in Darwin in Darwin Harbour, showing Asteronella diatoms moving and feeding on the mud.*



Animation 3. *Micro-movie spanning 500 μm of an animal caught in sticky, muddy marine snow and apparently trying to shed the muddy coating.*

The muddy marine snow layer in each arm of Darwin Harbour is important to the fate of fine sediment because it enhances the trapping of fine sediment (Ayukai and Wolanski, 1994).

8. CONCLUSION

Although macrotidal, Darwin Harbour is poorly flushed, especially in the dry season when the residence time in the upper reaches is of the order of 20 days. Much of the riverine fine sediment remains trapped in mud flats and mangroves with little escaping to the sea. The complex bathymetry of headlands and embayments generate complex currents comprising jets, eddies, and stagnation zones that can trap pollutants inshore. The tidally averaged circulation may control the location of the sand banks, indicating a feedback between the bathymetry and the water circulation. The environment in Darwin Harbour has the potential to degrade and the water circulation in the harbour must be considered when planning developments.

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CHAPTER 27

AN ESTUARINE ECOHYDROLOGY MODEL OF DARWIN HARBOUR, AUSTRALIA

ERIC WOLANSKI, A. DAVID MCKINNON, DAVID WILLIAMS, AND DANIEL M. ALONGI

1. INTRODUCTION: DARWIN HARBOUR - THE NEED FOR AN ECOHYDROLOGY ESTUARINE MODEL.

Throughout human history, the coastal plains and lowland river valleys have usually been the most populated areas over the world (Wolanski et al., 2004a). This is degrading estuarine and coastal waters through pollution, eutrophication, increased turbidity, overfishing, and habitat destruction (Lindeboom, 2002). The pollutant supply does not just include nutrients, but also includes mud from eroded soil, heavy metals, radionuclides, hydrocarbons, and a number of chemicals including new synthetic products. Darwin Harbour (Figure 1) is no exception. Taking the Harbour to include all waters inshore of Gunn and Charles Points, it covers 3,227 km^2, and the drainage is 2,417 km^2. The harbour is an estuary with three arms, each of which drains a seasonal river with negligible flow in the dry season. Several pollution sources exist in this estuary, mainly on the east side. Urbanization, industrialisation, dredging, dredge spoil discharge, sewage discharge, shipping, agriculture (fertilizers, pesticides, herbicides), aquaculture wastes, two harbours and several marinas, all represents threats to Darwin Harbour. By comparison, the west side is the least disturbed.

To prevent major environmental degradation as happened in the other harbours described in this book, there is a need for a science-based integrated management plan that considers the whole Darwin Harbour catchment as the fundamental planning unit. For science to help in this process, it must provide useful data and tools. One such scientific tool is a model to quantify the human impact on the ecosystem health of the harbour. The health of an estuary or a coastal water body needs to include a number of potentially conflicting variables (Balls, 1994), requiring the use of modelling tools. This chapter describes such a model for Darwin Harbour.

The model is kept as simple as possible to be practical while remaining realistic. In the model, flushing and mixing processes (that are readily measured in the field

E. Wolanski (ed.), The Environment in Asia Pacific Harbours, 477–488.

or that are assessed from hydrodynamic modeling studies) parameterize the water and fine sediment dynamics. The ecological sub-model incorporates the ten state variables important in controlling the ecosystem.

