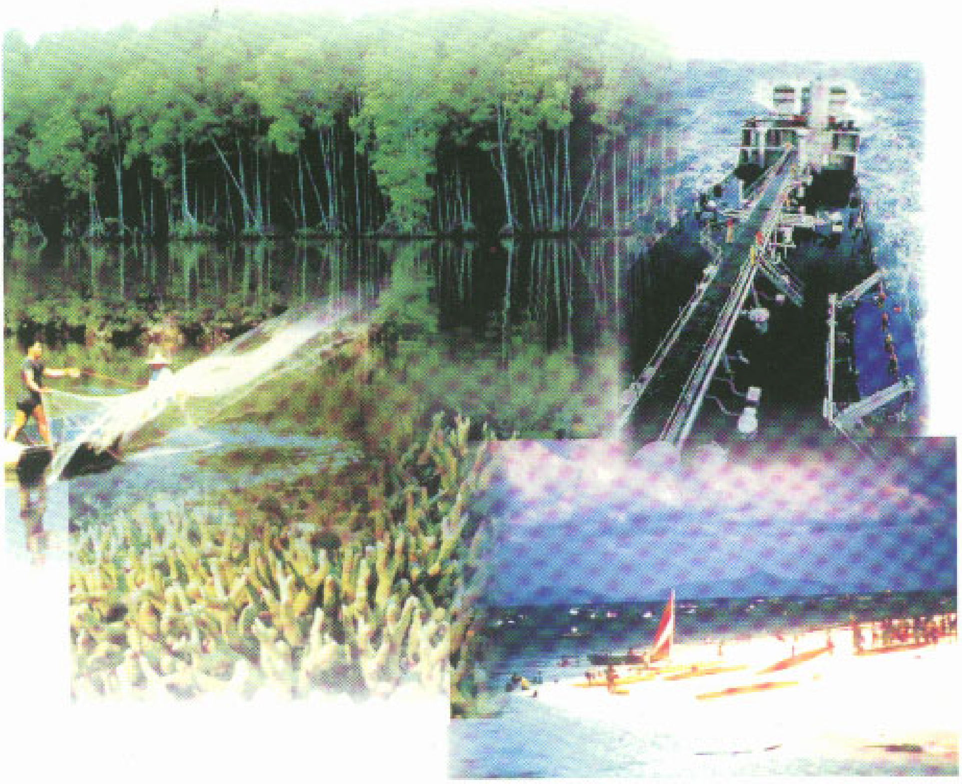
Malacca Straits: Refined Risk Assessment



GEF/UNDP/IMO Regional Programme for the Prevention and Management of Marine Pollution in the East Asian Seas

Malacca Straits: Refined Risk Assessment

1999

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of Marine Pollution in the East Asian Seas

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MISSION STATEMENT

The primary objective of the Global Environment Facility/United Nations Development Programme/International Maritime Organization Regional Programme for the Prevention and Management of Marine Pollution in the East Asian Seas is to support the efforts of the eleven (11) participating governments in the East Asian region to prevent and manage marine pollution at the national and subregional levels on a long-term and self-reliant basis. The 11 participating countries are: Brunei Darussalam, Cambodia, Democratic People's Republic of Korea, Indonesia, Malaysia, People's Republic of China, Republic of the Philippines, Republic of Korea, Singapore, Thailand and Vietnam. It is the Programme's vision that, through the concerted efforts of stakeholders to collectively address marine pollution arising from both land- and sea-based sources, adverse impacts of marine pollution can be prevented or minimized without compromising desired economic development.

The Programme framework is built upon innovative and effective schemes for marine pollution management, technical assistance in strategic maritime sectors of the region, and the identification and promotion of capability-building and investment opportunities for public agencies and the private sector. Specific Programme strategies are:

- Develop and demonstrate workable models on marine pollution reduction/prevention and risk management;
- Assist countries in developing the necessary legislation and technical capability to implement international conventions related to marine pollution;
- Strengthen institutional capacity to manage marine and coastal areas;
- Develop a regional network of stations for marine pollution monitoring;
- Promote public awareness on and participation in the prevention and abatement of marine pollution;
- Facilitate standardization and intercalibration of sampling and analytical techniques and environment impact assessment procedures; and
- Promote sustainable financing mechanisms for activities requiring long-term commitments.

The implementation of these strategies and activities will result in appropriate and effective policy, management and technological interventions at local, national and regional levels, contributing to the ultimate goal of reducing marine pollution in both coastal and international waters, over the longer term.

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GEF/UNDP/IMO Regional Programme
for the Prevention and Management
of Marine Pollution in the East Asian Seas

CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF BOXES

LIST OF ABBREVIATIONS AND ACRONYMS

ASEAN	-	Association of Southeast Asian Nations
BCF	-	bioconcentration factor
BOD	-	biochemical oxygen demand
BQ	-	background quotient; MEC/background contaminant concentration
COD	-	chemical oxygen demand
CPUE	-	catch per unit effort
EEC	-	European Economic Commission
EU	-	European Union
GESAMP	-	Group of Experts on the Scientific Aspects of Marine Pollution
GM	-	geometric mean
HELCOM	-	Helsinki Commission
LOEC	-	lowest observed effects concentrations
MAFF	-	Ministry of Agriculture, Fisheries and Food, UK
MEC	-	measured environmental concentration
MEHRA	-	marine environmental high risk areas
MEL	-	measured effects level
MPN	-	most probable number
NOAEL	-	no observed adverse effects level
NOEC	-	no observed effects concentration
PAH	-	polycyclic aromatic hydrocarbon
PEC	-	predicted environmental concentration
PEL	-	predicted effects level
PNAEL	-	predicted no adverse effects level
PNEC	-	predicted no-effects concentration
PNEL	-	predicted no-effects level
RDA	-	recommended daily allowance
RQ	-	risk quotient: MEC(or PEC)/STD (or Threshold)
SMEIS	-	Straits of Malacca Environmental Information System
STD	-	standard
STW	-	sewage treatment works
TBT	-	tributyltin
TDI	-	tolerable daily intake
TSS	-	total suspended solids
US FDA	-	United States Food and Drug Administration

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The Malacca Straits Demonstration Project is coordinated by Mr. S. Adrian Ross, Senior Programme Officer, GEF/UNDP/IMO Regional Programme for the Prevention and Management of Marine Pollution in the East Asian Seas.

EXECUTIVE SUMMARY AND RECOMMENDATIONS

The Malacca Straits Demonstration Project was implemented as part of the GEF/UNDP/IMO Regional Programme for the Prevention and Management of Marine Pollution in the East Asian Seas for the purpose of:

1. Demonstrating environmental risk assessment-risk management as a viable framework for managing both land- and sea-based sources of marine pollution in subregional sea areas; and
2. Packaging the approaches, methods and experience of the Malacca Straits for use elsewhere in the East Asian Seas region, where similar environmental management issues are apparent.

The definition of environmental risk is "*the likelihood that an environmental condition caused by human activity will cause harm to a target*". For the Malacca Straits project, the targets of interest were the Straits ecosystem and the people living in the coastal areas. In addition, the effects of marine pollution on the social welfare and economics of the littoral States were also evaluated.

The Malacca Straits risk assessment was conducted in two stages. First, an initial risk assessment was implemented as a screening mechanism for identifying priority environmental concerns on a Straits-wide basis, and the related data gaps and uncertainties. From the initial stage, the targets of interest were refocused to human health, habitats and commercial and non-commercial marine species. In addition, the refined risk assessment was conducted at two levels. Risks to the Straits as a whole were considered, in which the Straits were treated as a single compartment and a single average exposure concentration was estimated. For selected contaminants, risks to local areas within the Straits were estimated, by calculating local exposure concentrations in the vicinity of specific human activities or natural resources.

This report provides information on the rationale of environmental risk assessment, the methodology that has been developed and applied in the Malacca Straits initiative, the results of the work and recommendations for improving risk assessment as a management tool in the Malacca Straits. The report also delves into management interventions, and provides some conclusions and recommendations for strengthening marine pollution risk management in the Straits.

The following is a summary of the conclusions and recommendations from the Malacca Straits refined risk assessment. Appendix 1 contains the summary from the earlier initial risk assessment.

1) From state of the environment reports, risk assessment procedures were used to (a) ascribe likely causes to damage identified in various natural resources; and (b) define likely harm to ecological systems and human health from levels of contaminants either measured in or predicted for the Straits of Malacca. The former is

referred to as retrospective risk assessment and the latter as prospective risk assessment. Retrospective risk assessment involved insofar as it was possible a systematic analysis of evidence for both levels of damage and presumed causative agents. Prospective risk assessment involved a comparison of exposure concentrations with critical thresholds above which ecological or human health impacts were likely, and these were summarized in the form of risk quotients.

2) In terms of the retrospective risk assessment, there was evidence for decline in mangroves, peat swamp forests, coral reefs, seagrass beds and soft-bottom habitats. Much of this could be ascribed to habitat destruction but with the possible involvement of pollution as well. Coral reefs were probably being impacted largely from increased sediment loads that were in turn related to mangrove removal.

3) Similarly, reductions in fish stocks could be largely ascribed to the impacts of overfishing, but again pollution was probably a contributory factor, and loss of nursery sites as a result of mangrove and peat swamp removals was also of importance.

4) There was evidence of some gastrointestinal problems for human health that might be due to contamination of seafoods harvested from the Malacca Straits, but clearly a wide range of causal factors could have been operative here.

5) From a prospective risk assessment point of view, likely problems for harm to ecological systems were identified for various heavy metals in the water column and sediment, pesticides, particularly in sediments, and also possible problems arising from suspended solids. Problems arising from either TBT or nutrients were not identified, but both of these results were viewed with skeptism because specific responses in targets (i.e., respectively imposex in gastropods and eutrophication) were suggested in the retrospective studies.

6) Specific attention was given to problems arising from oil and hydrocarbons for ecological systems. The analyses were carried out both for long-term exposure to materials deriving from land- and sea-based sources and for short-term exposure following accidental spillages from oil tankers. On the former, evidences were found for likely harmful effects especially from water column exposures, and much of these could be ascribed to land-based industrial activities. On the latter, the likelihood of accident was calculated on the basis of historical experience and, further, how the likelihood of ecological impact could be assessed for particular circumstances was illustrated.

7) For human health prospective risk assessment, likely problems were identified for human health arising from the consumption of both metal- and pesticide-contaminated seafood. The extent of these risks varied as a result of differences in seafood consumption rates and from place to place and with the particular condition of subjects (i.e., with respect to reproductive condition and age). Coliform bacteria also presented problems, but largely for dermal exposure, and were therefore a hazard for bathers. The risk assessment gave no indication that health problems might arise from oil and hydrocarbon exposure, but in view of the high concentrations of oil and hydrocarbons throughout the Straits, it is believed that this should be given more careful consideration.

8) Uncertainty analyses were carried out for all the prospective risk assessments to varying levels of sophistication; i.e., ranging from simple inspection of the extent to which confidence limits around risk quotient values overlapped the critical value, to Monte Carlo resampling procedures, which allowed quantification of the probability that RQs exceeded critical values. Variability in all elements of the risk assessment–measured environmental concentrations, predicted environmental concentrations and critical threshold values–were found to be important to a greater or lesser extent in all the assessments.

9) On this basis, a number of general recommendations are made viz.:
    a) That regional agreement be reached between all littoral States on which standards should be used in future risk assessments.
    b) That coordinated regional monitoring programs be developed to enable high quality and mutually acceptable MECs to be available for future risk assessments.
    c) That regionally agreed exposure models be developed for coordinated use in future risk assessments.

10) On the basis of both retrospective and prospective assessments, a number of recommendations are made with regard to risk management:
    a) The development of an agreed and coordinated approach between littoral States to mangrove clearance.
    b) The development of rational implementation of controls on fishing intensity be agreed at a regional level.
    c) The identification of a number of sites where immediate management action is required with regard to heavy metals and oils and hydrocarbons.
    d) For human health protection, food contamination from metals deserves serious attention, and pesticides also require consideration. Here, monitoring for likely contamination should be more extensive and restrictions on where food organisms are collected need to be contemplated.
    e) For human exposure to sewage infection, immediate measures need to be taken to prevent exposure on the most contaminated beaches and with more long-term provision for improved sewage treatment.
    f) All of the above (i.e., 10a-e) bear primarily on the likelihood of risks arising from chronic exposure. Accidental exposures will lead to acute effects, and this is particularly evident with respect to oil spills. Management strategies here could be designed to be more proactive reducing the potential for contact between high risk vessels and particularly vulnerable habitats.

TERMS OF REFERENCE AND AIMS

Preamble

Describing and assessing states of the environment are central parts of environmental protection. Yet knowing what to measure and how to relate observed changes either to the consequences of some contaminating processes or to the implementation of some environmental protection measure is far from straightforward, largely because in a complex world, it is often hard to identify specific causes and effects. Several state of the environment reports have, nevertheless, been compiled at various scales, from global to regional, national and even very local with environmental impact assessments at particular sites invariably involving a report on state of the environment before a project begins. Of particular note here, though, are the state of the environment reports compiled for other marine bodies such as the North Sea and Baltic Sea (GESAMP, 1990; HELCOM, 1990; OSPARCOM, 1993).

All these reports tend to consider aspects of the environment that are conveniently measured and to judge state either in terms of the presence of hazardous substances or changes in selected variables through time. Again, though, a problem with the latter approach is that it is often hard to pick up relevant and significant changes in a naturally dynamic world. Noise often overwhelms the observations; and anyway change in itself is only of importance if it is counter to the natural dynamics.

The *Malacca Straits Environmental Profile* (Chua et al., 1997) is a 'snapshot' of the state of the environment of the Straits. The document was used as the basis for an initial risk assessment. From the initial risk assessment (including the identification of data gaps), additional follow-up documents (listed in Appendix 2) were supplied by the littoral States which contributed to the completion of a refined risk assessment, with particular focus on selected key issues.

The risk assessment approach implies the presumption that it is possible to specify the likely consequences for human health and ecological systems of human influences, often with regard to the chemical quality of the environment. It is further often presumed that there will be states of the environment, arising out of human influences in terms of processes and emissions that are associated with low probability of harm to human health and ecosystems. This implies that human activities can be managed to achieve these levels, often without the need to impose zero emission requirements. There is, therefore, an important distinction to be made between contamination (the presence of a substance of human origin in the environment) and pollution (the presence of a substance at levels sufficient to cause adverse effects). As a further step, risk/benefit approaches which are touched upon in this report, recognize that environmental protection measures while bringing benefits to the environment can bring costs to the economy. The possibility of taking these considerations into account in establishing appropriate control measures is outlined.

Risk and risk/benefit approaches also require more precision in defining what it is that risk is assessed for and hence what the targets are, what endpoints are therefore important, and consequently what should be monitored in state of the environment reports. This is not just a matter for science–in defining natural states–but for society at large–in defining what it is about those natural states that public wants and is prepared to protect (Forbes and Forbes, 1994).

In the report that follows, the second to the sixth sections provide background, with the fourth section translating the principles that are discussed into the approach that has been used in the assessment of information, as provided in the *Malacca Straits Environmental Profile* and supplementary documents. The initial and refined risk assessments are carried out in the seventh to the tenth sections. Conclusions and recommendations are given in the last section. In carrying out this exercise, there were three main aims: (1) to illustrate how to apply the risk assessment approach; (2) to identify circumstances of high risk that should invite urgent attention; (3) to identify areas of need in terms of information, measurement and possibly research. Because of the nature of many of the observations and the limitations on key assumptions that are employed as a basis of the work, a number of conclusions may need revision in the light of more information. The analyses are as systematic and as transparent as possible to facilitate future amendments.

Project Outline

Objectives

1) To complete an initial risk assessment, utilizing available information on sources, exposure and effects of land-based and sea-based activities, and the pollutants derived therefrom, on the living and non-living resources of the Straits of Malacca.

2) To complete a refined risk analysis of land- and sea-based sources of pollution and their effects on living and non-living resources in the Straits, using both the results of the initial risk assessment and updated information from the three littoral States to produce a comprehensive document on environmental risk assessment of the Malacca Straits.

The refined risk assessment focused on two priority activities and contaminants in the Malacca Straits, as identified in the initial risk assessment, namely:

a) Human health effects, by exploring further:

    - fish/seafood consumption
    - contamination of fish/seafood by metals, pesticides and hydrocarbons

b) Ecological effects, by exploring further measured environmental concentrations for hydrocarbons and hydrocarbon composition, and their impact on the ecosystem.

Only available secondary data from institutions within the three littoral States, IMO and regional bodies were input to the project (listed in Appendix 2). There was no primary data collection.

Work program outline for the initial risk assessment

The work program for the initial risk assessment consisted of:

1) Preparation of a draft report of the initial risk characterization/uncertainty analysis of the Malacca Straits, highlighting:

    a) major polluting sources and activities (land- and sea-based) in the Malacca Straits and their effects on living and non-living resources;

    b) delineation of the assessment endpoints that are the most significant indicators of ecological, human health and societal risk resulting from pollutive land- and sea-based activities;

    c) spatial and temporal scales of the assessment;

    d) important interactions between land- and sea-based activities and interactions with living and non-living resources in and along the Straits;

    e) combined effects of multiple and diverse stresses on the ecology of the Straits; and

    f) the systematic effect of a catastrophic event, namely a shipping accident and the subsequent spillage of oil and/or dangerous chemicals, on the ecology of the Straits.

2) Identification of data gaps/uncertainties in the *Malacca Straits Environmental Profile* which need to be addressed as part of a more comprehensive risk characterization/estimation on the Malacca Straits; and

3) Formulation of an action plan for completing a comprehensive risk assessment of the Malacca Straits, utilizing available expertise and resources within the littoral States and the region, and leading to the development of a risk management program for the subregion.

Work program outline for the refined risk assessment

The work program for the refined risk assessment consisted of the following steps:

1) Review and analyze available data as provided by the littoral States, retrospectively and prospectively, with a view to updating and/or verifying the methodologies, conclusions and recommendations of the initial risk assessment, the primary focus being human health (i.e., contamination and consumption of fish/seafood) and ecological (i.e., hydrocarbon impact on the ecosystem) effects;

2) Employ more sophisticated pollutant fate and oil spill trajectory models, developed as part of the follow-on work in the Malacca Straits Demonstration Project in the prospective analysis. Develop a series of scenarios that can be run using the models or, alternatively, the scenarios analyzed as part of the model development and demonstration; and

3) Test techniques for improving uncertainty analyses and report the results.

SOURCES OF INFORMATION

The material upon which the risk assessment was based has been largely from the *Malacca Straits Environmental Profile* (Chua et al., 1997), hereafter referred to as the *Profile* and documents with more up-to-date information as listed in Appendix 2. Reference to material from these sources shall be specified routinely as *Profile* Table, *Profile* Figure, *Profile* p., and/or *Appendix Document* followed by the list number (in Roman). Source references were not cited again when they are listed either in the *Profile* or the *Appendix Documents*. Much of this information has been incorporated into a comprehensive regional database and GIS for the Straits. This is now known as the Straits of Malacca Environmental Information System (SMEIS), a Microsoft Windows based software system. A description of the structure and contents of SMEIS as well as a user manual can be found in the two documents referred to at the end of our *Appendix Document* list (i.e., numbers XIV and XV).

Reliance has been made on the data, and to some extent standards, compiled within the provided literature. The presumption has therefore been that the data used within the risk assessment were reliable. There are techniques for assessing the reliability of data of these kinds; in particular assessing the methodology and techniques used in their production (e.g., was good laboratory practice followed?), sampling and experimental design (e.g., was there sufficient replication?), statistical analysis and interpretation of results (e.g., were appropriate tests used and appropriate levels of significance applied?). These are described in more detail in the *Environmental Risk Assessment Manual: A Practical Guide for Tropical Ecosystems* (MPP-EAS, 1999a). Future risk assessments will need to address these issues more systematically and rigorously. Because of the methodology, risk assessment is very dependent upon the reliability of standards, and which shall be discussed again in this report.

THE STRAITS

This section provides a brief description of the geography, ecology and socioeconomic aspects of the Straits as background to the risk assessment. It therefore emphasizes features likely to be of importance in influencing exposure and effects scenarios.

Bounded by three littoral States with broadly differing economies, the Straits provide a natural channel between the Indian and Pacific Oceans. In consequence they are the second busiest shipping lane in the world, currently with about 300 vessels passing through per day (*Profile* Table 4-6). At the same time, the euryhaline conditions, rich nutrient levels, shelter from strong currents and wave action, together with high but uniform temperatures (see below) and adequate tidal flushing contribute to high biological productivity and diversity with a rich mix of fauna and flora from both the Indian Ocean and Pacific Ocean (*Profile* chap. 2).