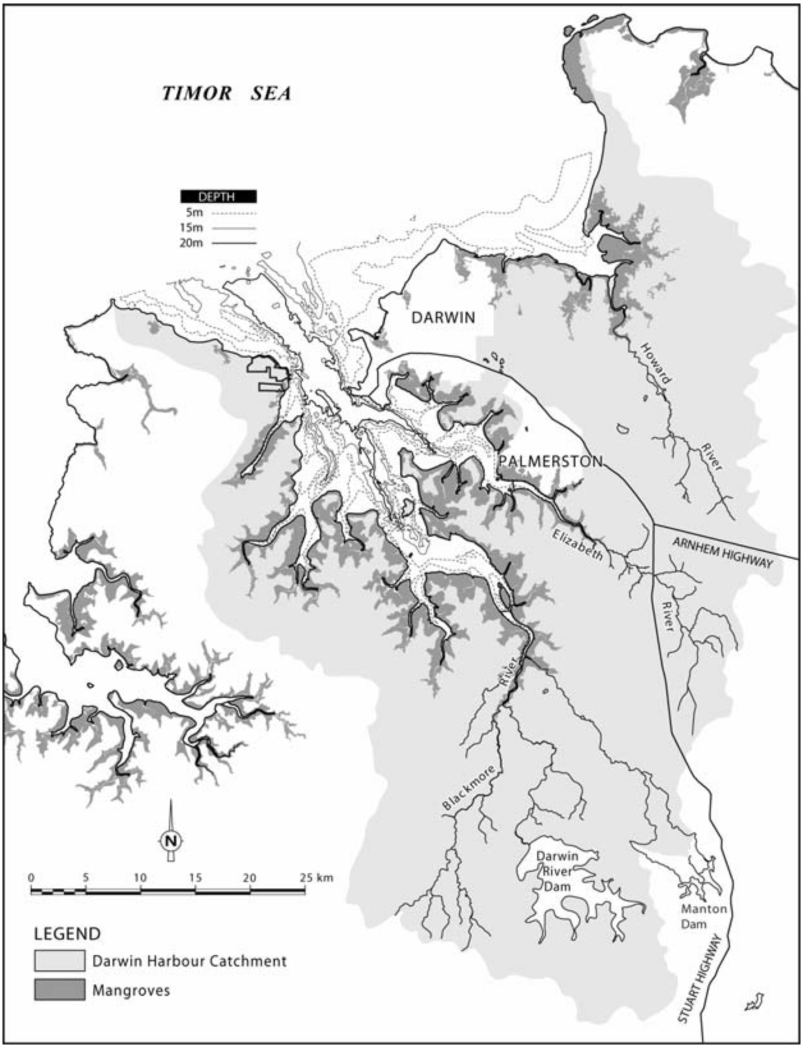

Figure 1. *Map of Darwin Harbour showing the drainage area (shaded), the rivers, the mangroves, the urban areas (white within the drainage area), and the bathymetry (depth in m). The three main arms of the Harbour are called the East Arm, Middle Arm and West Arm.*

2. THE SCIENCE BEHIND THE MODEL

The model is process-based. The dominant physical and biological processes are detailed in the previous chapters in this book by Williams et al. and McKinnon et al., and are sketched in Figure 2.

Fine sediment is found in suspension in Darwin Harbour, either entering with runoff water or resuspended from the bottom. This sediment is largely trapped within an estuarine turbidity maximum (ETM) zone; in our field studies we found an

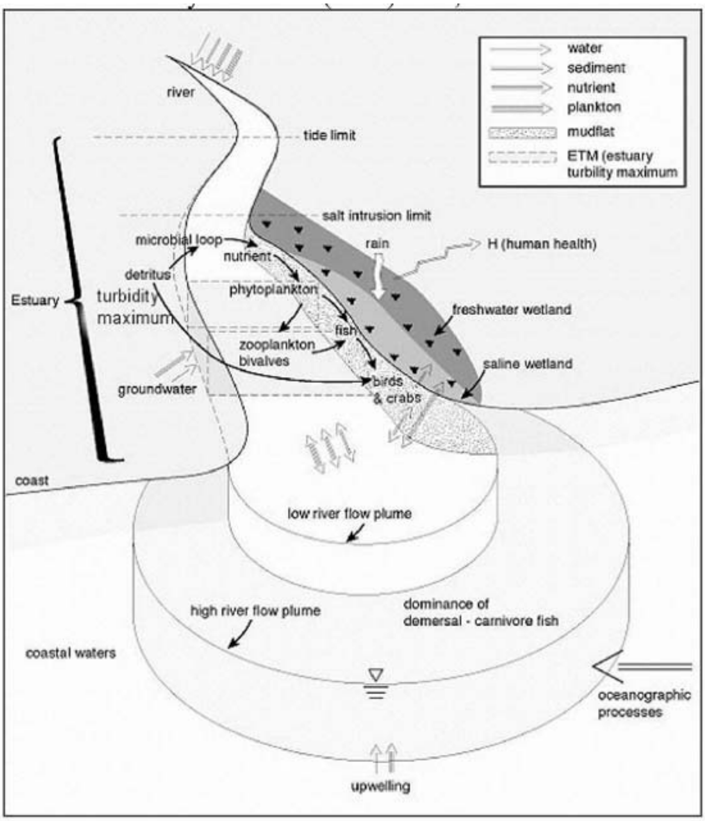

Figure 2. *Sketch of the dominant processes operating in an estuary. (Adapted from Wolanski et al., 2004a).*

ETM within each of the three arms of the harbour. Sediment particles and aggregates within the ETM give rise to marked changes in water quality. Nutrients are mainly in particulate form (i.e. absorbed to the mud particles in suspension) in freshwater during the wet season when runoff brings them in the Harbour. These nutrients are released in solution in saline water and are recycled during the dry season. Therefore one key process (Figure 3) is the leaching and microbial transformation of nutrients from the particulate to dissolved form (Day et al, 1989). Silt and clay sediments are located mostly in the intertidal areas of the harbour and restricted subtidally to shoal areas in close proximity to the mouth of the East Arm. Nutrient recycling in these deposits is slow (Alongi, unp. data) so the bulk of nutrient transformations appear to occur within pelagic microbial food webs.

The model recognises the key properties observed in the field, namely that the bulk of nutrient cycling occurs in the water column rather than in subtidal sediments, that the rates of pelagic respiration are greater than the rates of sediment respiration, that the rates of nitrogen turnover in the water column are fast, on the order of hours only (and not days), indicating that fine sediment stirring by the strong tidal currents maintain high organic loads in the water column to be degraded by bacterioplankton communities, and that there is no measurable benthic denitrification (Alongi and Trott, unp. data).

Darwin Harbour is modelled (Figures 2 and 3) as a converter of living phytoplankton to detrital particles and is also a conveyor of detrital matter to the coastal ocean (Knox, 1986; McKinnon et al., 2005 – this book). Fishes help transfer energy and matter from estuarine plants to upper trophic levels. Our model assumes that most of the organic matter is processed through the detritus-based food webs, as it is in other coastal ecosystems (Alongi, 1998). Zooplankton, planktivorous fish, and surface deposit-feeding shrimps consume a proportion of the primary production of the phytoplankton and microphytobenthos. Mangroves provide large quantities of organic detritus. In addition, there is an input of detritus from land runoff in the wet season. Detrital particles and their associated microorganisms provide the basic food source for secondary consumers.

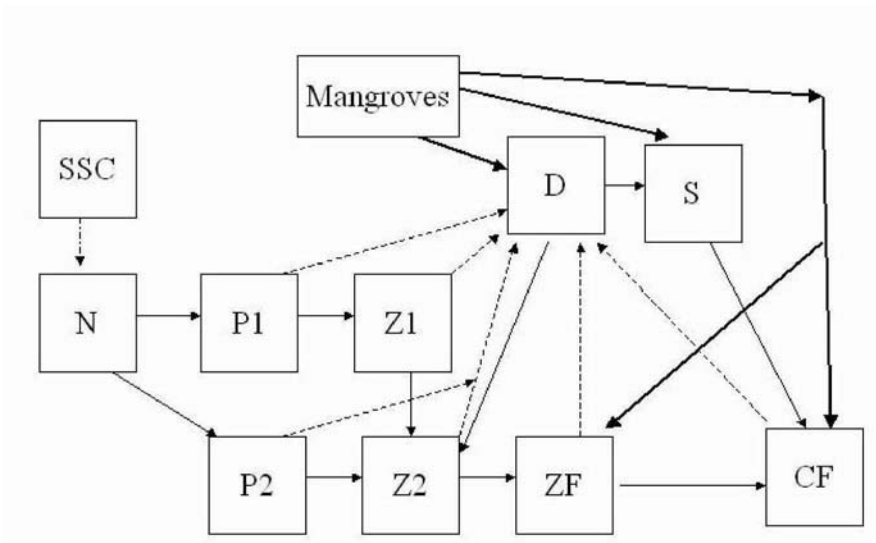

Figure 3. *Structure of the estuarine food web in the ecohydrology model (see text). Thin continuous arrows indicate a prey-predator relationship, thick arrows the role of mangroves, and discontinuous arrows the other mass transfers.*

3. THE ESTUARINE ECOHYDROLOGY MODEL

As described by Williams et al. (2005), Darwin Harbour is vertically well-mixed mixed estuary for most of the year. Thus the distribution of a salinity S is determined from the solution of the 1-D advection-diffusion equation (Fischer et al., 1979):

$$\partial(SA)/\partial t + \partial(QS)/\partial x = \partial(EA\partial S/\partial x)/\partial x \quad (1)$$

where t is the time, x is the distance along the estuary (x=0 at the tidal limit), Q is the flow rate (driven by tides, evaporation and runoff), E is the longitudinal eddy diffusion coefficient, and A is the cross-sectional area. Mixing, which determines the diffusive outflow, is characterized by the longitudinal eddy diffusion parameter E. Equation (1) is solved for a series of cells of volume V spread along the length of the harbour of total volume of 4.26×10^9 m^3. The funnel-shaped estuary cells have a constant along-channel length dx, and V varies between 2.6×10^6 m^3 at the tidal limits and 4.6×10^8 m^3 at the outer boundary of the harbour. The time step dt is set to 1 day, thereby averaging over the tides.

In the wet season (~ 3 months duration) there is river runoff and consequently two open boundaries are used; the first one is located at the tidal limit and the second one at the mouth.

In the dry season, which lasts the rest of the year, there is no river runoff. During that time, Darwin Harbour is an inverse estuary, similar to many other tropical estuaries (Wolanski, 1986). In the dry season the flushing time of water from the arms of Darwin Harbour is about 20 days (Williams et al., 2005 – this book).

This period of time for is sufficient for biological processes to make important transformations relative to those observed from physical processes alone. To model these it is recognised that the constituents C, such as nutrients, plankton, detritus, fish, and shrimps, are non-conservative. Equation (1) is modified by replacing S by C and by including a sink-source term ΔC (Thomann, 1980) derived from the ecological sub-model. The ecological sub-model is based on the non-linear Lotka-Volterra equation (Flindt et al., 1997; Kot, 2001; Brauer and Castillo-Chavez, 2001; Edwards and Andrew, 2001):

$$\partial C/\partial t = \beta\, X\, (1 - C/C_o)\, H(Y,Y_{o1}) - \delta\, C + \alpha \quad (2)$$

where C is the predator concentration, Y is the prey biomass, β is the predator growth rate, C_o is the predator saturation biomass, and Y_o is the prey starvation biomass, i.e. the biomass at which the predator is unable or unwilling to spend energy to find prey. H is the Heavyside function, i.e. H=0 if $Y < Y_o$, and H=1 if $Y > Y_o$. δ is the death-excretion rate, excluding death by predation. α is a source term that parameterises the fact that the fringing wetlands (mainly mangroves and riparian ecotones, together with the tidal creeks that drain them) can be an important source of detritus, as well as a nursery and refuge for juvenile and sub-adult shrimps and fish (Blaber, 1997).