Many of these natural biological resources are exploited along both coasts of the Straits. Chief among these are:

1) Fisheries that include both demersal and pelagic species and involve a variety of techniques; with most fishing intensity apparently concentrated in the NW half of the Straits;

2) Mangroves that are exploited extensively along the entire lengths of the east and west coasts for timber, and that are also being removed to make way for aquaculture. Most of the mangrove swamps occur on the Indonesian side of the Straits (about 80%). Also of importance are seagrass beds that are abundant but patchy throughout, and corals that are patchy and not very abundant in the Straits themselves. The mangroves and seagrass beds provide nursery grounds for many species of fish, including commercially exploited ones, and so there is a relationship between the availability of these habitats, fish stocks and sustainable yield; and

3) Extensive aquaculture on both east and west coasts also depends upon sound ecology, while at the same time potentially causing problems for the environment through the release of organic wastes and chemicals.

The human population densities on either side of the Straits are similar (about 11 million along the west, about 10 million along the east and about 3 million in Singapore; *Profile* Table 7-1), but the major forms of employment are different, with a predominant emphasis on agriculture and fisheries together with derivative industries and those based on natural resources in Indonesia, a mix of agriculture, fisheries and various heavy and light manufacturing industries in Malaysia, and an almost exclusive emphasis on manufacturing and commercial activities in Singapore. The provision of sewage facilities also differs appreciably between the littoral States, being very limited on the Indonesian side, limited on the Malaysian side, but very complete in Singapore.

There are roughly similar numbers of river catchments on both Indonesian and Malaysian coasts and so, with similar amounts of rainfall, the presumption must be that there are similar volume outflows and runoffs from both coasts (a figure of 90 million m^3 per year is quoted for the Indonesian side; *Profile* Table 2-2). However, the quality of these inputs is likely to differ with those from the west coast being influenced by the agricultural economy, and those from the east coast having more of an industrial quality.

Land use activities, together with mangrove removal are contributing to increased erosion, especially in the NW half of the Straits and these, together with contributions from river loads, agricultural runoff and aquaculture, are leading to increased total suspended solids in the water column of the Straits and sedimentation with consequent impact on mangroves, seagrass beds and corals through increased oxygen depletion, light attenuation and physical cover.

It was calculated that the total volume of the Straits was about $10^{12}m^3$, so dilution and removal of contaminant loads associated with flushing could be considerable. However, water movements are complex, with dominant surface movements from SE to NW. Movements of sediments, though, at least on the Indonesian side, seem to be in the opposite direction with erosion in the NW half and accretion in the SW half.

The high but constant temperatures (26-30°C; *Profile* chap. 2) within the Straits are likely to have implications for both exposure to and effects of contaminants. On the exposure side, the high temperatures may mean increased rates of biodegradation and hence losses of contaminants (compared to temperate systems). On the effects side, high temperatures are likely to mean relatively rapid rates of contaminant uptake and high levels of metabolism as compared with temperate conditions, under which most published ecotoxicological effects have been measured.

The episodic rainfall events of high intensity but short duration (*Profile* chap. 2) are likely to have three consequences for exposure and effects scenarios. First, the episodes of high rainfall are likely to be associated with considerable contamination from storm water runoff, involving both dissolved and particulate materials. Second, the dilution effect on salinity, causing values to fall to as low as 6.8‰ (*Profile* chap. 7) is likely to lead to osmotic stress in marine species which may exacerbate the effects of stress arising from contaminant exposure. Third, reducing salinity will alter the bioavailability of many contaminants (e.g., the bioavailability of some heavy metals, such as Cd, increases with decreasing salinity as an increasing fraction of dissolved metal exists as free ions; Forbes, 1991).

In summary, the Straits represent a unique ecological system with high productivity and diversity and a rich mix of fauna and flora. The intricate hydrodynamics together with complex interactions within and between the water body and land-based activities complicate the understanding of the effects of human activities on the Straits. Following sections on definition of terms and general approach, these complex interactions are defined further before proceeding to the detailed risk assessments.

DEFINITION OF KEY TERMS

Environmental risk assessment involves estimating the likelihood of harm being done to human health and/or ecosystems through factors emanating from human activities that reach their targets via the natural environment. Hence, it usually combines an understanding of the potential that factors have to cause harm (hazard identification) with an understanding of the likely levels of exposure in targets (exposure assessment).

A summary of definitions of all key terms, modified from van Leeuwen and Hermens (1995) is given in Box 1.

There are two kinds of questions that can be addressed using the systematic approach of environmental risk assessment:

1) What evidence is there of problems with human health, habitats and/or species in particular places and what are the likely causes? This is known as the **retrospective approach** and is akin to epidemiology; and

2) What problems might be caused to human health, habitats and species by conditions that exist now or in the future? This is known as the **prospective approach**.

Box 1. Key Terms Used in Risk Assessment.

Effects assessment - The component of a risk analysis concerned with quantifying the manner in which the frequency and intensity of effects increase with increasing exposure to a substance.

Exposure assessment - The component of a risk analysis that estimates the emissions, pathways and rates of movement of a chemical in the environment, and its transformation or degradation, in order to estimate the concentrations/doses to which the system of interest may be exposed.

Hazard identification - Identification of the adverse effects that a substance has an inherent capacity to cause, or in certain cases, the assessment of a particular effect. It includes the identification of target populations and conditions of exposure.

Risk - The probability of an adverse effect on humans or ecological systems resulting from a given exposure to a substance. It is usually expressed as the probability of an adverse effect occurring, e.g., the expected ratio between the number of individuals that would experience an adverse effect in a given time and the total number of individuals exposed to the risk factor.

Risk assessment - A process that entails some or all of the following elements: hazard identification, effects assessment, exposure assessment and risk characterization. It is the identification and quantification of the risk resulting from a specific use or occurrence of a chemical including the determination of exposure/dose-response relationships and the identification of target populations. It may range from largely qualitative (for situations in which data are limited) to fully quantitative (when enough information is available so that probabilities can be calculated).

Clearly, the two approaches are related in that prospective analyses provide a causal basis for assertions made in retrospective analyses, and retrospective analyses can provide a check on the predictions for prospective analyses and indeed help to define appropriate issues for prospective analyses.

Risk assessment ought therefore to start by identifying what entities are a cause for concern and hence are the objects of interest and ultimately of protection. These define the **assessment endpoints**. For example, if the interest is in a particular species and its likelihood of extinction, then the assessment endpoints could be in terms of population densities of that species and the population dynamics controlling them. But these properties are often difficult to address directly, so more often risk would be expressed in terms of levels of contaminants known to cause adverse effects in standard ecotoxicological test systems. These measures, that act as surrogates for the entity that is of prime interest, are known as **measurement endpoints**.

For prospective, but especially retrospective studies, a range of measurement endpoints can be used from ecosystem to molecular levels. Measurements at suborganismic levels are often referred to as *biomarkers* and can be of considerable use as indicators of exposure from both an ecological and human health perspective (IEH, 1996). However, to be of use in risk assessment they have to be demonstrably relevant to the assessment endpoints identified for the targets. Rarely is this test of relevance applied. We would counsel against the indiscriminate use of biomarkers in a risk assessment context simply on the grounds of sensitivity and convenience (Forbes and Forbes, 1994).

A detailed methodology for both prospective and retrospective risk assessment is given in MPP-EAS (1999a).

Risk assessment can be carried out to various levels of detail and sophistication, from a purely qualitative level that involves descriptive techniques, to semiquantitative scoring systems, to a fully quantitative level that involves rigorous probabilistic statements over specified time frames (Box 2). Considerations of geographical scale are also important: the interest may be in very localized conditions and targets, regional ones or global ones. Thus, the concern might be with a localized population or habitat downstream of a particular industrial emission site, all populations or habitats in a subscribed region such as the Straits, or populations and habitats distributed on a global scale.

Assessments of risk provide a likelihood of occurrence of some harm on the basis of an understanding of all the variables involved. Rarely, however, is there complete understanding, and so there are uncertainties about the likelihoods that arise out of the analysis. Uncertainty analysis involves estimating the degree of variability in estimations of the probability of effects, which again can be carried out either qualitatively (describing where the uncertainties are) or quantitatively (using modeling to compute the range of possible outcomes that might arise from random variation in the variables of the risk assessment).

Box 2. Some Examples of Different Approaches to Risk Assessment.

Assessment of risks involves combining understanding of hazard with exposure (see text). Here are some examples of how this can be achieved.

1. Scores

In an environmental management system, managers are asked to assess each aspect of their production line for potential to cause environmental problems (=hazard), and the extent to which their systems and procedures would prevent this (=exposure). Using tables, each is scored 1 (good) to 5 (bad), and scores are combined by multiplication to give indices of risks of problems from the business to the environment: 25=very high, 1=very low. (Calow, P. and Streatfeild, C. 1995. *DIY Environmental Risk Profile*. Sheffield Regional Green Business Club, Sheffield, UK).

2. Ratios

These compare estimates of environmental exposure levels with estimates of likely effect levels. Then a ratio of one over the other gives an index of risk. The bigger or smaller the ratio, the greater or lesser the chances of harm. Whereas the probability of a ratio exceeding a selected value can be quantified, precise probabilities of harm occurring cannot be specified. This approach is used for the risk assessment of the Malacca Straits.

3. Probabilities

If one can specify the frequency distributions of exposure concentrations and of effect concentrations, then precise probability statements of effects can be computed from the extent that one distribution overlaps the other. If the effects are in terms of mortalities in populations, or species extinctions, or impairment of ecosystem functions, the probability statements would respectively be as follows: P of population size reduction of a particular magnitude; P of reduction in biodiversity (loss of a certain number of species); P of reduction in energy flow or cycling of matter, or rate of decomposition etc. within an ecosystem. Here the P values mean probability of effect and could be expressed as fractions or percentages. Probabilistic assessments are the ideal, but are rarely achievable due to lack of data and/or understanding.

Texts reviewing these and other techniques include:

Calabrese, E.J. and Baldwin, L.A. 1993. *Performing Ecological Risk Assessments*. Lewis Publishers, Chelsea, MI, USA.

Rodricks, J.V. 1992. *Calculated Risks. The Toxicity and Human Health Risks of Chemicals in our Environment.* Cambridge University Press, Cambridge, UK.

Suter, G.W. 1993. *Ecological Risk Assessment.* Lewis Publishers, Boca Raton, USA.

THE APPROACH HERE

Based largely on available information in the *Profile* and *Appendix Documents*, both retrospective and prospective analyses were carried out addressing respectively the following questions:

1) What evidence is there for problems with human health, habitats and species (including commercial fish stocks) in the Straits? And what are the likely causes?

2) What problems might be caused to human health, habitats and species by conditions that exist now or in the future?

The main categories of targets in these contexts were:

1) Human health;
2) Habitats (i.e., mangroves, peat swamps, seagrass beds, coral reefs, soft-bottom habitats); and
3) Species (i.e., commercial and non-commercial marine species).

Appropriate assessment and measurement endpoints were identified. The general philosophy was to identify systematically each of the two main elements of risk: potential harm (H) and likelihood of exposure to potential harm (E), such that Risk = f(H)(E), where f means 'function of'.

The differences between different scales of risk were distinguished. In particular, risks to the Straits as a whole were considered (in which the Straits were treated as a single compartment and a single average exposure concentration for the entire Straits was estimated) and, for selected contaminants, risks to local areas within the Straits (in which, for example, local exposure concentrations in the vicinity of individual rivers were estimated).

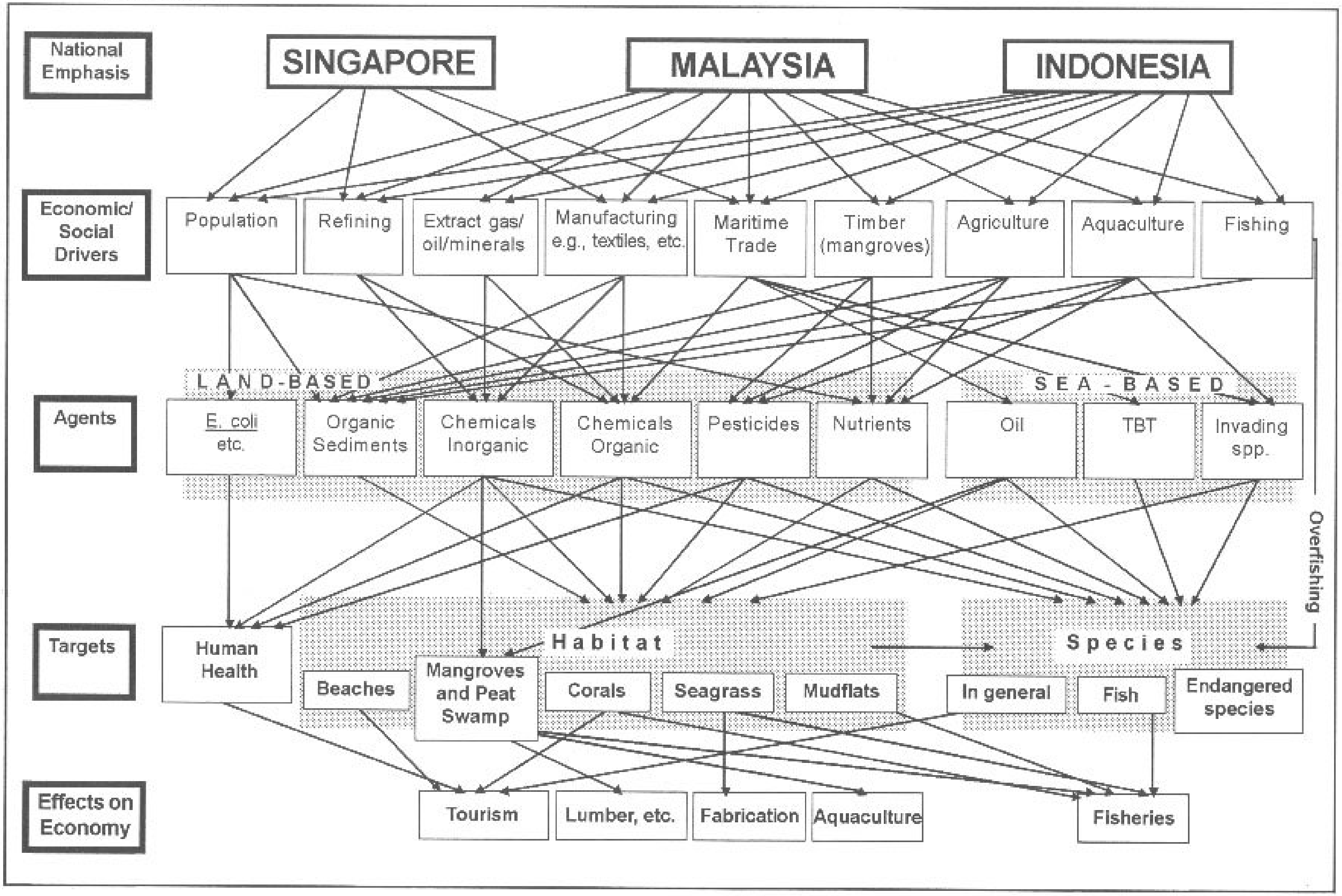
Uncertainty assessments were carried out. These involved both qualitative and more quantitative methods. For a more detailed description and explanation of uncertainty analysis, see MPP-EAS (1999a).

Societal risks were dealt with separately. These involved considering how environmental degradation and its control impact the economy. This involved risk-benefit analyses that draw upon the risk assessments; but they might also influence which risk assessments should be done as societal priorities. They are also key in risk management.

RISK PATHWAYS

For perspective, a qualitative indication of risk pathways was considered to draw attention to key issues. The risk pathways in Figure 1 illustrate the complex relationships between the potential causes of problems for human health and the environment and their consequences in the Straits. The sources of hazards are ultimately related to economic and other social drivers that are non-uniformly distributed among the littoral States. The consequences of pollution will have knock-on effects to the economy, again not evenly among the littoral States. But equally any controls are likely to have impacts on the economies of all that depend on the Straits, both within and outside the littoral community. These considerations ought not to influence the way the risk assessment is carried out, but they may influence judgments about priorities for action and hence at what issues the risk assessment is directed. Ultimately they will influence what management actions are taken, when it will be important to weigh benefits to human health and the environment with costs to the economy. These considerations will never be far from an analysis of complex risk pathways of high economic importance, involving a range of social, national, governmental and commercial interests.

Figure 1. Risk Pathways Illustrating Relationships Between Potential Causes of Human Health and Environmental Problems and Their Consequences in the Straits.



RETROSPECTIVE ASSESSMENT

Introduction

The key ingredients of a retrospective risk assessment are that:

1) targets and endpoints should be identified as precisely as possible;
2) significant adverse changes should therefore be identified;
3) possible causes of these changes should be identified; and
4) possible consequences of the changes for ecosystems and human welfare should be identified.

The distinguishing features of the approach are that it should be systematic and transparent. In this section, therefore, this discipline has been applied to information provided for habitats and species. For each, the summary includes: evidence for decline, attributed causes and likely consequences. 'Attributed' is emphasized because in this section, those views presented on causation are expressed in the *Profile* and the *Appendix Documents*. With retrospective assessment, views about causation are always based more on expert judgment and weight of evidence than is the case in the experimental sciences. Nevertheless, there are tests that can be applied to improve confidence in these kinds of approaches (Suter, 1993; MPP-EAS, 1999a), but due to the lack of detail in the data provided here, these have not been applied. Therefore, the attributed causes are treated as hypotheses that will need to be considered further in the light of the prospective analysis presented later and the availability of more precise information. In what follows, a distinction between habitats and biodiversity is maintained for convenience. Clearly the two are closely interrelated.

Habitats

Assessment here is usually in terms of the extent and quality of living space for dependent species. Extent can be further classified into numerical (i.e., numbers of patches) and areal units. When the habitat matrix is biological, its quality is measured in terms of the diversity of species and/or the health of the constituent organisms. Otherwise, it is measured in terms of ability to support usual ecological and human requirements. Below, each of the main habitats listed in the *Profile* and *Appendix Documents* are considered systematically in terms of: evidence for decline, attributed causes and likely consequences. A summary of the evidence available for declines in key habitats and their ecological and economic consequences is provided in Table 1. The analysis is entirely qualitative, but indicates the relative importance among habitats and littoral States.

Mangroves

Evidence for decline

From *Profile* Table 3.17 the area occupied by mangrove forests along the Straits is 386,100 ha in Indonesia (77.5% of total mangrove area), 111,409 ha in Malaysia (22.4% of total mangrove area) and 600 ha in Singapore (0.1% of total mangrove area). There is indisputable evidence for decreases in the total area occupied by mangroves in all three littoral States, much of which appears to be due to intentional exploitation or removal of the mangroves. For Indonesia, it has been estimated that 55% of the original mangrove areas in Sumatra remained by 1987 and only 29% by 1993 (*Profile* chap. 3). For Malaysia 17% of the mangrove area was lost between 1965 and 1985 (*Profile* chap. 3); another estimate of loss is around 35% (*Profile* chap. 7; see also *Appendix Document* I-M). For Singapore, the percentage of the coastline occupied by mangroves has declined from 10-13% in 1819 to 0.5% in 1993 (*Profile* chap. 3). Approximately 81% of the area occupied by mangrove forests in Singapore was lost during the last 20 years (*Profile* chap. 2).

Table 1. Retrospective Analysis of Declines in Key Habitats for the Malacca Straits.

Habitat Type	Areal Extent	Decrease in Quantity	Decrease in Quality	Ecological Consequences	Economic Consequences
Mangroves	Large	Large	Moderate [S]	***	**
Peat swamps	Large	Large	NI	***	**
Coral reefs	Small	NI	Moderate - Large	**	*
Seagrass beds	Moderate	NI	Moderate [S]	**	*
Soft bottoms	Large	No Decrease	Moderate	**	**

Areal extent is an estimation of the relative abundance of each habitat type as large, moderate, or small; evidence reported in the *Profile* for decreases in habitat quantity (i.e., areal extent) and quality indicate a large decrease, moderate decrease, minimal decrease or no decrease. Judgments on the relative seriousness of consequences for the ecology of the Straits or the economies of the littoral States are indicated by number of stars (more stars = more seriousness). NI indicates that no information was provided in the *Profile*. A superscript 'S' indicates information from Singapore only.

Several species of mangrove are listed in *Profile* Table 3-18 and the *Appendix Documents* (see especially VI, Table 7.2), but there is little information (and no quantitative estimates) of any changes in species diversity.

Attributed causes

On the Indonesian side of the Straits, the major cause of mangrove decline is clearance for brackish water ponds (tambaks) (*Profile* chap. 3) and overexploitation for timber and charcoal (*Appendix Document* I-I). On the Malaysian side and for Singapore, the main cause of mangrove loss has been clearance for development (*Profile* chap. 3). Other major causes of mangrove loss are: sedimentation (due to poor upland management) and pollution (e.g., from pesticides, oil, untreated sewage, industrial discharges) (*Profile* Table 2-6). According to a review by Peters et al. (1997), mangroves in general are not very susceptible to heavy metals (because they are immobilized as sulfides in the anaerobic sediments), can be very sensitive to oil spills and are also susceptible to herbicides.

Consequences

The destruction of mangrove forests has resulted in: (1) reduced protection from coastal erosion; (2) reduced protection from floods and typhoons; (3) reduced nursery grounds for commercial and non-commercial fish and invertebrates, which potentially has economic implications, as correlations have been found between the extent of mangroves and fisheries yield (*Profile* Figures 3.20 & 3.21); (4) a loss of critical habitat for endangered species and for conserving biodiversity (*Profile* Table 2-12 & chap. 3); and (5) possibly economic consequences for the timber industry (although these are limited, *Profile* Figure 3.18).