The ecosystem model represents mathematically through Equation 1-2 the interactions summarized in Figure 3 between nutrient concentration (N; representing NO_3), suspended sediment concentration (SSC), phytoplankton concentration (as P1 and P2; P1 representing respectively picoplankton cell counts derived from flow cytometry (McKinnon et al., this volume) and total chlorophyll, zooplankton concentration (Z1 and Z2, representing respectively all nauplii and adult copepods), the concentration of detritivores such as shrimp (S), detritus concentration (D), zooplanktivorous fish concentration (ZF), and carnivorous fish concentration (CF). This model structure recognizes the contribution of picoplankton to primary production in tropical waters (Furnas and Mitchell, 1987) and microzooplankton as the most important consumers of this fraction of the primary production (Calbet and Landry, 2004). Here we use copepod nauplii as an index of microzooplankton abundance. All dying matter becomes detritus. Settling is not included in the model, because the animals (eg zooplankton) are mobile and can swim in the water. The model is equally complex at the lowest and highest trophic levels, which increases the model robustness (Jorgensen and Bendoricchio, 2001). Thus the ecosystem model equations are:

Nutrients (N)

$$\partial N/\partial t = -\beta_{NP1}\, P1\, (1 - P1/P_o)\, H(N,N_{o1}) - \beta_{NP2}\, P\, (1 - P2/P_{o2})\, H(N,N_{o1})$$

$$+ \alpha_N + \gamma_{SSCN}\, SSC \qquad (3)$$

Phytoplankton (P)

$$\partial P1/\partial t = \beta_{NP1}\, P1\, (1 - P1/P1_o)\, H(N,N_{o1}) - \beta_{P1Z1}\, Z1\, (1 - Z1/Z1_o)\, H(P1,P1_{o1})$$

$$+ \alpha_{P1} - \delta_{P1}\, P1 \qquad (4)$$

$$\partial P2/\partial t = \beta_{NP2}\, P2\, (1 - P2/P2_o)\, H(N,N_{o1}) - \beta_{P2Z2}\, Z2\, (1 - Z2/Z2_o)\, H(P2,P2_{o1})$$

$$+ \alpha_{P2} - \delta_{P2}\, P2 \qquad (5)$$

Zooplankton (Z)

$$\partial Z1/\partial t = \beta_{P1Z1}\, Z1\, (1 - Z1/Z1_o)\, H(P1,P1_{o1}) - \beta_{Z1Z2}\, Z2\, (1 - Z2/Z2_o)\, H(Z1,Z1_{o1})$$

$$+ \alpha_{Z1\,-}\, \delta_{Z1}\, Z1 \qquad (6)$$

$$\partial Z2/\partial t = \beta_{P2Z2}\, Z2\, (1 - Z2/Z2_o)\, H(P2,P2_{o1}) + \beta_{DZ2}\, Z2\, (1 - Z2/Z2_o)\, H(D,D_{o1})$$

$$- \beta_{Z2ZF}\, ZF\, (1 - ZF/ZF_o)\, H(Z2,Z2_{o1}) + \alpha_{Z2} - \delta_{Z2}\, Z2 \qquad (7)$$

Detritivores (S)

$$\partial S/\partial t = \beta_{DS}\, S\, (1 - S/S_o)\, H(D,D_{o1}) - \beta_{SCF}\, CF\, (1 - CF/CF_o)\, H(S,S_{o1}) + \alpha_S - \delta_S\, S \qquad (8)$$

Carnivorous fish (CF)

$$\partial CF/\partial t = \beta_{SCF}\, CF\, (1 - CF/CF_o)\, H(S,S_{o1}) + \beta_{ZFCF}\, CF\, (1 - CF/CF_o)\, H(ZF, ZF_{o1})$$

$$+\alpha_{CF} - \delta_{CF}\, CF \qquad (9)$$

Zooplanktivorous fish (ZF)

$$\partial ZF/\partial t = \beta_{Z2ZF}\, ZF\, (1 - ZF/ZF_o)\, H(Z2,Z2_{o1}) - \beta_{ZFCF}\, CF\, (1 - CF/CF_o)\, H(ZF,ZF_{o1})$$

$$+\alpha_{ZF} - \delta_{ZF}\, ZF \qquad (10)$$

Detritus

$$\partial D/\partial t = - \beta_{DS}\, S\, (1 - S/S_o)\, H(D,D_{o1}) - \beta_{DZ2}\, Z2\, (1 - Z2/Z2_o)\, H(D,D_{o1}) + \alpha_D$$

$$+ \delta_S\, S + \delta_{P1}\, P1 + \delta_{P2}\, P2 + \delta_{Z1}\, Z1 + \delta_{Z2}\, Z2 + \delta_{CF}\, CF + \delta_{ZF}\, ZF \qquad (11)$$

In these equations the subscripts denote the constituents. For instance, δ_{ZF} is the death-excretion rate of ZF, and β_{Z2P2} is the growth rate of Z2 from consuming phytoplankton P2, i.e. the rate of mass transfer from P2 to Z2. In the nutrient equation, a new parameter was introduced, γ_{SSCN}, denoting the leaching rate of nutrients from the particulate phase (i.e. absorbed on the fine sediment) to the dissolved phase. The parameter α is equal to zero in the absence of mangroves and is assumed to be proportional to the ratio between the mangrove area fringing a cell and the surface area of a cell, with a maximum value α_{max} of 0.05.