In conclusion, the greatest risks to the mangrove forests are associated with intentional clearance of the mangrove areas for other purposes. The area of the Straits at greatest risk lies along the Indonesian side, along which the mangroves constitute a relatively large area of coastline, which, according to the figures above, appear to be experiencing the greatest rates of decline, and for which management programs to date appear to be relatively ineffective (e.g., compared to Malaysia; *Profile* chap. 3).

Peat swamp forests

Evidence for decline

The area occupied by peat swamps in Sumatra has decreased from an original area of 7.3-9.7 million hectares (about 25% of all tropical peat lands; *Profile* chap. 2) to approximately 3.6 million ha (or approximately a 50% reduction). The area occupied by peat swamps along the west coast of Peninsular Malaysia is approximately 299,145 ha, with about 77% of this area described as 'disturbed and logged-over'

(*Profile* chap. 2). There were no exact figures provided in the *Profile* of the area occupied by peat swamps in Singapore (*Profile* Figure 2-9). However since they typically occur in connection with mangroves, this area is relatively small.

Attributed causes

At least in Indonesia, losses of peat swamps have occurred largely from logging (there are many commercially valuable tree species), transmigration programs and land conversion to rice, palm and coconut plantations (*Profile* chap. 3).

Consequences

The consequences of peat swamp loss are likely to be similar to those for mangroves (see above) with which they form a common ecosystem. In particular, the high biodiversity of peat swamps has been emphasized (*Profile* chap. 2).

Coral reefs

Evidence for decline

Coral reefs are found in smaller patches than in other areas in the ASEAN Region (*Profile* chap. 2). However, there were neither estimates provided in the *Profile* of the total area occupied by coral reefs in the Straits nor of losses of coral reef area. Estimates of coral reef condition for Indonesia did not include reefs from the Riau Archipelago (where most of the Indonesian reefs in the Straits are concentrated), but concluded that 42% of Indonesian coral reefs as a whole were in 'poor' condition, 29% were in 'fair' condition, 24% were in 'good' condition and only 5% were in 'excellent' condition. *Appendix Document* I-I indicates that there is better coral development in Eastern Indonesia compared to Western Indonesia due to more favorable growth conditions. In Riau, the best coral development occurs in the eastern entrance to the Straits with > 50,000 ha (50-70% were in 'good' condition). According to *Appendix Document* I-I, despite turbid water, Malacca reefs are growing relatively well in the Riau Province. Fringing coral reef flats of Cape Rachado represent the only surviving and growing coral along the coastline in Peninsular Malaysia (*Appendix Document* IV). Coral reefs are found only in the South Islands of Singapore (*Appendix Document* VIII). For Malaysian coral reefs in the Straits, most were rated as 'fair' (i.e., defined as having between 24 and 49.9% live coral cover). None of the stations received an 'excellent' rating (i.e., 75-100% live coral cover). Stations around the island of Renggis were rated as 'poor' (i.e., 0-24.9% live coral cover), whereas corals around the islands of Perhentian, Redang and Tenggol were rated as good (i.e., 50-74.9% live coral cover) (*Profile* Table 2-8). Singapore's coral reefs were described as 'among the most stressed in Asia' (*Profile* chap. 2). Species diversities in the coral reefs were reported in *Appendix Document* I-I, but there were no analyses of changing diversity.

Attributed causes

For Malaysia, sedimentation was rated as the greatest cause of coral reef decline, followed by fishing intensity and population pressures, then fishing damage and pollution from various sources (*Profile* Table 2-9). For Singapore, massive land reclamation has been cited as the most serious cause of coral reef decline (*Profile* chap. 2). Pollution from metals, oil spills and pesticides can have adverse effects on corals (Peters et al., 1997).

Consequences

The consequences of coral reef loss include reduced physical protection (and hence increased erosion) of shorelines, loss of biodiversity (some of which has commercial value), reduced fishery production and economic losses from reduced tourism (*Profile* Table 3-26).

Seagrass beds

Evidence for decline

The distribution of seagrass beds along the Malacca Straits is reported as less extensive than in other ASEAN waters (*Profile* chap. 2), but no quantitative data on areal coverage (or losses thereof) were provided in the *Profile*. Seagrass no longer forms extensive beds in Singapore, but occurs in isolated patches (*Appendix Document* VIII). Of a worldwide total of about 50 known seagrass species, 12 were recorded as occurring in Indonesia, 9 along the west coast of Peninsular Malaysia and 11 in Singapore (apparently in the late 1950s; *Profile* Table 2-10). The number of species in Singapore had declined to 7 by the 1990s (*Profile* chap. 7).

Attributed causes

The primary cause of seagrass decline appears to be from intentional destruction for conversion to coastal aquaculture (*Profile* chap. 2). Other major causes of loss include natural disasters (such as storms and disease), deposits of mining spoils and tailings, excessive deposition of silt in association with deforestation and blast fishing (*Profile* chap. 2 & Table 3-26). Pollution from metals, oil spills and pesticides can have adverse effects on seagrass beds as well (Peters et al., 1997).

Consequences

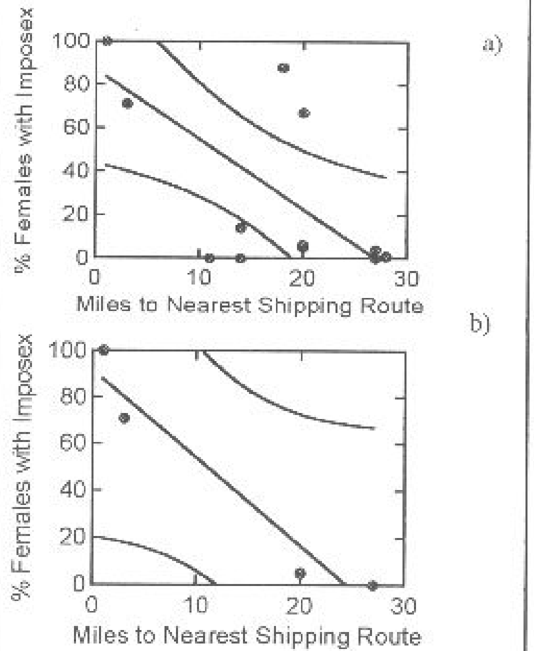
The most important consequences resulting from the loss of seagrass beds are a reduction in buffering of wave action (possibly leading to increased coastal erosion), reduced stabilization of sediment (with corresponding negative impacts on nearby coral reefs and mangroves), reduced biodiversity, loss of harvestable invertebrates, macroalgae and grass, and loss of nursery grounds for fishes including some of commercial importance.

Soft-bottom habitats

Evidence for decline

The area of the Straits covered by sandy and muddy bottoms is reported as 'extensive' (*Profile* chap. 2), but no exact figures for areal coverage were given in the *Profile* (cf. *Profile* Figure 2-18). There is little evidence that the total area of coverage of soft-bottom habitats is declining. Changes of concern are mainly in terms of the quality of this habitat particularly with regard to its ability to support commercial and non-commercial species. No quantitative estimates of the diversity or density/biomass of benthic species were provided in the *Profile*. *Appendix Document* XII provides measures of soft-bottom species diversity, but not any information on changes over time. However, evidence for a decline in the quality of soft-bottom habitats is provided by examination of effects on female reproductive systems in gastropods in terms of percent female imposex, possibly caused by TBT pollution (*Profile* Table 7-26). There is a significant negative correlation between percent imposex and distance to the nearest shipping route (Figures 2a & b).

Figure 2. Incidence of Imposex in Relation to Distance from the Nearest Shipping Channel.



a) For all species shown in *Profile* Table 7-26; Spearman rank correlation coefficient (r_s)=0.641, P<0.05, df=12; Least-squares regression: y=86.9 - 3.2x; n=14; r^2=0.525; p=0.003.
b) For *Murex occa*; r_s=1, P<<0.05, df=2; y=91.8 - 3.75x; n=4, r^2=0.937; p=0.032.

Attributed causes

Decreases in the quality (i.e., species diversity and density/biomass) of soft-bottom habitats can be attributed to two main causes: 1) physical disruption by trawling, and there is some indication that the intensity of trawling has increased since the 1960s (*Profile* chap. 3); and 2) contamination of sediments from pollutants from various sources (see Prospective Analysis).

Consequences

A decline in the quality of soft-bottom habitats has had economic consequences in terms of contamination of marine food products (*Profile* chap. 7) and may be a contributing factor in the observed decline in catch-per-unit-effort (CPUE) for demersal fisheries. An economically important consequence for sandy beach areas is the negative impact on tourism (e.g., due to increased amounts of tar). Observations from other regions, such as the North Sea and Baltic Sea (HELCOM, 1990; Clark, 1992) suggest that changes in the composition and density of non-commercial benthic communities are likely consequences of soft-bottom habitat pollution and physical disturbance, but quantitative data for the Malacca Straits were not included in the *Profile* nor in the *Appendix Documents*.

Biodiversity

Assessment endpoints here ought to be in terms of population density and species diversity. The density measures possibly ought to relate to the sizes thought by some to represent thresholds leading to inevitable extinction, although this concept is not without dispute (Caughley, 1994). Measurement of diversity is also not without controversy, and here it is probably best to consider species numbers only (Olsgard and Gray, 1995). Table 2 summarizes the results of the retrospective assessment for biodiversity of non-commercial and commercial species in the Straits.

Non-commercial species

Evidence for decline

For non-commercial species there are few quantitative data in the *Profile* or the *Appendix Documents*, so the evidence for decline is somewhat anecdotal: 'the increasing deterioration of environmental conditions in the Straits and increasing human activities have resulted in changes of species composition, the disappearance of other species, and the increasing numbers of endangered species'. Two indigenous fish species (*Alosa toli* and *Lactarius lactarius*) were abundant in the pre-1950s. The former is now cited as rare and the latter extinct. Sitings of sting rays have decreased and dugongs that were once common in the Straits are now scarce (*Profile* chap. 7). A list of threatened or protected species associated with the mangrove ecosystem is given in *Profile* Table 2-12. Endangered species associated with the seagrass ecosystem include the sea cow (dugong) already mentioned, the green turtle and the rabbit fish (*Profile* chap. 2). The RED LIST of Singapore cites 52 species of fish, 13 species of coral and anemones and 12 species of crustaceans declared extinct, and more than 50 other species considered threatened (*Profile* chap. 7).

Attributed causes

The major causes of reductions in biodiversity include loss of major habitats and direct ecotoxicological effects of contaminants of various kinds.

Consequences

Possible implications of reduced biodiversity, apart from aesthetic and tourist attraction, include loss of contribution to the stability and functioning of the ecosystems of which the lost or now rarer species were (are) a part. However, these contributions are not straightforward (Lawton, 1994).

Commercial species

Evidence for decline

For pelagic fish in the Indonesian sector, a reducing CPUE has been observed (*Profile* Table 3-15). Also, there is evidence that fishermen are moving their activities from the Straits to other waters (*Profile* chap. 3). For the Malaysian side, there has been a fall in total catch and catch rates, a fall in CPUE, and a fall in the ratio of commercial to trash fish. On the other hand, catch rates (kg/h) of prawns have apparently increased in Malaysia, prompting

Table 2. Summary of Retrospective Analysis of Decreases in Biodiversity in the Straits.

Biodiversity of:	Evidence for Decline	Ecological Consequences	Economic Consequences
Non-commercial Species	Little	Unknown	Unknown
Commercial Species	Much	Unknown	Considerable

The amount of evidence for decline given in the *Profile* is rated as 'little' or 'much'. The seriousness of ecological and economic consequences resulting from the decline is rated as 'unknown', 'minimal' or 'considerable'.

the comment that, 'the allegations that the prawn resources of the west coast of the Peninsula have declined or has been depleted are unrealistic. It is more acceptable to consider the up and down trend of the stock as the result of natural fluctuation' (*Appendix Document* I-M).

Attributed causes

The major cause of the decline in commercially exploited species is overfishing, with catches exceeding maximum sustainable yields for shrimp, demersal but not pelagic fish stocks in the Indonesian portion (*Profile* chap. 3), and for all categories of commercial fish from the west coast of Peninsular Malaysia (*Profile* Table 3-16). This, at least for Malaysia, is associated with more effective fishing methods and a marked increase in the number of fishermen since the 1960s. A number of species are taken from soft-bottom habitats including seaweeds, horseshoe crabs, shrimps, crabs, bivalves, gastropods, sea cucumbers and sea urchins. There is a multimillion dollar cockle industry along the west coast of Malaysia (*Profile* chap. 2). White prawn, tiger prawn and greasyback prawns are fished. Prawn fisheries exceed potential yield (*Profile* chap. 3). Approximately 14,000 tonnes of sergestid shrimp (*Acete*) are removed annually (*Profile* chap. 3). Approximately 8,000 tonnes of mangrove crab, *Scylla serrata*, are landed from the mangrove areas in Malaysia. 'The crabs are probably overfished; the size of the crabs landed are usually small'.

Also implicated in the reductions in biodiversity are the losses of nursery grounds (discussed above) and both chronic and acute pollution (for examples of the latter, see *Profile* Table 7-23).

Consequences

Possible consequences of reduced biodiversity of commercially-exploited species involve impacts on the economy and contributions of lost or reduced fish species to the ecology of the Straits ecosystems (see above).

Human Health

The targets here are obvious. The assessment endpoints should be in terms of increased morbidity and increased death rates. These might be associated with particular acute conditions, for example arising out of specific accidents or with chronic exposure to long-term and possibly lower levels of contamination.

Evidence for decline

Appendix Document I-I states that in Indonesia, diarrhea was the second leading cause of death among children under 5 years of age. However, the number of cases of diarrhea has declined by 28% between 1994 (5,230,366 cases) and 1996 (3,757,132 cases). The causes of gastrointestinal and other communicable diseases have been attributed to a lack of clean water, decent housing, garbage and waste disposal, and food hygiene. So causes are not easily related to conditions in the Straits per se (i.e., arising from seafood consumption or dermal exposure to pollutants). Malaysia reported a decline in the incidence of acute gastroenteritis between 1991 and 1995, from 1,790 per 100,000 persons to 1,263 per 100,000 persons.

Implications for Choice of Endpoints in Prospective Analysis

Currently there are few quantitative measures in terms of endpoints that have direct relevance to the condition of the key targets, whether habitats, species or humans. Hence, it will be necessary in the subsequent analysis to work with measurement endpoints that bear generally on the condition of the targets. This is a very usual situation with environmental risk assessments and often entails comparisons of generalized measures of exposure with generalized measures of effect. These risk quotient techniques do not lead to precise statements about the likelihood of effects (Calow, 1995), but they do provide indices of risk that can act as a useful starting point for more detailed analyses, and they are used widely in a regulatory context (Smith and Hart, 1994).

PROSPECTIVE ANALYSIS

Introduction

A prospective risk assessment should estimate the likelihood of adverse effects to appropriate targets from environmental conditions that exist, or might exist, within the Straits. This, therefore, involves comparing measured or predicted environmental concentrations (respectively, MECs and PECs) and for humans measured and predicted exposure levels (respectively, MELs and PELs) with either adverse effects in targets, or with critical, threshold no-effect levels of substances. Conventionally, these are referred to as no observed effect concentrations (NOECs) for ecological systems and no observed adverse effect levels (NOAELs) for humans. Techniques are discussed in more detail in MPP-EAS (1999a). Here, how the techniques have been applied to the risk assessment of contamination within the Straits is indicated.

PECs often involve combining a level of release from a source with presumptions about subsequent distribution, dilution and breakdown of the substance under consideration.

Hence:

PEC = f [distribution, dilution, breakdown]

where f means 'function of'. There are many more or less sophisticated models that incorporate these basic features (Mackay, 1994). For the Straits, quoted outflows of rivers are often taken as sources and PECs are estimated on the following basis. To predict environmental concentrations in the Straits ($PEC_{Straits}$) from information on land-derived contaminant loadings, an extremely simple, one-compartment model of the system has been used, which takes into account total dilution within the Straits, presumes thorough mixing and hence ignores the complexities of distribution, and (conservatively) ignores breakdown of contaminants. Total volume of the Straits has been calculated by assuming a symmetrical geometrical configuration with triangular cross-section, having an average width of 60,000 m (33 nautical miles), a depth of 30 m and a length of 1×10^6 m. This gives an estimated volume of approximately 10^{12} m^3. From the current speed of 1 knot (=1,853 m/hour) specified in *Profile* chap. 2, a flushing rate of once per 500 hours was calculated, or approximately once per month, but conservatively, it was rounded down to 10 times per year. In consequence, conservative estimates have been taken with respect to volume and flushing so that PECs will be maximized. Flushing could be as high as 20 times per year, but using 10 is precautionary. It is further presumed that there is thorough mixing, no backgrounds from other than the river inputs reported in *Profile* tables and no removal by either biological, chemical or physical means.

An alternative approach to predicting environmental concentrations at points around discharges into the Straits (PEC_{Local}) is to presume that concentrations in the discharges apply without dilution at the point of release and therefore have local effects in the Straits at that level. In other words, environmental concentrations in a river=local environmental concentrations in the Straits. Again, this presumes no dilution, no mixing, no loss and no background from the waters of the Straits.

The sources might be various kinds of facility on- or offshore, such as refineries and tankers, and these might be considered in groups (e.g., all refineries in

Singapore) or as particular industrial plants and ships. Releases from facilities such as these might arise during normal operations or by accident. These can be summarized as follows:

$$PEC_{operational} = f\,(\text{amount produced or carried})\,(\text{release})\,[\text{distribution, dilution, breakdown}]$$

For estimation of releases, either information given in the *Profile* or the *Appendix Documents* (e.g., as operational activities involving oil releases from shipping) or from standard scenarios of losses through both controlled and fugitive sources in industrial processes (European Commission, 1996) shall be used. Note that the terms in [ ] are as before.

$$PEC_{accidents} = f\,(\text{likelihood of accident})\,(\text{amount of toxic substance})\,[\text{distribution, dilution, breakdown}]$$

where the likelihood of accidents depends upon such factors as the likelihood of mechanical failure, management failures, adverse conditions, etc. and is often treated in itself as the output of the risk assessment (i.e., assessment of probability of accident). This approach is used in considering the likelihood of accidents to shipping within the Straits. Note that the terms in [ ] are the same as before.

For humans, the main concern has been exposure through food ingestion. Here, the PEL depends upon concentrations in food tissue. These are either derived from direct analysis or, indirectly, from exposure concentrations to which food organisms (e.g., fishes) are exposed:

$$PEL=f(PEC)(BCF)$$

where BCF=bioconcentration factor of the food organism. As with PECs, PELs can refer to broad groups of people, or populations in particular places or in a particular subpopulation (e.g., identified by age, sex, etc.). It should also be noted that if the PEL is defined (e.g., from acceptable intakes - see below) the critical PEC can be defined, and this approach is sometimes used.

As already noted (see Retrospective Assessment), it is rarely possible to be precise about targets of effects, or about those features of targets that should be measured in the risk assessment. One therefore relies on general assessments of likelihood of effects from concentrations of likely effectors. These are either derived from standards (STDs) often taken from the *Profile/Appendix Documents* or predicted no-effect thresholds, predicted no-effect concentrations (PNECs), predicted no-effect levels (PNELs) and predicted no (adverse) effect levels (PN(A)ELs). Both STDs and PNECs are calculated in similar ways. Their basis is toxicological and ecotoxicological effects information, often from standard tests. Lowest no-effect or effect concentrations are reduced by appropriate assessment, or uncertainty, factors to an extent that in part depends upon judgments about the quality of the data to give STDs and PNECs. These factors are supposed to take into account uncertainties about extrapolation from a limited number of species in laboratory conditions to many species in more complex field conditions (ECETOC, 1993). For humans, observed or predicted no-effect levels ((P)N(O)ELs) are divided by uncertainty/safety factors to give threshold values sometimes referred to as levels of concern (LOCs). The basis of assessment factors used here is discussed in ECETOC (1995) and MPP-EAS (1999a).

For the simplified ecological risk assessment, MECs and/or PECs are compared with PNECs and/or STDs. Ratios known as risk quotients (RQs) are used, where

$$RQ = \frac{(\text{MEC or PEC})}{(\text{PNEC or STD})} \tag{1}$$

For human health risk assessment:

$$RQ = \frac{(\text{MEL or PEL})}{(\text{PNEL or LOC})} \tag{2}$$

This does not give a precise probability of adverse effect. However, when RQ is greater than or equal to one (environmental concentration greater than effects level), it is presumed that there is a likelihood of effect that increases with the size of the ratio. On the other hand, when RQ is less than one (environmental concentration less than effects level), the likelihood of effect is low and not of concern. There are other more complex and apparently more sophisticated ways of carrying out risk assessments, but the data provided are generally not detailed nor robust enough to allow these approaches.

There are uncertainties in both the denominators and the numerators of the RQs; the PNECS, STDs, PNELs and TDIs depend on the reliability of the ecotoxicological and toxicological data upon which they are based and their relevance to the circumstances under consideration; the MECs and MELs are dependent on the reliability of sampling and analytical techniques; and the PECs and PELs are dependent on the assumptions incorporated into the models used in making the predictions and the reliability of input data. Sometimes, it is convenient to distinguish between uncertainty due to lack of understanding (e.g., in the derivation of PNECs and PNELs or in constructing the models used to generate PECs and PELs) and those due to stochastic effects such as variability among sampling sites used in MECs and MELs. As already noted, the most one can usually do is describe these uncertainties and make guesses as to their likely effects on predicted risks. This is especially the case with PNECs and STDs. Sensitivity analyses can be used to investigate the extent to which assumptions incorporated into the predictive models affect outputs. On the other hand, it is sometimes possible to obtain an impression of the effects of the stochastic variability by examining variances in the RQs and their components. When the component elements are distinguishable, there are standard methods for considering overall effects. Monte Carlo estimations, or related resampling techniques, are often employed to estimate the variance of derived variables, such as ratios (van Leeuwen and Hermens, 1995). This approach has been taken for several contaminants in refining the risk assessments (e.g., heavy metals and human health).

Otherwise, variability is examined in RQs and used to make judgments about the likelihood of particular observed values being greater than the critical threshold of one given the stochastic uncertainty in the observations. For this purpose, all one needs to know is if a given value of RQ signals a problem (greater than 1) or a situation of no concern. Since many of the input data (particularly MECs) tend to be skewed to the right, and hence approach a log-normal distribution, it is more accurate to estimate means and variances (of MECs and RQs) following logarithmic transformation of the raw data. In such cases, the critical value of RQ will be zero.

For certain substances that occur naturally, i.e., metals, there may be background concentrations. When these were available, it was presumed that they were from unpolluted areas and a measure of contamination (defined by GESAMP as 'raised levels of the chemical compared with natural background levels'; Olsgard and Gray, 1995) was calculated as the $MEC_{Straits}$ divided by the background concentration. Although a high level of contamination does not necessarily equate with a high level of biological effects, any substance present in the environment as a result of anthropogenic activity at concentrations greatly in excess of natural levels deserves careful consideration.

Heavy Metals

Concentrations (MECs) of a variety of heavy metals in water, sediments and biota from different stations in the Straits were presented in the *Profile*. For this risk assessment, it was assumed that these levels are representative of the Straits in general. Hence, they are referred to as $MEC_{Straits}$ and the risk quotients derived from them as $RQ_{Straits}$. These were compared with several different metal standards, namely, the Malaysian interim standard for marine quality (*Profile* Table 7-3), the Indonesian required standard for fisheries uses of marine waters (*Profile* Table 7-4) and the Danish standards for environmental water quality (MST, 1996). The latter are maximum limits permitted in Danish waters and are equal to or lower than levels permitted by the European Union. Also included (after the initial risk assessment) are further standards for Malaysia, Indonesia and the People's Republic of China. Some of these are reported as effluent and wastewater standards and presumably refer to conditions at end of pipe. The standards are summarized in Table 3.

Concentrations in water

Risk quotients for heavy metals measured in the waters off the west coast of Peninsular Malaysia are shown in Table 4. In addition to comparing the $MEC_{Straits}$ to various standards (from Table 3), a measure of contamination was also calculated by relating $MEC_{Straits}$ to published background metal levels from (presumably) unpolluted waters. This is important for metals for which there will be natural background concentrations. To demonstrate contamination, it is therefore necessary to assess the extent to which concentrations exceed background levels as indicated by the BQ ratios given in Table 4.

Table 3. Environmental Standards for Heavy Metals from Various Sources.

Metal	Table 7-3 Std	Table 7-4 Std	Danish (EU) Std	Indonesian Liquid Waste	Malaysian Effluent Discharges A (B)	Chinese Stds for grades I & (II/III)
As	100			50	50(100)	50, 100
Cd	10	0.2 (10)	2.5 (SW)	10	10(20)	5, 10
Cr	500		1	50	50(50)	100, 500
Cu	100	1.0 (60)	2.9	1,000	200 (1,000)	10, 100
Fe				1,000	1,000 (5,000)	
Hg	1	0.1 (0.3)	0.3 (SW)	1	5(50)	0.5, 1
Mn				500	200 (1,000)	
Ni			8.3	100	200 (1,000)	
Pb	100	0.2 (10)	5.6	30	100(500)	5, 100
Sn						
Zn			86	2,000	1,000 (1,000)	100, 1000

Values are given in µg/L. Standards from *Profile* Table 7-3 represent Malaysian Interim Standards for Marine Quality. Numbers for *Profile* Table 7-4 indicate 'required' concentrations, followed by 'allowable' concentrations in parentheses. Danish standards for Hg and Cd are specifically for seawater (SW); other values do not distinguish between freshwater and seawater. The Indonesian liquid waste standards for biotic use are from *Appendix Document* I-I. Malaysian effluent discharges are from *Appendix Document* I-M and are for catchment areas (A) and for outside catchment areas (B). Chinese standards are from *Appendix Document* XI.

Table 4. Metal Concentrations in Water for 1985-1991.

Metal	Highest mean MEC (µg/L)	$RQ_{Table\ 7\text{-}4}$	$RQ_{DK\ Std}$	$RQ_{Table\ 7\text{-}3}$	FES	BG (µg/L)	BQ
As	8 (9)	?	?	0.08	Y	1-1.5	8
Cd	114 (9)	570	46	11	YYY	0.004-0.011	28,500
Cr	62 (110)	?	?	0.12	Y	0.15-0.5	413
Cu	34 (310)	34	11.7	0.34	YY	0.06-0.2	567
Hg	68 (2)	680	227	68	YY	0.0005-0.0025	136,000
Pb	108 (9.8)	540	19	1	YYY	0.001-0.05	108,000

MECs and risk quotients ($RQ_{Straits}$s) are for the west coast of Peninsular Malaysia. The highest mean MECs are from *Profile* Table 7-3. MECs in parentheses are from *Appendix Document* I-M and are for a range of Malaysian stations measured in 1995 – they have not been used in the RQ analysis. RQs are the highest mean MECs divided by the appropriate standard (Table 7-3, Table 7-4, or Danish Std.) as indicated. FES=the relative frequency of samples exceeding the standard from Table 7-3; it is proportional to the number of Y's. BG=background values obtained from Laane (1992). BQ=highest mean value/background value.

Results of the initial risk quotient analysis (Table 4) can be summarized as follows:

1) On the basis of precautionary assumptions (i.e., using worst-case scenarios with the highest mean MECs), risk quotients for Pb, Hg, Cd and Cu generally exceed one.

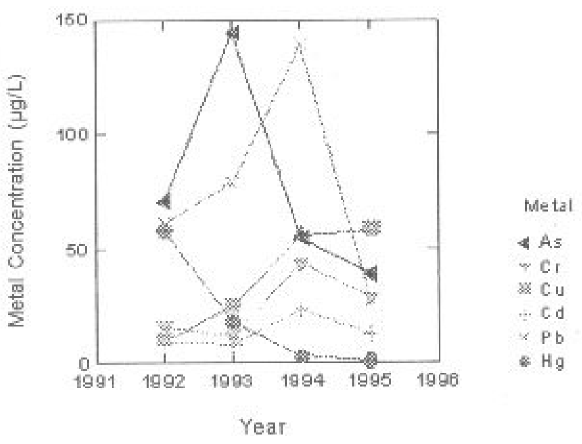
2) On the basis of risk quotients, the order of degree of risk is Hg > Cd > Pb > Cu. This is true regardless of which standards are used.

3) In terms of *Profile* Table 7-3 standards, Cd and Pb exceed environmental standards most frequently.

4) The degree of contamination in excess of natural background levels decreases in the order Hg > Pb > Cd > Cu > Cr > As, which is fairly consistent with the relative risks indicated by comparing RQs.

More recent MECs from Singapore (*Appendix Document* I-S, for 1996), Indonesia (*Appendix Document* I-I, for 1991/92) and Malaysia (*Appendix Document* I-M, for 1992-1995) give no higher concentrations than those in Table 4. The Malaysian time series suggest either no change or downward trends (i.e., Hg) but no upward trends (Figure 3).

Figure 3. Time Trends in Metal Concentrations in Water for Stations in Malaysia.



For Indonesia, river/estuarine concentrations for the Siak River (*Appendix Document* I-I) are given as: Pb, not detectable to 2 µg/L; Cd, not detectable to 4 µg/L; Cr, not detectable to 3 µg/L; Ni, not detectable to 1 µg/L.

The relative risks of heavy metal pollution were compared among different sites in the Straits from MECs provided in *Profile* Table 7-7, and these are shown in Table 5. The four metals for which STDs are available (i.e., Ni, Cu, Zn, and Pb) significantly exceed the critical log RQ value of zero (i.e., RQ=1; for an explanation of the use of log RQ, see below), as indicated by the fact that their lower 95% confidence limits exceeded zero. From these RQ_{Locals} copper consistently is associated with the highest environmental risks with RQs always exceeding 100 and in the Port of Singapore greater than 1,000 (cf. $RQ_{Straits}$). This is probably due to antifouling contaminants. Both nickel and lead also have RQs exceeding 10 at all sampling sites, and Zn has RQs close to or exceeding 10 at all sites. No single site ranked consistently highest or lowest for all of the measured metals, although Sentosa and Marina Bay appeared overall to be the least polluted of all of the sites.

Uncertainty analysis

There are two levels of uncertainty in these data: (a) based on standards and (b) based on variability in MECs. The variability in standards (Table 3) partly reflects purpose for use. None of the standards added after the initial risk assessment exceed the standards used in the initial risk assessment and the latter are presumed to be conservative

Variability across samples in Table 5 gives some impression of the variability in MECs that might exist generally in these kinds of data. An important question concerns the likelihood that observed RQs do not differ appreciably from the critical value of one given this variability. Thus, it is important to ask whether values appear to be above or below the critical value. This kind of question can be approached in a number of ways (e.g., Slob, 1994; van Leeuwen and Hermens, 1995), but here a look at the distributions of RQs relative to the critical value is warranted. Presuming a log-normal distribution of measured concentrations and hence RQs (which appears plausible on both theoretical grounds (Slob, 1994) and from inspection of the raw data), the

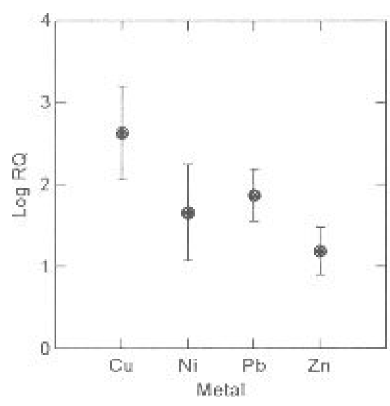
Table 5. RQ_{Local}s for Metal Concentrations in Water from Different Sites along the East Coast of the Straits.

Site	Ni	Cu	Zn	Pb
Port of Singapore	108	1345	28	No MEC
Sentosa & Marina Bay	24	138	9	54
East Coast	36	138	16	143
Johor Straits East	169	793	24	71
Johor Straits West	12	690	9	54
Mean log RQ	1.66	2.63	1.19	1.87
95% CL	1.07 - 2.24	2.06 - 3.20	0.90 - 1.48	1.55-2.19

Local RQs are based upon MECs given in *Profile* Table 7-7 and using Danish standards given in Table 3. Although MECs were provided in the *Profile* for Mn and Fe, there are no standards with which to compare them (see Table 3, above) so they have been omitted from this analysis. Mean log RQs and their 95% confidence limits are shown at the bottom of the table.

data were transformed and presented as mean log RQs ± 95% confidence limits in Figure 4. On the logarithmic scale in these plots a value of zero is equivalent to the critical value of RQ=1. Although the distributions of RQs were in general closer to normal following logarithmic transformation (untransformed distributions not shown), Cu and Pb remain somewhat skewed following transformation. Standard measures of variability (e.g., SEM, 95% CL, etc.) presume a normal distribution and will misrepresent the true variability to the extent that the distribution deviates from normality. Although many significance tests are robust to departures from normality, more sophisticated variance estimation techniques may be required for data that deviate widely from normality and that cannot be substantially improved by an appropriate transformation. For these data, despite moderate deviations from normality, there is little question that the log RQs for all metals are greater than zero.

Figure 4. Average Risk Quotients (transformed to logarithms) ± 95% Confidence Limits for Metals in Water.



All metals exceed the critical value of log RQ=0.

PECs

Using data from *Profile* Table 5-8 for river inputs of heavy metals on the west coast of Peninsular Malaysia, and using a one-compartment model, a $PEC_{Straits}$ (in µg/L) of 7×10^{-4} for Hg (all coming from Klang), 404×10^{-4} for Pb (most coming from Melaka), 2.6 for Cu (all from Klang) and 334×10^{-4} for Zn (all from Klang) are calculated. These give RQs based upon Danish standards as follows: 0.002 for Hg, 0.9 for Cu, 0.007 for Pb and 0.004 for Zn. Hence, all RQs, with the possible exception of Cu fall well below the critical value of 1. However, these figures may be of limited value. Of more significance will be the PEC_{Locals} for individual rivers and in particular for Klang and Melaka. Using average outflow data from *Profile* Table 2-3, a total annual outflow of approximately 10^{12} L/yr is calculated. Using this figure and applying metal loadings from *Profile* Table 5-11 gives the following PEC_{Locals} for the outflow from the Klang River: 25.8 µg/L for Cu and 0.089 µg/L for Pb. Again, using Danish standards (Table 3) gives the following RQ_{Locals} of 9.9 for Cu and 0.016 for Pb. This analysis therefore shows that there is a likely problem in terms of copper, but not lead at the outflow of the Klang River. *Profile* Table 5-9 shows that the Klang River has the greatest density of manufacturing industry of all the coastal river basins along the west coast of Peninsular Malaysia with a total of 612 industrial units, of which 214 are metal workings and 121 produce chemicals. It is also noted that a 1996 survey (*Profile* chap. 7) 'showed that, in general, heavy metal contamination in coastal waters was limited to certain areas close to industrial sites and estuaries'. Flow rates were not available for the Melaka River and therefore similar calculations for that system have not been carried out.

Concentrations in sediments

Measured concentrations of heavy metals in sediments were presented in the *Profile* for a number of stations (*Profile* Tables 7-5, 7-6, and 7-8). They are also reported in *Appendix Documents* I-S, I-I, I-M and XII. To date, there are no generally accepted sediment quality standards, and instead RQ estimates were based on water quality standards following van der Kooij et al. (1991). Briefly, threshold water concentrations were converted to critical sediment concentrations using the formula:

$$C_{sed} = \frac{(C_w \times K_{sw})}{r} \qquad (3)$$

where

C_w = the threshold concentration of metal in water (mg/L; here the water STD)
C_{sed} = the critical concentration of metal in sediment (mg/kg)
K_{sw} = the solids-water partition coefficient (L/kg)
r = an empirically derived concentration ratio between suspended matter:sediment (taken as 1.5 for metals and 2 for organics, van der Kooij et al., 1991)

The presumptions are therefore that the system is at steady state and that the chemical partitions accordingly between water and sediment phases, and furthermore that it is the toxicant concentration in porewater that is the sole source of exposure (cf. Forbes et al., 1996). The values of K_{sw} for metals were derived from Table 1 in van der Kooij et al. (1991) and are based on the Dutch Water Quality Database. As these authors noted, 'K_{sw} values show a great variability and depend on many physicochemical factors, e.g., salinity, pH, dissolved oxygen concentration, etc. In other countries, the K_{sw} values may differ substantially from the ones presented in Table 1'. Table 6 shows the median and lowest values of K_{sw} reported in Table 1 of van der Kooij et al. (1991) which were used to estimate C_{sed} values from the standards given in Table 3 (p. 26).

Table 7 calculates initial RQs from the highest MECs in *Profile* Tables 7-5, 7-6 and 7-8 using the various standards calculated on the basis of both median and lowest K_{sw}s (Table 6). Most of the highest MECs from the new data (*Appendix Documents*) are similar to those from the *Profile*. *Appendix Document* XII reports Pb concentrations around Singapore ranging from 16 to 250 µg/g and along the west coast of Malaysia from 1 to 180 µg/g, but no further details regarding location or number of samples in the high range were given. The very high value for Cu represents one site reported by Singapore (*Appendix Document* I-S). Ignoring this, the rank order for *Profile* and *Appendix* data sets are similar. Therefore, the risk assessment based on the *Profile* data was retained. It will be clear that there is a considerable amount of variability in these data and below. For the purposes of the initial risk assessment, however, the focus was on the RQs based on the Danish standards and lowest K_{sw}s since these contain standards for most metals and are moderately conservative (see bottom right column in Table 7). From this, the rank order of metals in terms of RQs (from highest to lowest) is Cu > Ni > Cr > Zn > Pb > Cd with Cu and Ni having values greater than one. From the tables, it will also be clear that the ranking of metals with regard to RQ was not dependent on whether lowest or median K_{sw}s were used. Copper had the highest RQs in all scenarios and

Table 6. Critical Sediment Concentrations (mg/kg) based on Water Quality Criteria for Selected Heavy Metals.

I. Using Median K_{sw}	Median K_{sw}	C_{sed} (Table 7-3)	C_{sed} (Table 7-4)	C_{sed} (DK STD)
As	10	700		
Cd	130	870	17.4 (870)	217.5
Cr	290	96,500		193
Cu	50	3,300	33 (198)	95.7
Hg	170	113	11.3 (33.9)	33.9
Ni	8			41.5
Pb	640	42,700	85.4 (4,270)	2,391
Zn	110			6,278
II. Using Lowest K_{sw}	**Lowest K_{sw}**	**C_{sed} (Table 7-3)**	**C_{sed} (Table 7-4)**	**C_{sed} (DK STD)**
As	5	400		
Cd	50	330	6.6 (330)	82.5
Cr	126	42,000		84
Cu	12	800	8.0 (48)	23.2
Hg	31	21	2.1 (6.3)	6.3
Ni	4			24.9
Pb	438	29,200	58.4 (2,920)	1,635
Zn	52			3,010

Values for K_{sw} (L/kg) were taken from Table 1 in van der Kooij et al. (1991) and water quality standards are shown in Table 3 (p. 26).

Table 7. Metals in Sediments (μg/g).

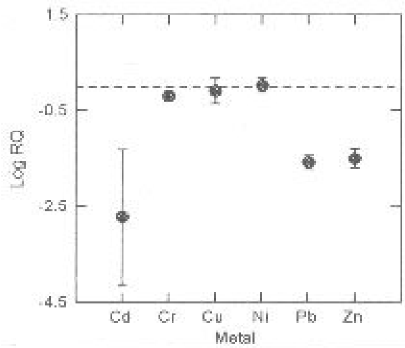
I. Using Median K_{sw}	Highest MEC (*Profile, Appendix*)	RQ (Table 7-3)	RQ (Table 7-4)	RQ (Danish Standard)
As	26, 13	0.0400		
Cd	5.5, 5	0.0060	0.3	0.03
Cr	69, 80	0.0007		0.36
Cu	229, 50 (1781)	0.0700	6.9	2.40
Hg	No MEC			
Ni	89, 40			2.10
Pb	134, 96 (250)	0.0030	1.6	0.06
Zn	428, 280			0.07
II. Using Lowest Median K_{sw}	**Highest MEC**	**RQ (Table 7-3)**	**RQ (Table 7-4)**	**RQ (Danish Standard)**
As	26	0.060		
Cd	5.5	0.020	0.8	0.07
Cr	69	0.002		0.80
Cu	229	0.300	28.6	9.90
Hg	No MEC			
Ni	89			3.60
Pb	134	0.005	2.3	0.08
Zn	428			0.10

The highest MEC values from *Profile* Tables 7-5, 7-6 and 7-8 are shown for each metal. In addition, highest MECs found in the *Appendix Documents* are also shown. RQs are calculated as the highest MEC (using the *Profile* values) divided by the sediment quality standards shown in Table 6. RQs were calculated using both median and lowest K_{sw}s (from Table 1 in van der Kooij et al. (1991).

these were always greater than one except when *Profile* Table 7-3 standards were used. Lead and cadmium had the lowest RQs (using Danish standards), and this contrasts with the water column situation where Cd and Pb had substantially higher RQs than Cu. Explanations for the lack of concordance between water column and sediment data include (but are not limited to): (1) that water and sediment samples were taken from different sites (cf. *Profile* Tables 7-7 and 7-8) and different metals were included in the two types of analysis (e.g., *Profile* Table 7-7 omits Cr, As and Sn in water, whereas Table 7-8 omits Fe in sediment); (2) that differences are associated with differences among metals in their partitioning between dissolved and particle-bound forms; (3) that sediment data are possibly more variable among sites than are water column data due to more restricted mixing, effects of organic matter content, particle size, etc.; 4) that sediments from some of the stations may be periodically dredged.