In Equation (2), because the model is run at a time step of 1 day, $Q = Q_f + Q_e$, where Q_f (>0) is the freshwater discharge and Q_e (<0) is the seawater inflow a result of evapotranspiration.

When applying Equation (2) to detritivores and fish, Q is modified to incorporate the horizontal swimming by the fish as fish swim, by kinesis or taxis following environmental cues (Wolanski et al., 1997; Humston et al., 2000).

4. APPLICATION TO DARWIN HARBOUR

The field observations of SSC values are used as an external input in the model. The model necessitates knowledge of a number of parameters used in Equations 3-11. Table 1 lists the adopted values of the parameters.

***Table 1**. Final values of the parameters. Rates are expressed as d^{-1}.*

β_{NP1}	0.2
β_{NP2}	0.3
β_{P1Z1}	0.07
β_{P2Z2}	0.05
β_{Z1Z2}	0.4
β_{Z2ZF}	0.4
β_{SCF}	0.15
δ_{CF}	0.15
δ_{ZF}	0.15
β_{SSCN}	0.9
β_{DS}	0.9
β_{DZ2}	0.7
δ_{P1}	0.05
δ_{P2}	0.17
δ_{Z1}	0.05
δ_{Z2}	0.1
$\alpha_{S,max}$	0.05
$\alpha_{CF,max}$	0.05
$\alpha_{D,max}$	0.05
$\alpha_{ZF,max}$	0.05

The along-channel distribution of observed and predicted values of N, P1, P2, Z1 and Z2, compare favourably with each other for the both the dry (Figure 4) and wet seasons (Figure 5). It is stressed again that in the dry season each parameter has only one open boundary condition, namely offshore (i.e. the observed oceanic values). Thus the internal ecological processes occurring within the hydrodynamic time scales strictly determine the success of the ecological model throughout Darwin harbour. This is therefore a very powerful model verification.

While the calibration appears successful, it is important for the user to also judge whether the solution is realistic and stable (Hilborn and Mangel, 1997). This was done by undertaking a sensitivity test to judge whether the model is unrealistically sensitive to a specific parameter, making it potentially unstable and unrealistic. Each test involves changing one parameter for the dry season conditions. The model is found to be robust because large but reasonable changes in the parameters do not lead to instabilities such as the destruction of trophic layers. The sensitivity tests reveal that the biomass of organisms is directly affected by its consumption of prey or by its being predated on by predators higher in the food chain. Indirect effects across two trophic levels are found to be generally small.

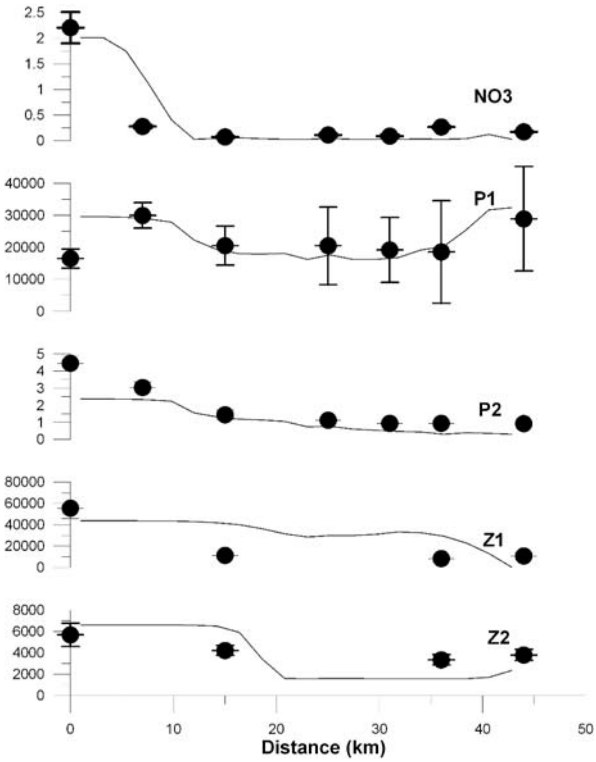

Figure 4. *Observed (circles and standard deviation of all the observations) and predicted (lines) along-channel distribution of the concentration of NO_3 (ppm), P1 (cells ml^{-1}), P2 (ppm), Z1 and Z2 (number l^{-1}) for all the dry season observations in 2003-2004 (number of observation n=7 except for P1 for which n=4). The distance is measured from the tidal limit.*

The model can be used to assess the importance of freshets in the season. It suggests (not shown) that the pulse-like fluctuations in river discharge increase the diversity and variability in plankton and fish communities and thus promote the ecosystem dynamics and robustness, in a similar manner as in the Guadiana Estuary in Portugal (Wolanski et al., 2004b).

5. EXAMPLES OF MANAGEMENT APPLICATION OF THE MODEL

The model can readily be used to test scenarios for human impacts from development. Three such scenarios were tested, namely (1) what would be the impact of doubling the suspended sediment concentration through increased land clearing, (2) what would be the impact of destroying all the mangroves, and (3) what would be the impact of a doubling of the nutrient concentration in the upper and middle regions as a result of sewage, farming or aquaculture, with no increase in SSC.

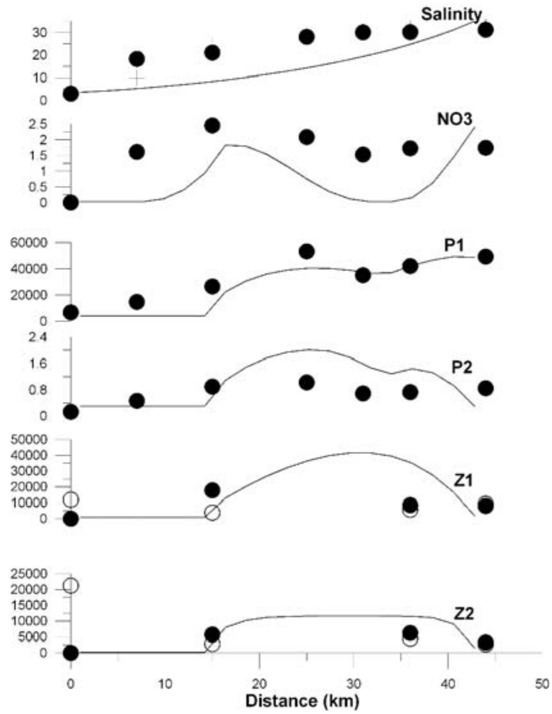

Figure 5. *Observed (February 2003 = ●; February 2004 = O; March 2004 = +) and predicted (lines) along-channel distribution of salinity, and of the concentration of NO_3 (ppm), P1 (cells ml^{-1}), P2 (ppm), Z1 and Z2 (number l^{-1}) during the wet season. The distance is measured from the tidal limit.*

In case 1, the model suggests (not shown) that the dissolved nutrient concentration would quadruple in the upper half of the harbour, P1 would increase by 50% in the middle region, P2 would quadruple in the middle region, Z1 would be little affected and Z2 would triple in the middle region of the harbour. In case 2, the carnivorous fish would decrease by 70%, and the detritivores by 50%, in the upper and middle reaches where they are most abundant. In case 3, P1 would increase by 50%, in the upper and middle reaches, P2 would quadruple in the middle reaches, Z1 would increase by about 30% in the upper and middle reaches, and P2 would quadruple in the middle reaches.

Thus the model suggests that the risk of eutrophication would be greatly increased, and the ecological services provided by the estuary greatly reduced, by such large-scale changes to land-use. This clearly demonstrates that, notwithstanding its macro-tides, Darwin Harbour has the potential to be degraded significantly as is the case in other harbours described in this book.

This estuarine ecohydrology model is able to provide answers to a number of practical questions. These answers must always be interpreted with caution because the model, like any ecosystem model, over-simplifies reality, and the data set is inadequate for a detailed calibration. In that sense the model predictions are somewhere between quantitative and qualitative. Detailed field studies are needed to better understand, and hence better parameterise in the model and the important processes driving the ecosystem. For the model to remain a useful tool, it is suggested that its complexity should be increased only as fast as additional physical and biological processes can be quantified through new field and laboratory studies. Martin (2005) has developed a trophic model of Darwin Harbour using Ecopath with Ecosim (Christensen and Walters, 2000). This model differs in its assumptions from the ecohydrological approach that we have used, and though it has a better description of trophic linkages, it lacks the spatial components and the linkage with the plankton and the hydrodynamics. In the future, a fully developed model for Darwin Harbour should include both approaches.

For science, the present model provides a tool to enable the exchange of information between oceanographers, biologists, ecologists, engineers, sociologists, economists and water-resources managers.

It is hoped that the model can also be useful for management. The model shows that it is possible to predict – within likely error bounds provided by the sensitivity tests – the consequences on the estuary ecosystem health of human activities in the catchment. The model does show that, to maintain the ecosystem services provided by the estuary, integrated coastal management needs to take the whole catchment as the fundamental planning unit. The ecologically sustainable solution to the management of Darwin Harbour is to adopt ecohydrology as the underpinning principle to guide the management of the entire catchment (Zalewski, 2002).

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CHAPTER 28

IS HARBOUR DEVELOPMENT ECOLOGICALLY SUSTAINABLE?

ERIC WOLANSKI

1. INTRODUCTION

Not one of the twelve harbours studied in this book was developed in an ecologically sustainable way. The data presented in this book quantify environmental impacts ranging from degradation to ecosystem collapse.