Sediment RQ_{Locals}, calculated from all the MECs in *Profile* Tables 7-5, 7-6 and 7-8 are presented as means ± 95% confidence limits (log scale) in Figure 5. Measured environmental concentrations exceed standards for Cu and Ni at several sites giving RQ_{Locals} greater than one. For Cu, the highest MEC was in the Port of Singapore with an RQ of about 6, and an even higher RQ would be obtained for the high MEC given in *Appendix Document* I-S as 1,781 μg/g. The high values of Cu and Zn recorded by Singapore have been attributed to antifouling paints. For Ni, highest values were found in the Riau stations with RQs ranging from about 1 to 3.6.

Figure 5. Average Risk Quotients (transformed to logarithms) ± 95% Confidence Limits for Metals in Sediments.



Cu and Ni are the only metals having confidence limits that overlap the critical value of log RQ=0.

Table 8 compares lowest and highest MECs with background levels obtained from Laane (1992). The BQ is the ratio of highest MEC to background value and shows a rank order from greatest to lowest of Cd > Cu > Zn > Pb > Ni > Cr. This ranking differs from the RQs based on standards. Differences between standards and background levels mean that background is often lower than the concentration that would cause adverse effects, but sometimes may be higher than an effect concentration if, inadvertently, the background site was polluted (e.g., Cr). Interlaboratory comparisons of metal concentrations from North Sea sediments (ICES, 1995) have shown that such methodological factors as separation of sediment into size fractions and the type of acid used for extraction can have a considerable influence on measured metal concentrations.

Table 8. Metal Concentrations in Sediments.

Metal	Lowest MEC	Highest MEC	BG	BQ
As	13	26		
Cd	nd	5	0.2-0.4	27.5
Cr	nd	69	60-90	1.2
Cu	4	229	15-40	15.3
Mn	72	524		
Ni	11	89	30-75	3.0
Pb	14	134	20-40	6.7
Sn	13	54	0.04-0.2	
Zn	21	428	50-100	8.6

MECs are from *Profile* Tables 7-5, 7-6 and 7-8 and are in units of mg/kg; nd=not detected. BG=background sediment metal concentrations obtained from Laane (1992). BQ=highest MEC/background sediment value.

Uncertainty analysis

There is clearly a great deal of variability in RQ values depending on which water quality standards are used and on whether median or lowest K_{sw} values are used in the calculation. Moreover, another source of variability arises from the MECs themselves. In Table 9, the relative importance of variability is characterized in each of these three elements by summarizing the factor difference between minimum and maximum values of these for each metal.

From equation 3, RQ is written as

$$RQ = \frac{MEC}{C_{sed}} = \frac{MEC/(C_w \times K_{sw})}{r} \qquad (4)$$

so alterations in any of the three elements defined in Table 9 will have equal weighting in terms of changes in RQ. Clearly, all three elements can introduce appreciable variability into the calculation of RQs, but from Table 9 by far the most important source of variability is due to the choice of different water quality standards. This is further emphasized by Table 3, which extends the range of possible standards from the initial risk assessment. Hence, this requires further clarification in terms of which standards are most appropriate for the Straits.

If one could more precisely describe the distributions of the variability for each of the elements of the RQ calculation, one could then use more rigorous uncertainty analyses to consider the extent to which this will lead to variability in the RQs. For example, Monte Carlo simulations could be used to randomly sample from the distributions to calculate a range of RQs, and from this, one would be able to predict the likelihood that any specified threshold value is exceeded. Other techniques that associate variances from different elements of the RQ calculation include Slob (1994). However, from the *Profile* only limited information on the extent and form of these variability distributions was available and so only the first stage uncertainty analysis that is summarized in Table 9 can be conducted.

Table 9. Uncertainty Analysis for Metals in Sediments.

Metal	Water Quality Stds.	K_{sw}s	MECs
As		3.4	2.0
Cd	50	9.9	5.5
Cr	500	6.2	69.0
Cu	100	12.4	54.5
Mn			7.3
Ni		5.0	8.1
Pb	500	7.9	9.6
Sn			4.2
Zn		4.1	20.4

Maximum factor difference (maximum/minimum) for water quality standards (Table 3), K_{sw}s (from van der Kooij et al., 1991) and MECs (from *Profile* Tables 7-5, 7-6 and 7-8).

Heavy metals and human health

It is possible to assess the risks to human health from the consumption of metal-contaminated seafood from information on per capita seafood consumption rates and the tolerable daily intake (TDI) rates for different metals. Tolerable intake rates for Cd, Pb, Hg, Cr, Ni and As, as defined by the US Food and Drug Administration (FDA), are shown in Table 10. Maximum tolerable intake rates for the essential metals, Cu, Fe, Mn and Zn were not found, but instead values for the recommended daily allowances (RDA) which are widely available on nutritional supplements were used. However, in doing this it should be noted that exceeding an RQ of one for an essential metal is less likely to cause a risk to human health than exceeding an RQ of one for a non-essential metal. In considering the risks associated with essential metals, it is useful to note that for Cr the maximum tolerable intake for adults has been set by the US FDA at 200 µg/day, whereas recommended daily allowances for this metal are 20 (1-10 yr) and 50 (adults) µg/day. Thus, for adults the difference between recommended intake and maximum tolerable intake is a factor of 4. In the absence of further information, this shall be used as a general rule of thumb for the remaining essential metals.

Daily intake levels of heavy metals can be calculated on the basis of estimated daily seafood consumption and metal concentration in seafood tissue, under the assumption that seafood (i.e., fish or shellfish) is the only source of metal intake, as follows:

$$\text{Daily Metal Intake } (\mu g/\text{person/day}) = \text{Daily Intake of Shellfish } (g/\text{person/day}) \times \text{Shellfish Metal Content } (\mu g/g \text{ shellfish tissue}) \quad (5)$$

Assessing human health risks associated with the consumption of contaminated seafood can be performed by comparing the metal intake rates (equation 5) to the tolerable daily intakes (Table 10):

$$\frac{\text{Daily Metal Intake } (\mu g/\text{person/day})}{\text{Tolerable Daily Intake } (\mu g/\text{person/day})} = RQ \quad (6)$$

Table 10. Metals in Seafood (i.e., fish and shellfish).

Metal	Source & Definition	Intake Level
As	US FDA, Tolerable daily intake	130 µg/person/day
Cd	US FDA, Maximum tolerable daily intake	55 µg/person/day
Cr	US FDA, Safe and adequate dietary intake	200 µg/person/day
Cu	Recommended daily allowance	5 mg/day: age 1-10 yr 15 mg/day: adults
Fe	Recommended daily allowance	8 mg/day: age 1-10 yr 14 mg/day: adults
Hg	US FDA, Tolerable daily intake estimated from FDA Action Level of 1 ppm.	16 µg/day
Mn	Recommended daily allowance	1 mg/day: age 1-10 yr 2.5 mg/day: adults
Ni	US FDA, Provisional maximum tolerable daily intake	1.2 mg/person/day
Pb	US FDA, Provisional tolerable total intake	6 µg/day: age 0-6 yr 15µg/day: age 7-adult 25 µg/day: pregnant women 75 µg/day: adults
Zn	Recommended daily allowance	1 mg/day: age 1-10 yr 2 mg/day: adults

Tolerable or acceptable levels of intake for non-essential heavy metals were obtained from the US FDA (http://vm.cfsan.fda.gov/); recommended daily allowances for essential metals are values provided for commercial nutritional supplements. The tolerable daily intake level for Hg was estimated from the US FDA action level of 1 ppm assuming an average seafood intake of 16 g/person/day (the USA average). Except where indicated the figures are assumed to have been estimated for a 60 kg adult.

Levels of concern, sometimes referred to as 'action levels', represent limits at or above which the US FDA will take legal action to remove products from the market. On the basis of tolerable daily intakes of heavy metals and estimated seafood consumption rates, levels of concern for metal contamination of seafood can be defined as follows:

$$\frac{\text{Tolerable Daily Intake } (\mu\text{g/person/day})}{\text{Seafood Consumption (g seafood/person/day)}} = \text{Level of Concern } (\mu\text{g/g seafood tissue}) \quad (7)$$

Thus, another approach for assessing risks associated with seafood consumption is to divide the measured metal concentration in seafood tissue by the level of concern for that metal. Alternatively, one can divide the tolerable daily intake of metal by the measured levels of metals in seafood to calculate the maximum level of seafood consumption that would not lead to metal intakes above the tolerable level.

Several estimates of seafood consumption rates in the littoral States have been reported, and these are shown in Table 11. It should be noted that the values in Table 11 are much higher than similar consumption rates reported for the USA. The latter were estimated by a Market Research Corporation of America 14-day survey (MRCA, 1988) and gave average and estimated 90th percentile daily intakes of molluscan bivalves by adults (18-44 yr) of 12 and 18 g/person/day, respectively. Average and 90th percentile intakes for crustaceans were 9 and 19 g/person/day, respectively.

Taking the tolerable daily intake rates for the metals shown in Table 10 and dividing these by the lowest and highest seafood consumption rates from Table 11 gives a set of levels of concern for the lowest (44 g seafood/day) and highest (116 g seafood/day) seafood consumption groups, and these are shown in Table 12.

Measured concentrations of heavy metals in fish and shellfish from the west coast of Peninsular Malaysia are given in *Profile* Table 7-9 and in *Appendix Documents* I-M and XII, and from Indonesia in *Appendix Document* VII. Worst case human health risks can be estimated by comparing the highest reported tissue metal concentrations with the levels of concern for the high seafood consumption group. These are shown in Tables 13 to 17.

From Table 13, it can be seen that the highest MELs reported in *Profile* Table 7-9 exceed the level of concern for the high seafood consumption group for Cd, Pb and Hg, but for the essential metals Cu and Zn only the recommended daily allowance for 1-10 year-olds for Cu is exceeded. From these data, it is possible to estimate the maximum level of seafood consumption that will not lead to a metal intake above the tolerable level. For Pb, the highest MEL is 1.63 µg/g. Dividing the TDIs for different age groups by the MEL, it is estimated that young children (0-6 yr) consuming seafood from areas with the highest MEL for Pb should consume no more than 26 g of seafood per week. Likewise, older children (7 yr-adult) should consume no more than 64 g seafood per week, pregnant women should consume no more than 107 g seafood per week and non-pregnant adults should consume no more than 322 g seafood per week.

Table 11. Seafood Consumption Rates in the Littoral States Reported from Several Sources.

Littoral State	Seafood Consumption (g/person/day)	*Appendix Document*
Indonesia		
Riau, 1993	56	I-I
Indonesia	44	*Profile*
Singapore	101	I-S
Malaysia	68.5	II
Malaysia, 1981-1983		
Segamat	116	I-M
Pendang	69	I-M
Tg. Malim/Slim River	98	I-M

For the data shown in Table 14, levels of the essential metals, Cu and Zn, were below the RDA. For cockles, maximum MELs for Cd and Pb exceeded the levels of concern for these metals. For bony fish, maximum MELs for Pb and Hg exceeded the levels of concern for these metals. There was no single station that had the highest concentration for all metals. For example, Kuala Sepetang had the highest MELs for Pb, whereas Sg. Besar had the highest values for Cd.

Table 12. Levels of Concern (µg metal/g seafood tissue) Estimated for the Lowest (44 g seafood/day) and Highest (116 g seafood/day) Seafood Consumption Groups in the Littoral States.

Metal	TDI (µg/person/day)	Level of Concern (low consumption group)	Level of Concern (high consumption group)
As	130	2.95	1.12
Cd	55	1.25	0.47
Cr	200	4.55	1.72
Cu	400 (1-10 yr)	9.09	3.45
	2,000 (adults)	45.45	17.24
Fe	8,000 (1-10 yr)	181.82	68.97
	14,000 (adults)	318.18	120.69
Hg	16	0.36	0.14
Mn	1,000 (1-10 yr)	22.73	8.62
	2,500 (adults)	56.82	21.55
Ni	1,200	27.27	10.34
Pb	6 (0-6 yr)	0.14	0.05
	15 (7-adult)	0.34	0.13
	25 (pregnant women)	0.57	0.22
	75 (adults)	1.70	0.65
Zn	5,000 (1-10 yr)	113.64	43.10
	15,000 (adults)	340.91	129.31

Seafood tissue that has a metal content exceeding the level of concern is considered to cause a risk to human health.

Table 13. Worst-case RQs for Tissue Metal Levels Reported in *Profile* Table 7-9 from the West Coast of Peninsular Malaysia.

Metal	Highest MEL	RQ (high consumption group)
Cd	1.11	2.36
Cu	7.70	2.23 (1-10 yr)
		0.45 (adults)
Hg	25.2	180
Pb	1.63	32.6 (0-6 yr)
		12.54 (7-adult)
		7.41 (pregnant women)
		2.51 (adults)
Zn	33.7	0.78 (1-10 yr)
		0.26 (adults)

Original references from which the data were obtained were published in 1983 and 1986. MELs are in units of µg/g tissue.

Taking Pb as an example, it is estimated that maximum seafood consumption rates for persons consuming bony fish from Terengganu should not exceed the following values: 65 g/week (0-6 yr); 162 g/week (7 yr-adult); 270 g/week (pregnant women); 116 g/day (non-pregnant adults).

With the exception of Cu and Zn for adults, the average MELs shown in Table 15 for all metals at all three sites exceed the levels of concern calculated for the high seafood consumption group. It should be emphasized that the MELs reported in *Appendix Document* I-M (and hence used here) were means (± SD) and not maximum values.

For Pb, which tended to have the highest RQs, it is estimated that *Anadara granosa* consumed from areas of the highest MELs (i.e., Kuala Juru culture bed) should not exceed the following levels: 9 g/week (0-6 yr); 23 g/week (7 yr-adult); 38 g/week (pregnant women); 113 g/week (non-pregnant adults).

With the exception of Zn for adults the maximum tissue levels in the tiger prawns are equal to or exceed the level of concern for all metals for the high seafood consumption group. For Pb, which tended to have the highest RQs, it is estimated that consumption of prawns with the highest MEL should be very limited and should not exceed the following levels: 5 g/month (0-6 yr); 13 g/month (7 yr-adult); 22 g/month (pregnant women); 66 g/month (non-pregnant adults) (see Table 16).

Table 14. Maximum Metal Tissue Levels Measured in Cockles (25 individuals per metal) and Bony Fish (4-10 species and 12-36 individuals per metal) from Stations on the West Coast of Peninsular Malaysia.

Specimen Type	Metal	Station with Highest MEL	Maximum MEL (µg/g)	RQ (high consumption group)
Cockles (Latin name not given)	Cd	Sg. Besar, Selangor	1.11	2.36
	Cu	Kuala Sepetang, Perak	3.3	0.96 (1-10 yr) 0.19 (adults)
	Hg	Sg. Besar, Selangor	0.128	0.91
	Pb	Kuala Sepetang, Perak	1.63	32.6 (0-6 yr) 12.54 (7-adult) 7.41 (pregnant women) 2.51 (adults)
	Zn	Juru, Mainland Penang	33.7	0.78 (1-10 yr) 0.26 (adults)
Bony fish (mixed species)	Cd	Selangor	0.308	0.66
	Cu	Terengganu	3.225	0.93 (1-10 yr) 0.19 (adults)
	Hg	Air Tawar	0.758	5.41 0.93 (1-10 yr)
	Pb	Terengganu	0.647	12.94 (0-6 yr) 4.98 (7-adult) 2.94 (pregnant women) 1.00 (adults)
	Zn	Selangor	20.48	0.48 (1-10 yr) 0.16 (adults)

Data were cited in *Appendix Document* I-M. Original source was published in 1986. Note that the data for Pb are from the same source as those cited in *Profile* Table 7-9.

Both Cd and Pb exceed the levels of concern for the high seafood consumption group. The mouth of Sungai Asahan appears to be particularly enriched in Pb, and for this reason bivalves should rarely, if ever, be consumed from this area. The maximum MEL for Hg is also close to or exceeding the level of concern, whereas the maximum MELs for the essential metals Cu and Zn are close enough to the RDA that these probably do not pose a significant risk (see Table 17).

Appendix Document XII reports Pb levels in bivalves from the Straits in the range 2-7 µg/g and in tiger prawns (*Penaeus monodon*) in the range 0.06-5.9 µg/g. Comparison with levels of concern in Table 12 indicates these tissue concentrations to be of concern.

Appendix Document I-M provides heavy metal sediment data for 1994 from culture beds of *Anadara granosa*. Relating these to the measured levels in bivalve tissue collected from the same area can be used to estimate a bioconcentration factor for each metal. The estimated bioconcentration factors can be used to predict the tissue levels of *Anadara granosa* at other sites for which only sediment (and no tissue) data were available. The results of these analyses are presented in Tables 18 and 19.

The PEC_{tissue} calculations shown in Table 19 suggest that persons consuming large amounts of seafood from either of these sites would be at risk in terms of both Cd and Pb. It is critical to note in this regard that the Lekir sediments are within the maximum permissible limits established by the Malaysian Government.

<u>Uncertainty analysis</u>

There are several major sources of uncertainty involved in assessing the risk to humans from consuming metal-contaminated seafood. The first is uncertainty in tolerable daily intakes, which may be adjusted as more

Table 15. **Metal Tissue Levels in *Anadara granosa* from Malaysian Sites Reported *in Appendix Document* I-M.**

Site	Metal	Mean MEL (μg/g)	RQ (high consumption group)
Kuala Juru, culture bed			
	As	128.33	114.58
	Cd	0.73	1.55
	Cu	3.89	1.13 (1-10 yr)
			0.23 (adults)
	Fe	985.90	14.29 (1-10 yr)
			8.17 (adults)
	Pb	4.66	93.2 (0-6 yr)
			35.85 (7-adult)
			21.18 (pregnant women)
			7.17 (adults)
	Zn	113.64	2.64 (1-10 yr)
			0.88 (adults)
Kuala Sepetang, culture bed			
	As	75.86	67.73
	Cd	0.72	1.53
	Cu	4.08	1.18 (1-10 yr)
			0.24 (adults)
	Fe	443.12	6.42 (1-10 yr)
			3.67 (adults)
	Pb	1.63	32.6 (0-6 yr)
			12.54 (7-adult)
			7.41 (pregnant women)
			2.51 (adults)
	Zn	72.24	1.68 (1-10 yr)
			0.56 (adults)
Kuala Juru, mudflat			
	As	150.81	134.65
	Cd	0.95	2.02
	Cu	5.45	1.58 (1-10 yr)
			0.32 (adults)
	Fe	1332.43	19.32 (1-10 yr)
			11.04 (adults)
	Pb	Not reported	
	Zn	149.49	3.47 (1-10 yr)
			1.16 (adults)

Note that in contrast to Tables 13 and 14, the reported MELs are means and not maximum values. The original source of the tissue data was published in 1995. Metal tissue levels are means. The MELs for the mudflat are for bivalves in the smallest size class reported (i.e., 0.12 g dry weight), which tended to have somewhat higher MELs relative to larger size classes. Note that the data for Pb are from the same source as those cited in *Profile* Table 7-9.

information becomes available regarding the toxic effects of metals in humans. Many of the TDIs in Table 10 are based on an 'average' adult; tolerances may vary widely as a function of age, weight, sex, etc. The analysis assumes that seafood consumption is the sole source of metal exposure; if other dietary sources as well as non-dietary exposures are significant, total daily intake of metals will be underestimated. Finally, the amount of seafood consumed is highly variable, but a critical factor in determining tolerable levels of metals in fish tissue.

Uncertainty in standards

The question of standards is of particular concern for Pb. Whereas the US FDA has defined relatively conservative TDIs for Pb and distinguishes among different age groups, standards for other countries vary

Table 16. Measured Tissue Metal Levels in the Tiger Prawn, *Penaeus monodon* from Malaysia.		
Metal	Maximum MEL (µg/g)	RQ (high consumption group)
Cd	6.10	12.98
Cr	5.50	3.20
Cu	88.33	25.60 (1-10 yr) 5.12 (adults)
Fe	297.40	4.31 (1-10 yr) 2.46 (adults)
Mn	22.90	2.66 (1-10 yr) 1.06 (adults) 0.52 (adults)
Ni	11.90	1.15
Pb	32.00	640.00 (0-6 yr) 246.15 (7-adult) 145.45 (pregnant women) 49.23 (adults)
Zn	67.60	1.57 (1-10 yr)

The data are from *Appendix Document* XII and were from a study in which the prawns were reared in water with metal concentrations 'about equal to the concentration found in water at the Ko-Nelayan hatchery/Inanam River'. The original data are derived from several articles published in the early 1990s.

considerably. The US FDA figures were developed using information on the lowest levels of Pb exposure associated with adverse effects (i.e., neurobehavioral and cognitive development). They are calculated as the LOEC for each age group divided by 10. These levels are meant to be applied to short-term and chronic exposures and not meant to apply to acute exposure incidents. Pregnant women are considered to be at an increased risk due to increased susceptibility to Pb of the fetus, especially during the period of neuronal development.