A simple measure of the impact of harbours and urbanisation is the fisheries. In Tokyo Bay the total annual fish catch of fish decreased by a factor of ten, from 1×10^5 tons yr^{-1} in the 1950's to less than 5×10^4 tons yr^{-1} in the 1970's; this has not improved since even though environmental protection measures are being implemented. In the Changjiang Estuary, the biomass of *Stolephorus*, a traditional commercial fish species in the region, has been considerably reduced after 1990 and shows almost no landing records in last 5-10 years; the catch of *Stolephorus* decreased from 560 tons yr^{-1} in the 1960s to 5-10 tons yr^{-1} in 1990; and the commercial fishing for crabs in this region has shut-down after 1980s and all catches of crabs were banned in 1990 owing to a population crash. Fish kills are now occurring nearly routinely in the Pearl River Estuary and in Hong Kong waters from hypoxia and harmful algae blooms (HAB). Pearl Harbor is degraded but much less seriously; 99% of the oysters died in 1972; however fish and invertebrates survived into the 1990s when water quality improved. Bangkok estuarine waters can, in the dry season, become anoxic and inhospitable to fish; the situation is severe but not hopeless in the upper Gulf of Thailand; the catch per effort decreased by a factor of ten from about 290 kg hr^{-1} in 1963 to 20-30 kg hr^{-1} in the 1990s; fishery production declined greatly from 1986 to 1995. Fish stocks in Ho Chi Minh City estuarine waters have virtually collapsed and only aquaculture remains. In Manila Bay the trawl catch per unit effort decreased from 46 kg hr^{-1} in 1947 to 10 kg hr^{-1} in 1993; the demersal biomass decreased from 4.61 mt km^{-2} in 1947 to 0.47 mt km^{-2} in 1993. Hypoxia events are now reported in Klang Harbour, clearly inhospitable to fish. In Singapore waters, marine fish species composition between 1934 and 1973 showed no loss of species but definitely less abundance; wild fisheries are now negligible; occasional fish kills have occurred from spills and anoxia especially in Johor Strait. Jakarta Bay has very little intact fisheries; the remaining fisheries are

E. Wolanski (ed.), The Environment in Asia Pacific Harbours, 489–493.

comprised of more opportunistic fish species that can exist in the now heavily polluted waters. By comparison Darwin Harbour fisheries are still rich but they are completely non-managed in a free-for-all philosophy as if there was no tomorrow; in 2000, 37% of Darwin residents spent some of their time fishing and one in every five resident households owned a pleasure boat used at least partly for recreational fishing; the total number of hours fished annually in 2000 was estimated to be 540,481 hours; some fish spawning aggregations have disappeared; the resulting ecosystem impact is unknown and basically not studied, and this lack of data precludes sustainable management.

There are other parameters of environmental degradation. These include in all harbours studied in this book some, but not all, of the following symptoms: harmful algae blooms, anoxia and hypoxia (oxygen depletion), and poisoning by fecal pollution, 4,4'- DDE and 4,4'-DDT, pesticides, PAHs and phenol, the production of toxins that accumulate through marine food chains to poison marine mammals, seabirds and humans. In extreme cases of grossly polluted, poorly flushed waters a parameter of pollution is simply gross stench.

Another measure of environmental degradation is a decrease in biodiversity. In the Pearl and Changjiang estuaries for instance, the number of species of plankton and benthos has decreased dramatically, the community composition of plankton and benthos is now much simpler, and the biodiversity is degraded.

In some cases such as the Changjiang Estuary the ecosystem has essentially collapsed; its historical role of providing multiple ecological services to society is changing to a simplified service system, e.g. the estuary essentially provides only land for settlement and waterways for transportation and trade.

Protecting endemic species from invasive species is low on the priority list of port operators. Invasive species are increasingly common, mainly introduced through ballast water in large bulk cargo carriers and vessel fouling, especially of slow moving barges and dry docks. These can overwhelm some endemic species.

2. THE BAGGAGE OF HISTORY

The historical practice in these harbours has been to combat human poverty and to develop economically at the accepted, explicit cost of environmental degradation. This policy appears still to prevail in several harbours such as the Pearl and Changjiang estuaries, Jakarta Bay, and Ho Chi Minh City.

In all harbours the environmental damage has been done. In some cases it is being vigorously addressed. In the case of Tokyo Bay, Singapore and Pearl Harbor, after these countries became wealthy and strictly controlled pollution, the waters became clear, giving the appearance of health; a high diversity of marine life is still present; this indicates that port waters can support marine species; however the ecosystem is severely depleted, it does not provide ecosystem services to maintain the quality of life that the population could enjoy if the marine environment was rich and healthy.

The key mechanisms preventing ecosystem recovery in wealthy countries' harbours appear to be habitat loss, habitat degradation and habitat modification. These key habitats are mangroves and salt marshes, mud flats, seagrass meadows, and coral reefs. The pace of converting marine habitats to meet development needs appears unsustainable. The key culprits appear to be seabed dredging, infilling, and the dumping of dredged spoils, and land reclamation for settlement, industry, harbours and aquaculture. Two other major culprits are over-fishing and the intentional or accidental introduction of species - and diseases – mainly from ships.

In less wealthy but emerging countries, there appears to be a will to restore environmental quality, such as is the case of Manila Bay; the practical means to do so are limited.

3. A PATHWAY TO ECOLOGICAL SUSTAINABILITY

The case studies in this book demonstrate that, using only strict pollution control measures, it is unrealistic to expect urbanised waters to provide the quality of life that a wealthy population expects. Because these waters are semi-enclosed and generally poorly flushed, there are no boundaries. Thus the ecosystem services provided by the waters will not be restored by relying only on zoning for industry, ports, urban areas, and some marine protected areas, and fishing quotas.

Remedial measures based on engineering and technological fix have not been successful in restoring the ecological processes of healthy, robust waters in harbours and the urbanised coast; they will not reinstate the full beneficial functions of this ecosystem; the impacts on the quality of life of the human population is severe.

As the case studies in this book demonstrate, engineering alone is not a solution, though engineers (and am I one) often justify their environmental impacts by the statement that it is 'the world's best practice' – this is a fallacy! There is no such established world's best practice for environmental management of harbours, as the case studies in this book demonstrate.

Instead, the successful management of the urbanised coast requires an ecosystem-based, basin-wide approach (eg Zalewski, 2000 and 2002; Wolanski et al., 2004; Norse and Crowder, 2005). This necessitates changing present practices by official institutions based on municipalities and counties as an administrative unit, and the narrowly-focused approaches of managers of specific activities (e.g. farming and fisheries, water resources, port operators, urban planners, wetlands management, nature conservationists). Without this change in thinking and management concept, these waters will continue to degrade, whatever 'integrated coastal management plans' are implemented. This necessitates the establishment of political authorities to coordinate all activities within the drainage basin, the urban, harbours, and industrial areas, and the waters. This necessitates political will and determination, and in most cases this is generally lacking as the issue of quality of life is often seen as non-urgent. A welcome exception in all these cases is Tokyo Bay where the recent legislation recognises that the whole drainage basin and the bay must be managed as an ecosystem. People living along its coastline are expected to use it wisely, and the wishes of the local people are respected. Some habitat restoration is under planning.

Habitat restoration to improve ecosystem abundance may not be possible in Singapore because it has literally run out-of-room.

In Hong Kong, the issues are largely still simply to use engineering solutions to prevent ecosystem collapse; at times the system is overwhelmed both by local pollution and pollution brought in from upstream, as well as habitat destruction. The system resembles that of the Changjiang Estuary, i.e. an area primarily seen as valuable for land reclamation and shipping – its ecosystem survival is still not a political priority although the situation is slowly improving.

In less developed countries, there is room for optimism in some cases, such as in Manila and further down the scale of practicality for Jakarta Bay, because political coordination and determination do show signs of emerging. In both cases, the authors of the chapters arrive at the same conclusions, namely that the key element that prevents ecologically sustainable development is the lack of recognition of the key role of ecohydrology. The river basin and the coastal waters where the harbour is located form an ecosystem. It can only be managed in an ecologically sustainable way by an independent port and bay management authority – as opposed to an advisory committee - that involves all stakeholders in industry, government and the community, and comprises an executive body.

Where the practice has been to scatter ports all over the area, such as in Ho Chi Minh City and the Peal Estuary, in a free-for-all development strategy, the political process that is necessary to restore the environment has been made harder and has barely started. The environmental costs are staggering.

Which brings us to the case of Darwin Harbour. Its ecosystem is pristine compared to all the other harbours described in this book. There are large-scale current and proposed developments including increasing human population and urbanisation, doubling the size of the LNG gas plant, another pipeline through the Harbour to the expanded gas plant, further clearing of mangroves, a 10-fold increase in the size of the port, sea cage aquaculture, a Helium plant, dredging for port expansion, waterfront developments, sand mining, and an increase in shipping. Untreated or primarily treated sewage is still dumped in the harbour. Major developments have proceeded with minimal environmental impact and remediation studies. Fishing is large-scale and practically unregulated. Some of these fisheries may survive longer simply because the habitats (e.g. the land and the mangroves) are protected by Aboriginal traditional owners on the west bank – this may raise a social inequity issue as the Aborigines may not benefit from the service they provide to the community by helping to provide fish because the fisheries laws make these fish free for all. Despite the potential of Darwin Harbour ecosystem to degrade, there does not appear to be a cohesive management plan backed by political will, to ensure long-term ecological sustainability. Each development is assessed in isolation. An ecosystem-ecohydrology based approach is needed to avoid repeating the experience in the Asia Pacific harbours described in this book – namely, once the environment is degraded, the ecological services to sustain the quality of life of the human population cannot be fully restored. As a parent I view natural resources as a bank account that we hold in trust for future generations. If the politicians fail in Darwin Harbour to learn from the mistakes of other Asia Pacific harbours, we leave our children with a serious debt to pay and we fail a critical test as a society.

4. THE ROLE OF SCIENCE

As demonstrated by the twelve harbours described in this book, engineering and technology by themselves do not provide a long-term solution to sustainable development. Statements in the engineering community of 'world's best practices' when discussing environmental sustainability of harbours are nonsense and not backed by facts as the case studies in this book illustrate.

Instead an ecohydrology-ecosystem approach to harbour development is needed. Science has key role to play to help develop this approach (e.g. Beach, 2002; Norse and Crowder, 2005). Besides providing knowledge in traditional disciplines, such as physical and biological oceanography, taxonomy, and toxicology, scientific research is also needed to explain the workings of the ecosystem and its response to human-induced stressors. It can thus quantify the effects of biota and biotic processes on mediating the urbanised waters' response to changing hydrology, sediment, pollutants and nutrient flux. When that knowledge is gathered, then science can, in collaboration with engineers and technologists, provide science-based remediation measures at the basin scale, with elements of ecohydrology, habitat manipulation, and phytotechnology at their core, to strengthen the ability of the biota to sustain and adapt to human-induced stresses.

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