Appendix Document XII lists permissible limits for Pb in seafood. They are in µg Pb/g tissue: Spain, 5.0; Malaysia, 2.0, Italy, 2.0; Chile, 2.0; Australia, 2.5. For comparison, it is relevant to note that the US FDA level of concern for Pb in seafood is 1.6. This is based on an estimated seafood consumption rate of 16 g/day. Many of the countries listed above are likely to have higher seafood consumption rates than the USA (and therefore should have lower tissue limits to account for this), and second, the consumption rates are likely to vary widely among countries suggesting that similar permissible limits for Pb in seafood tissue will lead to widely varying Pb intake rates. *Appendix Document* XII quotes a value of 50 µg Pb per kg body weight per week established as a provisional tolerable weekly intake by WHO. For a 60 kg adult, this would give a Pb intake of 429 µg/day –a value nearly six times higher than the US FDA TDI for adults - and for a 10 kg child (about 1 year old) a Pb intake of 71.4 µg/day–a value 12 times higher than the US FDA TDI for small children.

Although a seafood consumption rate of 116 g/day for the high consumption group is likely to be an overestimate as far as children are concerned, the analysis does not take into account non-food sources of uptake. Results of a survey of seafood consumption rates in the USA indicate that adults (18-44 years old) consume 2-3 times as much shellfish per day as 2-5 year olds. If the relative rates of consumption also hold for the littoral States, this would suggest that children from 2 to 5 years old in the low consumption group (44 g seafood/adult/day) would consume approximately 15-22 g seafood/day, and children 2-5 years old in the high consumption group (116 g seafood/adult/day) would consume approximately 39-58 g seafood/day. These values can be compared to the recommended maximum seafood consumption rates given in the previous section.

An apparently substantial source of Pb intake in children is associated with the ingestion of non-food materials (e.g., soil). Children under 3 years have been shown to ingest about 30-100 mg of soil and dust per day. In addition, part of the reason that children are more susceptible to Pb is that whereas adults absorb 5-15% of ingested Pb, children absorb around 50% of ingested Pb.

For the purposes of Malacca Straits risk assessment, the US FDA TDIs for Pb have been chosen as these are both recent and appear to be based on fairly thorough toxicological information. Given the serious toxicological consequences of Pb exposure, particularly for children, it is essential that the standards used by the littoral States be given more thorough evaluation.

Uncertainty in MELs

Maximum detected concentrations of Hg, Pb and Cd reported in *Profile* Table 7-9 all exceed the levels of concern. However, since *Profile* Table 7-9 only provides ranges, it was not possible in the initial risk assessment to determine how frequently fish tissue exceeds tolerable levels of heavy metals for average

Table 17. Tissue Metal Levels in Molluscs from Indonesia (North Sumatra) Measured in 1996.

Species and Site	Metal	MEL (µg/g)	RQ (high consumption group)
Anadara granosa, mouth of Sg. Asahan	Cd	3.74	7.96
	Cu	5.06	1.47 (1-10 yr)
			0.29 (adults)
	Hg	0.13	0.93
	Pb	39.39	787.80 (0-6 yr)
			303.00 (7-adult)
			179.05 (pregnant women)
			60.60 (adults)
	Zn	50.55	1.17 (1-10 yr)
			0.39 (adults)
Anadara granosa, Sg. Deli estuary	Cd	1.29	2.74
	Cu	1.42	0.41 (1-10 yr)
			0.08 (adults)
	Pb	1.79	35.8 (0-6 yr)
			13.77 (7-adult)
			8.14 (pregnant women)
			2.75 (adults)
Anadara indica, mouth of Sg. Asahan	Cd	5.74	12.21
	Cu	5.90	1.71 (1-10 yr)
			0.34 (adults)
	Hg	0.20	1.43
	Pb	41.26	825.20 (0-6 yr)
			317.38 (7- adult)
			187.55 (pregnant women)
			63.48 (adults)
	Zn	74.54	1.73 (1-10 yr)
			0.58 (adults)
Lingula unguis, Sg. Deli estuary	Cd	1.83	3.89
	Cu	1.96	0.57 (1-10 yr)
			0.11 (adults)
	Pb	2.42	48.4 (0-6 yr)
			18.62 (7-adult)
			11.00 (pregnant women)
			3.72 (adults)

Data are from *Appendix Document* VII. There was only one MEL (presumably a mean) reported for each metal.

Table 18. Measured Sediment and Bivalve Tissue Concentrations of Heavy Metals from Mudflats in Kuala Juru, Malaysia.

Metal	Sediment MEC (µg/g)	*Anadara granosa* tissue (µg/g)	BCF
Cd	2.55	0.73	0.29
Cu	40.8	3.89	0.10
Fe	11,485	985.9	0.09
Pb	43.8	4.66	0.11
Zn	266	113.64	0.43

The data are from 1994/95 and were reported in *Appendix Document* I-M. BCF values were calculated as $[C_{tissue}]/[C_{sed}]$. It should be noted that this equation differs from that used in van der Kooij et al. (1991; eqn. 7), but that the predicted tissue concentrations will be the same using either equation.

and high fish consumers. It is noted that the *Profile* states that 'shellfish were relatively safe for human consumption. However, there is a common perception among the more educated sector ... to avoid shellfish, particularly the blood cockles, as a precautionary measure against heavy metal poisoning'. Apparently this has an economic impact on the cockle industry in Malaysia (*Profile* chap. 7).

To take into account uncertainty in the estimates of risk, Monte Carlo simulations were performed in which variability in both MELs and seafood consumption rates were accounted for. The general approach was to use the relevant seafood consumption rates to estimate levels of concern,

Table 19. Predicted Tissue Levels of Heavy Metals for *Anadara granosa* based on Measured Sediment Concentrations.			
Metal	Sediment MEC	PEC_{tissue}	RQ (for high consumption group)
Predicted Tissue Levels for the Straits of Tuba:			
Cd	1.85	0.54	1.15
Cu	5.28	0.53	0.15 (1-10 yr)
Pb	26.94	2.96	59.20 (0-6 yr) 21.14 (adults)
Zn	35.31	15.18	0.35 (1-10 yr)
Predicted Tissue Levels for Lekir:			
Cd	6.1	1.77	3.77
Cu	6.3	0.63	0.18 (1-10 yr)
Pb	1.8	0.20	4.00 (1-10 yr) 1.43 (adults)
Zn	64	27.52	0.64 (1-10 yr)

Sediment MECs for the Straits of Tuba are *from Appendix Document* I-M and are from 1995; sediment MECs for Lekir are from *Appendix Document* I-M and are from 1994. The RQs are based on levels of concern estimated for the high seafood consumption group (116 g/day) shown in Table 12.

which were then fit to a uniform distribution (i.e., to consider the minimum, median and maximum consumption rates equally likely). However, it can be noted that preliminary simulations with different distributions showed that the choice of distribution for the consumption rates/levels of concern had little influence on the resulting probability estimates. Since most of the information on metal tissue levels was from the Malaysian side of the Straits, the range of seafood consumption rates reported from Malaysia (Table 11) was used. Specifically the consumption rates of 69, 98 and 116 g/day were considered to be equally likely and to give a measure of uncertainty in the denominator of the RQ. With regard to MELs (i.e., the numerator in the RQ), one of two approaches was taken depending on how the original data were reported. For cases in which MELs were given as mean ± SD, these values were used to generate a log-normal distribution of MELs. For cases in which ranges of MELs were reported, the data were fit to a triangular distribution with the minimum and maximum values providing the edges of the distribution and the mean (if reported) or geometric mean, calculated from the range (if the mean was not reported), representing the most likely value for the MEL. Random samples (with replacement) were taken from the distribution of MELs and levels of concern 1,000 times to generate a sample of RQs. From the resulting distribution of RQs, estimates (from the area under the curve) could be made of the probability that the RQ exceeded a given value (e.g., 1, 10 or 100). All Monte Carlo analyses were performed using Crystal Ball v. 4 (Decisioneering Inc.).

Taking the data on cockles and bony fish shown in Table 14, and using the Malaysian fish consumption rates of 69, 98 and 116 g/day, the probability that RQ exceeds one was calculated for each station separately. To run the Monte Carlo analysis, the MELs were fit to a triangular distribution (with the reported mean for the cockles or the calculated geometric mean for the bony fish, and the minimum and maximum reported values providing the edges of the distribution). The three fish consumption rates were used to calculate levels of concern for each metal, which were fit to a uniform distribution. The results of the uncertainty analyses are reported in Tables 20 (bony fish) and 21 (cockles).

Table 20. Uncertainty Analysis for Human Health Risks from Consumption of Bony Fish.			
Station	Hg	Pb_{adults}	$Pb_{0-6\,yr}$
Perak	68	0.6	92
Selangor	48	0.2	54
Terengganu	4	0.4	100
K. Kedah	0.2	0.8	16
P. Pinang	10	0	97
K.K. Mersuji	2	1	100
Air Tawar	63	0.4	98

Values shown are the probabilities that RQ exceeds one given the uncertainty in seafood consumption rates and MELs at 7 sampling stations in the Straits. For Pb, the probabilities for adults and young children are shown; probability values for older children and pregnant women are between these estimates. MELs are from *Appendix Document* I-M.

Table 21. Uncertainty Analysis for Human Health Risks from Consumption of Cockles.

Station	Cd	Pb_{adults}	$Pb_{0-6\ yr}$
Batu Maung, Penang Island	31	1.5	100
Juru, Mainland Penang	< 1	< 1	6
Bukit Tambun, Mainland Penang	< 1	< 1	8
Telok Tempoyak, Penang Island	< 1	2	100
Jelutong, Penang Island	< 1	< 1	70
Sg. Jejawi, Mainland Penang	< 1	< 1	97
Kuala Sepetang, Perak	< 1	33	100
Lekir, Perak	95	NA	NA
Sg. Besar, Selangor	65	< 1	100
K. Selangor, Selangor	< 1	< 1	100

Values shown are probabilities that RQ exceeds one given the uncertainty in seafood consumption rates and MELs at 10 sampling stations in the Straits. For Pb, the probabilities for adults and young children are shown; probability values for older children and pregnant women are between these estimates. Data are from *Appendix Document* I-M.

stations at which Pb values were very high the levels of concern for Cd were not exceeded, suggesting independent sources of input for these metals. Probabilities that RQ associated with consumption of cockles exceeded one were effectively zero for Cu, Zn and Hg at all stations (see Table 21).

Taking the more recent data on *Anadara granosa* shown in Table 15, an uncertainty analysis was performed by fitting the reported tissue data to a log-normal distribution and considering the Malaysian seafood consumption rates (69, 98 and 116) to fit a uniform distribution. As previously, a Monte Carlo analysis (with 1,000 runs) of these data was performed and the probability estimated with which the RQ, given the uncertainty in both seafood tissue levels and seafood consumption rates, exceeded the values of 1, 10 and 100. In addition, since data were presented for different size classes of bivalves and since these appeared to vary in metal content, uncertainty in RQs was calculated for the smallest and largest size classes separately. The results of these analyses are presented in Table 22.

At Perak, Selangor and Air Tawar stations there was a very high probability that the bony fish tissue levels exceeded levels of concern for Hg. Whereas the likelihood that the levels of concern for Pb were exceeded was very low for adults at all stations, it was extremely high for young children (0-6 yr). At five of the seven sampled areas there was close to a 100% likelihood that the 0-6 year-old level of concern for Pb was exceeded. Probabilities that RQ associated with the consumption of bony fish exceeded one were effectively zero for Cu, Zn and Cd at all stations

There was a high probability that levels of concern for Cd were exceeded at Batu Maung, Lekir and Sg. Besar. At Kuala Sepetang, there was a high probability that the level of concern for Pb for adults was exceeded, and at seven of the nine stations for which Pb was measured, there was a 70-100% probability that the levels of concern for Pb for young children were exceeded. It should also be noted that at several of the

Table 22. Uncertainty Analysis of Data from *Appendix Document* I-M on Metal Levels in *Anadara granosa* from Different Sites in Malaysia.

Metal	Kuala Juru Culture Bed	Kuala Sepetang Culture Bed	Kuala Juru Mudflat (small)	Kuala Juru Mudflat (large)
As	100 (100) [29]	100 (100) [<1]	100 (100) [49]	100 (100) [5]
Cd	70 (1) [<1]	55 (<1)	98 (<1)	62 (<1)
Cu (1-10 yr)	19 (<1)	27 (<1)	79 (<1)	16 (<1)
Fe (1-10 yr)	100 (49) [2]	100 (<1)	100 (77) [<1]	100 (45) [<1]
Fe (adults)	100 (12) [1]	100 (<1)	100 (26) [<1]	100 (<1)
Pb (0-6 yr)	100 (100) [10]	100 (97) [2]	NA	NA
Pb (7-adult)	100 (99) [3]	100 (40) [1]	NA	NA
Pb (pregnant)	100 (86) [1]	100 (6) [1]	NA	NA
Pb (adults)	100 (4) [<1]	91 (1) [<1]	NA	NA
Zn (1-10 yr)	99 (1) [<1]	88 (<1)	100 (<1)	100 (<1)
Zn (adults)	6 (<1)	1 (<1)	26 (<1)	<1

In the two mudflat samples 'small' refers to bivalves in the size class 0.12 g tissue wt and 'large' refers to bivalves in the size class 0.47 g tissue wt. Values presented in the table are probabilities that RQ exceeds 1, (10), and [100] given the uncertainty in MELs (fit to a log-normal distribution) and fish consumption rates (using 69, 98 and 116 g/day fit to a uniform distribution). Analyses were run only for those cases in which RQs from Table 15 were greater than one.

There are clearly problems with As and Pb at both Kuala Juru and Kuala Sepetang and a substantial likelihood that the RQs exceed 10 (for As perhaps even 100). Cd tissue levels also exceed levels of concern with a high likelihood. It is also very likely that bivalves from all three areas exceed the RDAs for Cu, Zn and Fe, particularly for children. For Cu and Zn, it appears unlikely that the RDAs are exceeded by a factor 10. However at Kuala Juru, Fe levels are likely to exceed the RDA for children by a factor 10, which may pose a significant health risk. Despite the fact that metal levels decreased with increasing bivalve size, these differences had a relatively minor influence on the RQ estimates. The largest effects were for Cd and Cu (for 1-10 year-olds).

The importance of uncertainty in metal levels for *Penaeus monodon* was examined (Table 16) and the results are presented in Table 23.

Table 23. Uncertainty Analysis for Data Presented in *Appendix Document* XII on Metal Levels in *Penaeus monodon*.

Metal	Avg. RQ	Probability that RQ > 1 (10) [100]
Cd	5.6	100 (2) [<1]
Cu (1-10 yr)	9.6	100 (54) [<1]
Cu (adults)	1.9	98 (<1)
Cr	1.0	60 (<1)
Fe (1-10 yr)	1.2	75 (1) [<1]
Fe (adults)	0.7	38 (1) [<1]
Mn (1-10 yr)	1.1	63 (<1)
Mn (adults)	0.4	<1
Ni	0.5	<1
Pb (0-6 yr)	240.3	100 (100) [99]
Pb (7-adult)	96.1	100 (100) [48]
Pb (pregnant)	55.5	100 (100) [4]
Pb (adults)	18.7	100 (96) [<1]
Zn (1-10 yr)	0.8	19 (<1)
Zn (adults)	0.3	<1

The avg. RQ was estimated by dividing the geometric mean MEL by the action level associated with a fish consumption rate of 98 g/day. To calculate the probability that RQ exceeds 1 (10) [100], MELs were fit to a triangular distribution with the range giving the minimum and maximum values and the geometric mean the most likely value. Uncertainty in fish consumption rates was estimated as in Table 22.

As was the case for most of the other seafood tissue, Pb was the biggest problem, and the average Pb level in shrimp exceeded the level of concern for young children by 100 times with a probability of close to 100%. Levels of concern for other age groups were exceeded by 10-fold with a probability of 100%. With the exception of Zn, Mn (both for adults) and Ni, the likelihood that the levels of concern were exceeded was high for all remaining metals. This is of most concern for Cd and possibly Cu for children which exceeds the RDA by a factor 10 with a 54% probability.

Much of the tissue data upon which the risk assessments were carried out were from the late 1980s to the early 1990s. Following this, there could either have been increases or decreases in tissue concentrations as a result of changing environmental circumstances. More recent data were not available on tissue concentrations but water metal concentrations from Malaysian stations were obtained for the years 1992-1995 (*Appendix Document* I-M). These data were analyzed to consider if there were any indications of time trends in the environmental concentrations to which seafood species would have been likely to experience. Taking Pb as an example, an analysis of variance (ANOVA) was performed on $\log_{10}$ transformed concentrations. This indicated that there was a significant difference among years (P=0.02), but multiple comparisons between pairs of years (Fisher's LSD test) showed that 1994 was significantly greater than both 1995 and 1992. Hence, there is no obvious trend, either increasing or decreasing, with time and this therefore suggests that trends would have been unlikely in tissue concentrations. Figure 3 suggests a similar conclusion for the other metals. Hence, there is no reason to believe that the risk assessment carried out on the older tissue data should not be applicable as well.

Dermal exposure

There could be risks to human health from dermal exposure to metals, e.g., through bathing. And indeed the Annex in the EC bathing water directive (EEC, 1976) indicates that standards for As, Cd, Cr(VI), Pb and Hg are to be specified, but to date this has not been done.

Pesticides

Concentrations in water

Levels of organochlorine pesticides in selected Malaysian rivers were reported in *Profile* Table 7-10 and for Indonesia in *Profile* chap. 7. There was no information given in the *Profile* for concentrations of other types of pesticides, although it appears that substances other than organochlorines are used (see *Profile* Table 5-12). Also, *Appendix Document* (I-I) reports that carbofuran is used widely on rice fields in Indonesia, and its potential effects are of concern. Data in *Appendix Documents* (I-I and I-M) were no newer than those from the *Profile*. Hence, it has been presumed that they are incorporated in the *Profile* data, and no new analyses have been carried out.

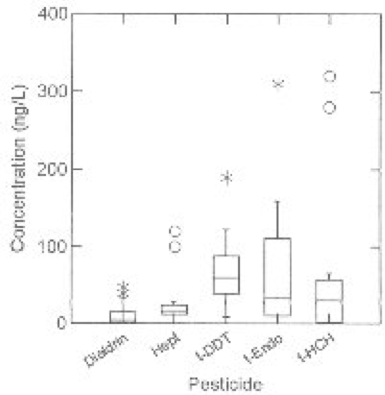
This analysis is based upon the worst-case assumption of little dilution of waters passing from rivers into the Straits. Both highest and median values were taken because the distributions of observations among individual rivers suggest the presence of a few high-concentration 'outliers' with the rest approximating to a normal distribution (Figure 6), but with considerable variability as indicated by the CVs which for four of the five pesticides exceed 100% (Table 24). The highest MECs therefore represent very worst-case scenarios. It is also noted that levels quoted in *Profile* chap. 7 from a 1991 survey for endosulfan and heptachlor in the Klang River were less than the median values quoted in Table 24. Concentrations for aldrin also quoted in the same survey range from 0.005 to 0.061 ng/L, which compare with a standard of 8 ng/L from *Profile* Table 7-11, indicating no cause for concern.

Another way of looking at the data in Table 24 is to take the median values as representative of general conditions within the Straits, $MEC_{Straits}$, and the highest values as indicative of particularly polluted sources, MEC_{Local}, (see below).

Using median MECs and Danish standards all RQs are greater than one, except for dieldrin, and are a factor of about 10 lower than RQs calculated using highest MECs. Using median MECs and aquatic life standards only endosulfan and DDT had RQs greater than one and, apart from DDT, these RQs were a factor of about 10 lower than those based on maximum MECs. Using highest MECs, all risk quotients exceed one.

The concentrations of several of the pesticides were significantly correlated (Figure 7) indicating that certain rivers were contaminated with more than one organochlorine compound. In particular, the Bernam River contained the highest concentrations of t-HCH, heptachlor and t-DDT, and the Selangor River contained the highest concentrations of t-endosulfan and dieldrin.

Figure 6. Box Diagrams of Measured Environmental Concentrations for Pesticides in Water from Malaysian Rivers.



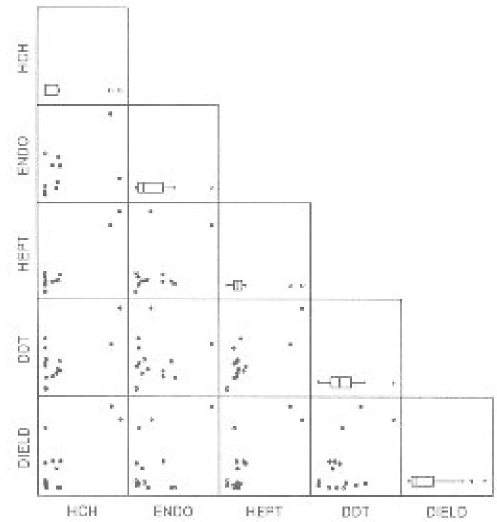
The horizontal lines within the boxes represent medians, the lower and upper box borders represent 25% and 75% quartiles, respectively; whiskers represent the non-outlier range; outside values are shown as asterisks, far outside values as circles.

Table 24. Pesticide Concentrations in Water.

Pesticide	Median MEC [CV]	Highest MEC	Aquatic Life Std.	RQ_{AQ}	Danish Std.	RQ_{DK}
t-HCH	30 [1.7]	320	130	2.5 (0.2)	10	32 (3)
t-Endosulfan	47 [1.1]	310	10	31 (4.7)	1	310 (47)
Heptachlor	16 [1.3]	120	60	2 (0.3)	4	30 (4)
t-DDT	59 [0.7]	190	4	48 (14.8)	2	95 (29.5)
Dieldrin	4 [1.2]	47	8	5.9 (0.5)	10	4.7 (0.4)

RQ analysis based on observations from 15 west coast Malaysian rivers (*Profile* Table 7-10). MEC=measured environmental concentration. CV=coefficient of variation. RQs were calculated by dividing the highest MEC (Median MEC) by either the Aquatic Life Standard (*Profile* Table 7-11)=RQ_{AQ} or the Danish Environmental Standard(Table 24)=RQ_{DK}. MECs and STDs are in units of ng/L.

Figure 7. Scatterplot Matrix Showing Correlations among Pairs of Pesticides in Water from Malaysian Rivers.



The box diagrams shown along the diagonal represent concentration distributions identical to those shown in Figure 6.

For the Indonesian side, there are no comprehensive estimates of pesticide residues in coastal waters. However, measured pesticide concentrations in water from the Siak River (quoted in *Profile* chap. 7) and the risk quotient estimates are shown in Table 25. These data in general do not give cause for concern, even on the basis of highest MECs. Aldrin is a possible exception, but here, the range of recorded values was from 0.04 to 8.17 ng/L, and from the *Profile* there was no indication as to the relative frequency of values in the sampling program.

Uncertainty analysis

As was the case for heavy metals, there are two sources of uncertainty in the estimates of risk from pesticides in water: (a) based on standards; and (b) based on variability in MECs.

With regard to standards, the differences here were not as great as for heavy metals, but the aquatic life standards (*Profile* Table 7-11) and the Danish standards varied by up to a factor of 13 with the Danish standards generally the lower of the two (except for dieldrin, Table 24).

One measure of among-site variability in pesticide concentrations is the coefficient of variation, which as shown in Table 24, is between 70% and 170% of the mean. Again, an important question concerns the likelihood that observed RQs do not differ appreciably from the critical value of one given this variability. A similar approach was taken as for metals, log-transforming the data, and presenting them as means ± 95% confidence limits in Figure 8. On the logarithmic scale in these plots a value of zero is equivalent to the critical value of RQ=1. Despite the considerable variation among sampling sites, these plots indicate that (using the more conservative Danish standards) the log RQs exceed one for DDT (i.e., RQ > 10), exceed zero for endosulfan and heptachlor, and overlap zero for dieldrin and HCH.

PECs

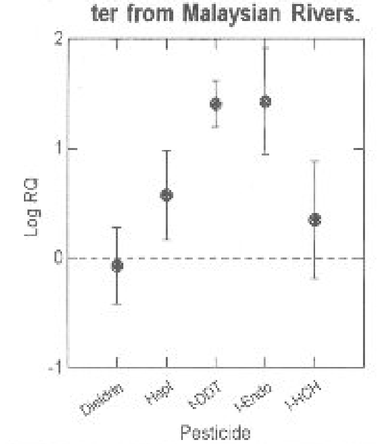
From *Profile* Table 5-15 (DDT use in Malaysian coastal areas bordering the Straits) and assuming that

Table 25. Pesticide Concentrations in Water from the Siak River, Indonesia, in ng/L Published in 1996.

Pesticide	GM MEC	Highest MEC	Log RQ_{Median}	Log $RQ_{Highest}$
pp-DDT	0.15	0.71	-1.43	-0.75
Endrin	0.58	11.17	-1.38	-0.10
Dieldrin	0.20	1.4	-1.60	-0.76
Aldrin	0.57	8.17	-1.15	0
Heptachlor	0.03	0.36	-3.30	-2.22
Endosulfan	0.53	3.15	-1.28	-0.50
gamma-HCH	0.21	2.14	-3.26	-2.25

GM MEC=the geometric mean of the measured environmental concentrations calculated from the ranges cited in the *Profile* in units of ng/L. RQs are based on Aquatic Life Standards (*Profile* Table 7-11) and calculated for both median and highest MECs. The critical value of log RQ is zero.

Figure 8. Means ± 95% Confidence Limits of log RQs for Pesticides in Water from Malaysian Rivers.



Log RQs for all pesticides are greater than or equal to the critical value of zero.

25% emulsifiable concentrate means 250 g ai. per liter and that 90% (Pimentel et al., 1991) of the pesticide applied washes off into the Straits, either via river water or even directly, using the one-compartment model, a $PEC_{Straits}$ of 4 ng DDT/L is calculated. This compares with a median MEC of 59 ng/L. It is interesting that these two figures are within approximately an order of magnitude, and clearly the PEC should be increased by a contribution from the Indonesian side which, given its emphasis on an agricultural economy, is likely to be at least equal to or possibly greater than that from the Malaysian side.

Table 26. Pesticide Concentrations in Sediment.

Pesticide	Log K_{ow}	C_w	Higest MEC (GM MEC) (µg/kg)	Critical Sediment Concentration (µg/kg)	$RQ_{highest}$ (RQ_{GM})
pp-DDT	6.11	4	5,392 (275)	79.4	67.9 (3.5)
Endrin	4.19	14	143 (31.6)	3.4	42.1 (9.3)
Dieldrin	4.78	8	4,374 (187)	7.4	591.1 (25.3)
Aldrin .	4.84	8	4,374 (477)	8.6	508.6 (55.5)
Heptachlor	4.92	60	243 (15.6)	77.0	3.2 (0.2)
Endosulfan-I	3.13	10	1,448 (309)	0.2	7,240 (1,545.0)
gamma-HCH	3.58	130	1,378 (52.5)	7.6	181.3 (6.9)

Log K_{ow} values were obtained from the Pesticide Properties Database (http://www.arsusda.gov/ppdb2.html). For some compounds, K_{ow}s were not available but were calculated from average K_{oc} values given in the database. MEC values are taken from *Profile* Table 7-12. Highest MEC values, as well as geometric mean (GM) MEC values, were used in RQ calculations. Critical sediment concentrations are based on equation 8 with C_w values from the Aquatic Life Standards in *Profile* Table 7-11. RQ=MEC/Critical sediment concentration.

Concentrations in sediments

As for metals, there are no generally accepted quality criteria for pesticide concentrations in sediments. However, following van der Kooij et al. (1991), one can calculate sediment threshold concentrations from published water quality criteria according to the equation:

$$C_{sed} = (C_w \times f_{oc} \times K_{ow} \times 10^{-0.21}) / r \qquad (8)$$

where

- C_{sed} = the critical pesticide concentration in sediment (µg/kg)
- C_w = water quality criteria for the pesticide (µg/L)
- f_{oc} = fraction of organic carbon in the sediment (taken as 0.05)
- K_{ow} = the octanol-water partition coefficient for the pesticide (L/kg)
- r = empirically derived concentration ratio between suspended matter:sediment (= 1.5 for metals and = 2 for organics)

On the basis of the aquatic life standards reported in *Profile* Table 7-11, critical sediment concentrations of organochlorine pesticides (Table 26) were calculated. These critical concentrations were compared to the highest and geometric mean (GM) MECs from *Profile* Table 7-12. As indicated in Table 26, RQs for all pesticides are well above the critical value of one when based on the highest MECs; all but heptachlor have RQs exceeding 10, and endosulfan has an RQ > 1,000. RQs based on GM MECs exceed one for all pesticides except heptachlor. The relative ranking of RQs differs to some extent depending on whether highest or GM MECs are used. However, in both cases endosulfan has an RQ an order of magnitude greater than the other pesticides, with dieldrin and aldrin having the next highest RQs and heptachlor having the lowest RQ.

Uncertainty analysis

There are several sources of uncertainty in this analysis. As was the case for heavy metals, the choice of water quality standard will influence the estimated critical sediment concentration. The aquatic life standards from *Profile* 7-11 were chosen for calculations. However, as Table 24 indicates, these standards can vary by over an order of magnitude compared to e.g., Danish Environmental Standards. Published K_{ow}s can vary widely (e.g., the log K_{ow} for dieldrin was reported to range from 3.692 to 6.2; http://www.arsusda.gov/ppdb2.html), and the choice of K_{ow} will also influence the critical sediment concentration. For compounds in which a range of K_{ow}s was reported (i.e., for dieldrin, endrin and heptachlor), the geometric mean of the reported log K_{ow} was selected. For some compounds (i.e., aldrin, DDT and lindane) only a list of K_{oc}s was reported. For these substances the average K_{oc} was taken and multiplied by 0.6 to estimate K_{ow} (van der Kooij et al., 1991). A standard f_{oc} of 0.05 was used to calculate C_{sed}. Sandy sediments with less than 5% organic carbon will have lower critical sediment concentrations than estimated in Table 26. Finally, there is uncertainty in the RQs arising from variability in the MECs. As only ranges were reported in *Profile* Table 7-12, one cannot assess the frequency with which critical sediment concentrations were exceeded.

Pesticides and human health

There was no information in the *Profile* regarding the concentrations of pesticides in fish or shellfish tissue. However, *Appendix Documents* I-M and V reported tissue values for a variety of species and pesticides from Malaysian stations.

The US FDA has set a level of concern in fish of 0.3 μg/g for dieldrin, aldrin and heptachlor. The level of concern for DDT is 5 μg/g in fish, and that for gamma-HCH (i.e., lindane) ranges between 0.1 and 0.5 μg/g for a variety of foods (no level specifically for fish) (http://vm.cfsan.fda.gov/~lrd/fdaact.txt). As levels of concern for endrin and endosulfan could not be found, a value of 0.3 μg/g was used for these substances. The US FDA levels of concern are based on US average fish consumption rates (about 16 g/day), which give tolerable daily intakes of about 80 μg/day for DDT, 1.6-8 μg/day for gamma-HCH and 4.8 μg/day for the remaining pesticides. If these values are recalculated on the basis of lowest fish consumption rates from the littoral States (about 44 g/day), they give the following levels of concern: 1.8 μg/g for DDT, 0.04-0.18 μg/g for gamma-HCH and 0.11 μg/g for the remaining pesticides. For individuals in the high consumption group (i.e., 116 g/day, Table 11), estimated levels of concern would be 0.69 μg/g for DDT, 0.01-0.07 μg/g for gamma-HCH and 0.04 μg/g for the remaining pesticides.

Comparing the seafood tissue data with the levels of concern for the high consumption group indicated a number of samples with RQs exceeding one, and these are shown in Table 27.

The *Profile* (Table 7.11) provides human health criteria for chlorinated pesticides in water. These are compared to maximum MECs from west coast

Table 27. Organochlorine Pesticide Levels in Tissues of a Variety of Marine Organisms.

Pesticide	Species, Year	Tissue Level (μg/g)	RQ (high consumption group)
Lindane	Fish, 1977	0.001-0.012	0.14-1.2
Lindane	Bivalves, 1985	0.017	0.24-1.7
Dieldrin	Shrimp, 1985	0.094	2.35
Dieldrin	Crabs, 1985	0.232	5.8
Dieldrin	Polychaetes, 1985	0.057	1.42
Dieldrin	Bivalves, 1985	0.052	1.3
Lindane	Bivalves (*Perna viridis*), 1992*	0.181	2.6-18.1
Lindane	Oysters (*Crassostrea belcherei*), 1992*	0.028-0.066	0.4-6.6
Lindane	*Trichogaster trichopterus*, 1982	0.01-0.1	0.14-10
α-endosulfan	*Trichogaster trichopterus*, 1982	5.13	128.2
β-endosulfan	*Trichogaster trichopterus*, 1982	1.70	42.5

Tissue levels are from *Appendix Documents* I-M and V. Values are shown only for those samples in which RQ was close to or exceeded one. For most values the survey year is shown.

*Year original reference published, survey year not given.

Table 28. Comparison of Measured Concentrations of Pesticides in Water with Human Health Criteria.

Pesticide	Human Health Quality Criteria (ng/L)	Maximum MEC from Malaysian Rivers (ng/L)	RQ Malaysia Rivers	Maximum MEC from Siak Estuary, Indonesia (ng/L)	RQ Siak Estuary
Lindane	2,000	320*	0.16	1,110*	0.56
Endosulfan	10,000	310	0.031	580	0.06
Heptachlor	50	120	2.4	130	2.6
DDT	100	190	1.9	1,170	11.7
Dieldrin	20	47	2.35	5,790	289.5

*value is for total HCH

Malaysian rivers (*Profile* Table 7.10) and to maximum MECs measured in the Siak Estuary in November 1995 (*Appendix Document* I-I). The highest concentrations of all pesticides in the Malaysian rivers are from either Sg. Bernam or Sg. Selangor.

Risks to humans from consuming fish can be estimated indirectly given information on tolerable daily intakes of pesticides and bioconcentration factors following van der Kooij et al. (1991):

$$C_{water} = \frac{C_{fish}}{BCF} \tag{9}$$

or

$$C_{sed} = \frac{C_{fish} \times K_{ow} \times 10^{-0.21} \times f_{oc} \times C_w}{BCF \times r} \tag{10}$$

where

C_{water} = measured concentration of pesticide in water (ng/L)

C_{sed} = measured concentration of pesticide in sediment (ng/kg)

BCF = bioconcentration factor (L/kg)

f_{oc} = fraction of organic carbon in the sediment (taken as 0.05)

K_{ow} = octanol-water partition coefficient (L/kg)

C_{fish} = predicted concentration of pesticide in fish tissue (ng/kg)

r = empirically derived concentration ratio between suspended matter:sediment (= 1.5 for metals and = 2 for organics)

Using these equations, the concentrations of pesticides in fish tissues can be predicted from water or sediment concentrations and compared to the level of concern for the pesticide.

Table 29 shows the predicted concentrations of pesticides in fish tissue based on measured water concentrations from Table 24 and estimated BCFs for each pesticide (estimated as K_{ow} x 0.05, van der Kooij et al., 1991) calculated using equation 9. RQ in this case can be calculated as the ratio of the predicted concentration of pesticide in fish tissue/level of concern for the low (44 µg seafood/day) and high (116 µg seafood/day) consumption groups. Fish consumed from areas of the highest water MECs would exceed the levels of concern for both low and high seafood consumption groups for all pesticides with the exception of endosulfan. Fish consumed from areas of median

Table 29. Predicted Pesticide Concentrations for Fish Tissue based on Water Concentrations (see equation 9).

Pesticide	BCF (L/kg)	C_{fish} (µg/g) Median, High	RQ_{44} Median, High	RQ_{116} Median, High
gamma-HCH	190	0.006, 0.061	0.15, 1.52	0.6, 6.1
Endosulfan	67	0.003, 0.021	0.03, 0.19	0.08, 0.52
Heptachlor	4,159	0.066, 0.499	0.6, 4.54	1.65, 12.48
DDT	64,412	3.800, 12.238	2.09, 6.72	5.51, 17.74
Dieldrin	3,013	0.012, 0.142	0.11, 1.29	0.3, 3.55

Human health RQs were calculated as the ratio of fish tissue concentration/level of concern for each pesticide for low (44 g/day) and high (116 g/day) seafood consumption groups. For gamma-HCH levels of concern of 0.04 and 0.01 for the low and high consumption groups, respectively, were used. Corresponding values for the remaining pesticides were: endosulfan, heptachlor and dieldrin, 0.11 and 0.04; DDT, 1.82 and 0.69. Calculations were made using MECs from Table 24. Median and highest C_{fish} values were calculated using equation 9 with median and highest water MECs, respectively.

MECs are predicted to have tissue concentrations exceeding the level of concern for the high consumption group for DDT and heptachlor but only for DDT for the low consumption group.

Table 30 shows the predicted concentrations of pesticides in fish tissue based on measured sediment concentrations from Table 26 and calculated using equation 10. RQs are again calculated as the ratio of the predicted concentration of pesticide in fish tissue/ level of concern for each pesticide and seafood consumption group. With the exception of DDT, heptachlor and endrin for the low consumption group, all pesticides exceed the levels of concern in terms of both median and highest predicted tissue levels.

Table 30. Predicted Pesticide Concentrations in Fish Tissue based on Sediment Concentrations (equation 10).

Pesticide	C_{fish} (µg/g) Median, Highest	RQ_{44}	RQ_{116}
DDT	0.892, 17.490	0.500, 9.600	1.300, 25.340
Endrin	0.102, 0.464	0.920, 4.220	2.560, 11.600
Dieldrin	0.606, 14.188	5.500, 128.980	15.160, 354.700
Aldrin	1.548, 14.186	14.080, 128.960	38.700, 354.640
Heptachlor	0.050, 0.788	0.460, 7.160	1.240, 19.700
Endosulfan	0.996, 4.666	9.060, 42.420	24.900, 116.640
gamma-HCH	0.170, 4.468	4.240, 111.700	17.000, 446.800

MECs are from Table 26. Geometric mean (GM) and highest C_{fish} values were calculated and compared to levels of concern for low (44 g/day) and high (116 g/day) seafood consumption groups. Levels of concern as in Table 29. BCF values were estimated as K_{ow} x 0.05 (van der Kooij et al., 1991).

It is interesting to note that the rank order of pesticides in terms of risk depends upon whether tissue concentrations are predicted from water or sediment concentrations. This probably reflects differences in partitioning of the different pesticides between water and sediment phases.

Using the human health criteria for pesticides in water from *Profile* Table 7.11 and equation 9, the maximum concentrations of pesticides can be predicted in seafood tissue derived from waters meeting the human health criteria. These are shown in Table 31 and compared to the levels of concern for low and high consumption groups. This analysis suggests that although human health criteria for water may be fulfilled, there may nevertheless be risks associated with the consumption of moderate to large quantities of fish or shellfish from such areas.

Uncertainty analysis

These estimates clearly demonstrate the importance of accurately estimating fish consumption rate for determining the human health risks associated with ingesting contaminated seafood. Other sources of uncertainty include additional dietary and non-dietary routes of pesticide exposure as well as uncertainties involved in estimating water and sediment concentrations.

The general approach used therefore is to compare worst-case intakes with precautionary, tolerable standards. Very often this will overestimate long-term intake, by focusing on worst-case assumptions about intake; i.e., individuals consume high levels of fish with high levels of contamination. Hence this does not take into account variable consumption and residue levels that may occur in nature.

Table 31. Predicted Tissue Concentrations (C_{fish}) of Organochlorine Pesticides for Seafood Derived from Areas Meeting the Human Health Criteria for Water Concentrations.

Pesticide	BCF (L/kg)	C_{fish} (µg/g)	RQ_{44}	RQ_{116}
DDT	64,412	6.44	3.54	9.33
Endrin	774	0.05	0.45	1.25
Dieldrin	3,013	0.06	0.55	1.5
Aldrin	3,459	0.07	0.64	1.75
Heptachlor	4,159	0.21	1.91	5.25
Endosulfan	67	0.67	6.10	16.75
Lindane	190	0.38	9.5	38.0

RQs were estimated for the low and high seafood consumption groups by dividing the predicted tissue concentration by the level of concern for each consumption group (from Table 12) (see Table 30).

Simulation techniques can be used to take this variability into account. As already noted, the traditional method involves Monte Carlo techniques. The Ministry of Agriculture, Fisheries and Food (MAFF), UK is currently assessing an alternative technique for use in evaluating pesticide contamination on fruit and vegetables called Latin Hypercube. In this technique, sampling is stratified from known probability distributions of consumption and food unit contamination. It thereby defines permutations of consumption and residue levels that can be taken into account in making predictions about likely exceedance of thresholds at any one meal, to predict the likelihood of acute as well as long-term exposure. Because the Latin Hypercube technique involves stratified sampling, it requires fewer iterations than the Monte Carlo method. Clearly, although this approach has been developed in the context of pesticide contamination, it could also be used for other kinds of contamination including heavy metals (see above).

Dermal exposure

There could be risks to human health from dermal exposure to pesticides, e.g., through bathing. And indeed the Annex in the EC bathing water directive (EEC, 1976) indicates that standards for parathion, γ-HCH and dieldrin are to be specified, but to date this has not been done.

Tributyltin

Concentrations in water

Using an environmental quality standard of 2 ng/L (as set in the UK; Langston, 1996), it is clear that all concentrations of tributyltin (TBT) in water samples from the west coast of Peninsular Malaysia as recorded in *Profile* Table 7-25 are in excess of this; and RQ values range between approximately one for Sg. Buloh to 140 for Port Klang (South Port Area). The distributions of concentrations and log RQs appeared normal with Port Klang detected as an outlier (log RQ=2.15). A surprising observation is that levels for open waters (four sites recorded in the Table), although on average higher, were not significantly different from levels for port areas (six sites recorded in the Table): mean for open water=40.2 (± 40.7=SD) ng/L; mean for Ports=75.5 (± 102.2=SD) ng/L; t_s=0.76, $P > 0.05$). Clearly, there is considerable variability in both sets of data with the highest value from Port Klang (South Port Area) and the lowest from Pulau Jemor, but nevertheless with considerable overlap between both kinds of area. Excluding the Port Klang outlier, the mean log RQ was 1.28 (95% CL=1.14-1.43) for the remaining stations.

Concentrations in sediment

In contrast with water column values, there are no quality standards set for TBT in sediments other than to prevent levels from increasing (Langston, 1996). However, based on available evidence, chronic exposure to approximately 0.1 μg TBT/g sediment dry wt represents potential problems for survival of sediment-dwelling molluscs, and above 1 μg TBT/g sed dry wt even tolerant polychaetes seem to be at risk. Waite et al. (1991) produced a classification of degree of contamination of UK sediments in areas used for recreational yachting and designated the limit for lightly-contaminated sites as greater than 10 ng/g sed dry wt, medium-contaminated sites as greater than 60 ng/g sed dry wt and highly-contaminated sites as greater than 300 ng/g sed dry wt. Even presuming a high water:sediment ratio of 50%, the levels of TBT in sediments as recorded in *Profile* Table 7-24 were well below 0.1 μg TBT/g dry wt and ranged from about 1 to 60 ng/g dry wt, classifying them as no more than lightly-contaminated sites according to the above criteria. This is surprising since sediments are thought to represent a potentially long-lasting reservoir for TBT with half-lives of the order of 1-5 years (in aerobic sediment) to possibly decades (in anaerobic muds) (Langston, 1996). The analysis suggests, therefore, that determining the extent to which there may be risks associated with TBT contamination, especially in sediments, needs more careful attention, perhaps through more intensive sediment monitoring of TBT in sediments emphasizing closed areas (e.g., harbors) and major shipping lanes. Analyses of a number of gastropod species along the southeastern Malacca Straits (Malaysian side) indicated a substantial incidence of imposex which was related to distance to the nearest shipping route (see Figure 2) and which was particularly high at the southernmost stations in the vicinity of the Strait of Johor (Swennen et al., 1996). Since TBT is the only pollutant presently known to cause such reproductive abnormalities, these observations provide strong evidence for significant TBT pollution.

TBT and human health

There are no formal standards for tissue concentrations of TBT for bivalves and fishes, but information in the literature suggests that tissue concentrations less than 1 μg TBT/g tissue dry wt are unlikely to have adverse biological effects (Waldock, 1994; Willows, 1994) and it is assumed that human consumption of seafood from waters meeting the water quality standard for TBT will not adversely affect human health (Zabel et al., 1988). Swennen et al. (1996) reported 7% imposex in gastropods from *Anadara* and *Paphia* fishery areas near Kuala Selangor, and Tong et al. (1996) found significant contamination with TBT in *Anadara granosa*, *Paphia* sp. and *Perna viridis* from the fish markets in Kuala Lumpur and Petaling Jaya. Even presuming a high water:tissue ratio of 10:1, the TBT concentrations from Tong et al. (1996) and shown in *Profile* Table 7-25, for bivalves sampled from markets are all well below the threshold of biological effects for the organisms sampled, ranging from less than 5 ng/g dry wt to 23.5 ng/g dry wt, and such concentrations are presumably not hazardous for human consumption. However, it is noted that the *Profile* (chap. 7) states that, 'the presence of TBT in the Straits may have a serious implication with respect to accumulation in food fishes and shellfish, which are staple foods. In view of its possible serious human health and economic implication, the use of TBT requires special control'.

In conclusion, despite indications that measured concentrations of TBT in water and sediments are relatively low, there is evidence for biological effects (i.e., imposex in female gastropods) of the type usually associated with TBT pollution. Therefore, further study of the environmental fate and ecological/human health effects of this substance is necessary.

Nutrients and Oxygen Demand

Nitrogen and phosphorus

Using loads quoted for Malaysia in *Profile* Table 5-8 and for Singapore in *Profile* Table 5-5, a $PEC_{Straits}$ of 1.8 μg/L for nitrogen and 2.6 μg/L for phosphorus was calculated using the one-compartment model. MECs for these nutrients from *Profile* (chap. 2) are 0.98 μg/L for nitrogen and 0.42 μg/L for phosphorus (both maxima for surface water). It is encouraging that these are within the same orders of magnitude. Concentrations reported in the Siak River (I-I) are considerably higher than these: 37-150 μg/L for NO_3-N; 1.5-8 μg/L for NO_2-N; 7-18 μg/L for PO_4-P . There has been much debate over critical nutrient levels for the marine environment, but there would appear to be no straightforward relationship between either nutrient concentrations or N:P ratios and eutrophication (Gray, 1992). However, the following values have been recorded for apparently unpolluted open waters: PO_4^{2-}: 0.5-0.9 μM; $NO_3^- + NO_2^-$: 6-12 μM (North Sea Task Force, 1993). Nutrient values for the North Sea in winter range from 70 to 560 μg/L for NO_3-N with most in open water tending to 100, and from 12 to 28 μg/L for PO_4-P with most in open waters less than 20 (Eisma, 1987). The People's Republic of China uses standards of 100-300 μg/L for nitrogen and 15-45 μg/L for phosphorus in coastal seas (*Appendix Document* XI). Indonesia uses a waste standard for biotic use of 10,000 μg/L NO_3 and 60 μg/L for NO_2; no standards were shown for PO_4.

The risk assessment therefore suggests that nutrients are unlikely to cause ecological problems. On the other hand, there are signs of eutrophication within the Straits (*Profile* chap. 7), and so this suggests that a more detailed risk assessment should be carried out.

Oxygen (BOD, COD and dissolved oxygen)

Biochemical oxygen demand (BOD) measures the amount of dissolved oxygen required to oxidize biodegradable organic compounds. Chemical oxygen demand (COD) measures the amount of dissolved oxygen required to oxidize organic compounds that are not biologically degradable (Frankel, 1995). Observations from the coastal waters of the west coast of Peninsular Malaysia give BODs ranging from 1.33 mg/L to 9.95 mg/L; however 'most other stations show levels between 3 to 4 mg/L'. On the eastern coast of North Sumatra, the BOD values ranged from 3.3 to 56.6 mg/L. This kind of range for Indonesia is confirmed in *Appendix Document* VII. Here, values are reported between 0.4 and 400 mg/L, but most are less than 30 mg/L. The BOD levels at four sampling stations in the Bengkalis Straits of Riau ranged from 8.14 to 13.64 mg/L with the highest levels usually found close to industries or sewage outfalls (*Profile* chap. 7). From *Appendix Document* VII highest values are associated with ports; and it is implied that offshore BOD is negligible.

PECs

To calculate a $PEC_{Straits}$, all BOD loads are summed and all sources included (domestic, agricultural and industrial) for coastal inputs from Malaysia (given in *Profile* Table 5-8), Singapore (*Profile* Table 5-5) and Indonesia (*Profile* Tables 5-2 and 5-7). On this basis, a total input of 3 x 10^5 tonnes/yr to the Straits was calculated, which from the one-compartment model gives a $PEC_{Straits}$ of 0.03 mg/L. This is low compared with the MECs. However, here it is probably more appropriate to consider local effects of discharges and therefore to make PEC_{Local} calculations.

Another possible way of calculating the domestic contribution to BOD to obtain a $PEC_{Straits}$ would be from the effective population of the littoral States (say approximately 20 million for Indonesia and Malaysia with Singapore excluded because of the provision of good STW) multiplied by the average BOD production, 1.5 x 10^7 mg/person/yr, as taken from *Profile* Table 5-2, which also gives a $PEC_{Straits}$ of 0.03 mg/L. In making this calculation, the data quoted for Indonesia were used but it was noted that these were for a coastal population of 110.76 million. It was presumed that this refers to the coastal population for Indonesia as a whole, rather than for the coastal population for the Malacca Straits (which is taken to be about 10.9 million; *Profile* chap. 2). It was noted from *Profile* chap. 5 that in 1989 the total coastal loading from sewage discharge for Malaysia, Indonesia and Singapore was about 5,000 tonnes per day, and was expected to increase to 6,000 tonnes by the year 2000.

From the calculations from *Profile* Table 5-2, it is estimated that an average BOD production per million of about 40 tonnes, which multiplying by the estimate of total littoral population (20 million) gives 800 tonnes per day. It is presumed, therefore, that the levels quoted in the *Profile* for 1989 and 2000 represent coastal areas in general and not specifically the Malacca Straits. Clearly, were this presumption to be incorrect, the predicted BOD would increase by about 5 or 6 times.

A PNEC/STD could not be found for BODs in estuarine/marine circumstances. However, for rivers, it is noted that those with a BOD of < 2 mg/L are considered not polluted (Clark, 1992). Hence, taking 2 mg/L as an interim standard, the RQs are calculated as shown in Table 32. Effluent standards are quoted as ten times this (*Appendix Documents* I-S, I-M and X), but dilution in the receiving waters is presumed.

Table 32. Measured and Predicted BOD Values for Various Regions along the Straits.

Source	BOD (mg/L)	Log RQ
Malaysian West Coast (full range)	3.64	0.26
Malaysian West Coast (most stations)	3.0	0.24
East Coast of North Sumatra	13.67	0.83
Bengkalis Straits of Riau	10.51	0.72
Mean ± CL for RQs based on MECs	0.51	(±0.31)
$PEC_{Straits}$	0.03	-1.82

For all MECs, a geometric mean BOD was calculated from the ranges given (*Profile* chap. 7). The $PEC_{Straits}$ was estimated using a simple, one-compartment model of the Straits. RQs were calculated using a standard of 2 mg/L.

Hence, the mean RQ for the MECs significantly exceeded the critical value of zero, with all of the individual values above zero. However, it is unclear how unbiased the samples were upon which this conclusion is based in terms of the Straits as a whole. The uncertainties associated with the choice of interim standard and the interpretation of the PEC value suggest that a more thorough analysis is required of this kind of contamination.

The data for COD were more limited, and for the eastern coast of North Sumatra ranged from 10.8 to 766.1 mg/L and for the Bengkalis Straits of Riau from 20.16 to 32.64 mg/L (*Profile* chap. 7). A coastal COD of 80-100 mg/L is quoted for Indonesia (*Appendix Document* VII). An effluent standard for COD of 30 mg/L is used by Singapore (*Appendix Document* X) and of 50 mg/L by Malaysia (I-M). Hence, presuming as for BOD, this should be reduced by a factor ten to represent dilution, RQs would be greater than one.

The data for dissolved oxygen were even more limited with a range of 0.78 to 6.93 mg/L from a survey of 43 stations along the Straits and a report from routine surveys by the Malaysian DOE between 1989 and 1994 of 'some areas' with DO content between 1 and 2 mg/L.

Total Suspended Solids

The interim standard for total suspended solids (TSS) adopted by the DOE in Malaysia is 50 mg/L. This was applied to the data quoted from routine surveys conducted by the DOE in Malaysia (quoted in *Profile* chap. 7) and the data in *Profile* Table 7-2. However, it is noted that *Appendix Document* I-I reports that suspended solids as low as 6 mg/L can significantly impact respiration of hard corals.

To calculate $PEC_{Straits}$, total inputs from Malaysia (given in *Profile* Table 5-8) and Singapore (*Profile* Table 5-5) were used to give a value of 288,554 tonnes/yr. There were no data in the *Profile* for Indonesia. Data in *Appendix Document* I-I suggest a 5 m tonnes/yr input for Indonesia, but it was unclear how much of this was of relevance for the Straits. Presuming that Indonesian sources generate as much again as Malaysia and Singapore gives a total of about 6×10^5 tonnes/yr. Using the one-compartment model, this translates into a $PEC_{Straits}$ of about 0.1 mg/L.

Log RQs are summarized in Table 33. For MECs from *Profile* Table 7-2, log RQs approximated to the critical value of zero, signaling cause for concern. However, the log RQ from the $PEC_{Straits}$ was considerably less than zero. This is not surprising because the $PEC_{Straits}$ neither took account of contributions from the Indonesian side nor any background levels in the Straits themselves due to, e.g., erosion or dredging. On the other hand, a considerable amount of uncertainty associated with the MEC data from *Profile* Table 7-2 is noted, as indicated by the fact that three out of the seven stations sampled exceeded the interim standard in less than 50% of the samples. From the routine surveys conducted by the Malaysian DOE, TSS ranged from 100 mg/L up to a maximum of 1,395 mg/L giving log RQs as indicated in Table 33. Dow (1995, cited in the *Profile*) reported that most stations in the Malaysian routine survey were close to the interim standard. On the other hand, mean TSSs from Malaysia reported in *Appendix Document* I-M from 1992 to 1995 range between about 300 and 700 mg/L.

It was also unclear as to how the interim standard had been calculated. A PNEC for TSS should take into account a number of biological considerations. Potential ecological effects of TSS include reduced light penetration (which may have a negative impact on photosynthetic organisms including corals), reduced visibility, destroyed spawning areas, reduced food supplies, reduced plant cover, anaerobic conditions caused by trapped organic matter, flocculent planktonic algae, adsorption or absorption of organic molecules and ions, adsorption of oil and toxic components, and impaired respiration caused by particles floating and blocking gills (Frankel, 1995). The interim standard might underestimate the effects of suspended solids on corals (*Appendix Document* I-I). Silt particles also trap toxicants and so enter food chains of importance to humans (*Profile* chap. 7).

Relative importance of defined sources of TSS

On the presumption that TSS does create environmental risks it will be necessary to quantify contributions from the major sources, as a basis for management and prioritization. Our $PEC_{Straits}$ was based on inputs from domestic sewage and industrial sources from the Malaysian side and Singapore. There are two other possible approaches to estimating the domestic contribution, i.e., by multiplying the effective littoral population (currently about 20 million excluding Singapore because it has good STW) by either (1) the fecal production per person or (2) the average TSS per person. On the basis of the latter, and taking TSS loadings from *Profile* Table 5-3, and adjusting to a total littoral population of 20 million (which assumes that TSS per capita on the Indonesian side is similar to that on the Malaysian), together with the one-compartment model gives a $PEC_{Straits}$ for domestic inputs of 0.02 mg/L. From *Profile* Table 5-3, one can also calculate a PEC_{Local} for selected rivers. Highest PECs were from the Klang and Perak Rivers, calculated by taking loadings (i.e., mass

Table 33. Risk Quotients for Total Suspended Solids Estimated from Measured and Predicted Environmental Concentrations.

Data Source	Log RQ
Based on range of values from routine DOE Malaysia survey (*Profile*)	about 0.3-1.5
Based on *Profile* Table 7-2	mean 0.21 (±0.285=95%CL)
Based on $PEC_{Straits}$	-2.7

to river) and dividing by the average annual flow rates, which were estimated from data given in *Profile* Table 2-3. The PEC for Klang is thus 16 mg/L and for Perak 2.2 mg/L. Both of these figures are considerably greater than the $PEC_{Straits}$ but are still below the interim standard of 50 mg/L. However, for the Siak River on the Indonesian side loadings are reported of mostly >50 mg/L and often >100 mg/L (*Appendix Document* VI). This is equal to or greater than the standard and suggests a bigger contribution of suspended solids from the Indonesian side.

Another possible major source of suspended solids is from the aquaculture industry, and it might be necessary to predict a maximum likely generation of solids from a particular area of culture and/or to assess likely generation of solids from existing areas. One way of doing this could be on the basis of production figures as follows: presume that energy and water contents of food and tissue are the same, then it is possible to use energy conversion efficiencies (Calow, 1977) to compute fecal production rates from tissue production rates viz. (1) presume that production (P) represents 40% of absorbed intake (A) of food (hence, $P/0.4=A$); (2) presume that 80% of food ingested is absorbed so that 20% of ingested food is lost as feces (F), hence $0.2I=F$ and therefore $A=4F$; 4F can be equated with P/0.4, so that $F=0.6P$. In other words, an amount equivalent to 60% of production is lost as feces, and on the assumption that these do not decompose, 60% production can be taken as the maximum contribution to suspended solids. Taking data from *Profile* Table 3-5 for brackish water pond fisheries of Sumatra and from *Profile* Table 3-10 for brackish water production in Malaysia, both for 1993, a production level of 39×10^{12} mg for Sumatra and 90×10^{12} mg for Malaysia is calculated giving approximately 129×10^{12} mg in total, which, by applying the usual one-compartment model, would maximally contribute 0.013 mg/L to the total suspended solids in the Straits. Assuming that the suspended solids measurements (*Profile* Tables 5-5 and 5-8) and the aquaculture activities (*Profile* Tables 3-5 and 3-10) are both representative of the conditions to date, one can say that aquaculture is contributing a maximum of about 13% of TSS and probably considerably less than this. Clearly, as the intensity of the aquaculture activity on both sides of the Straits changes, this situation is likely to change. Also, these calculations do not take into account the contribution of wasted food materials lost from aquaculture facilities and these may be as great, if not greater than fecal losses.

Another significant source of suspended solids is agricultural runoff (e.g., from pig farms). *Profile* Table 5-14 suggests that about 35,000 tonnes/yr of TSS go to river from approximately 0.5 million pigs, so the potential contribution of TSS per pig per year is approximately 0.1 tonne. From *Profile* Table 5-13 and information in the text, the total pig population for the west coast of Malaysia is about 2.5 million, giving a total TSS loading into the Straits of 250,000 tonnes/yr. This gives a contribution to TSS in the Straits of 0.025 mg/L, not taking account of contributions from pig farms in Indonesia and Singapore.

Table 34 summarizes the PECs for TSS derived from various sources and compares them with a PEC_{Total}. The individual components account for approximately 6% of the total PEC. As will be clear from the table, one source of difference is that data from all littoral States have not been consistently used; and another is that contributions from industrial sources have been ignored. Calculations from *Profile* Table 5-8 on West Coast Malaysian inputs suggest that these could account for about 50% of total TSS input. Interestingly, the textile industry alone seemed to have a 38% contribution.

Table 34. Relative Contribution of TSS from Different Sources Used to Estimate $PEC_{Straits}$.

Source-Specific PECs	PEC (mg/L)	Data Used
PEC_{Total}	0.1	From all States
$PEC_{Domestic}$	0.02	From all States
$PEC_{Piggeries}$	0.025	Indonesia and Malaysia
$PEC_{Aquaculture}$	0.013	Largely Malaysia

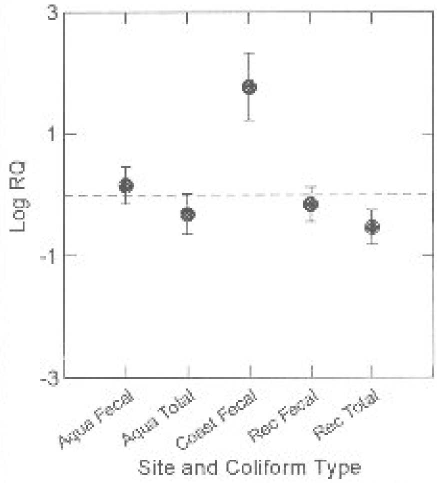
The other important comparison is between PEC_{Total} and MECs for the Straits. The latter are generally close to or greater than the interim standard of 50 mg/L. Hence, there is a considerable difference here, that has to suggest an additional substantial source of suspended solids. From the retrospective analysis, there are indications that this is associated with mangrove deforestation.

In conclusion, therefore, there is considerable uncertainty in these risk assessments both in terms of the validity of the interim standard and the MECs and PECs. Much of the variability in the MECs might be described as stochastic and due to variation in conditions among sites and among sampling occasions. The uncertainty associated with some of the assumptions used in calculating the PECs has already been discussed. There is, of course prima facie evidence for concern, and this needs following up with a careful consideration of the standard, the monitoring program and the assumptions incorporated into the PEC.

Coliforms

Coliform counts for marine waters, aquaculture sites and recreational sites on the west coast of Peninsular Malaysia from mid- to late 1980s to early 1990s are summarized, respectively, in *Profile* Tables 7-18, 7-19 and 7-20. No more up-to-date data were given in the *Appendix Documents*. A fecal coliform standard of 100 MPN/100 ml was adopted as used by the Malaysian DOE and quoted in the bathing water Directive of the EU (EEC, 1976). The total coliform standard of 500 MPN/100 ml was taken from the annex of the EEC Directive (1976). It is also of interest to note that the EEC Directive gives interim standards for total and fecal coliforms, respectively, as 10,000 and 2,000 MPN/100 ml. Moreover, the EU Directive on the quality required of shellfish waters (EEC, 1979), designed to 'contribute to the high quality of shellfish products directly edible by man', specifies a guideline standard for fecal coliforms of 300 MPN in the shellfish flesh and intervalvular liquid noting, however, that 'it is essential that this value be observed in waters in which live shellfish directly edible by man'. Singapore appears to use a standard of 1,000 MPN (*Appendix Document* IX). Taking the most strict guidelines for total and fecal coliforms, respectively, as 500 and 100 MPN/100 ml, the raw data in *Profile* Tables 7-18, 7-19 and 7-20 are converted to risk quotients. Figure 9 shows means and confidence limits (after the usual logarithmic transformation, see above) for the various categories of sites.

Figure 9. Comparison of mean log RQs for Total and Fecal Coliforms among Different Types of Stations in the Straits.

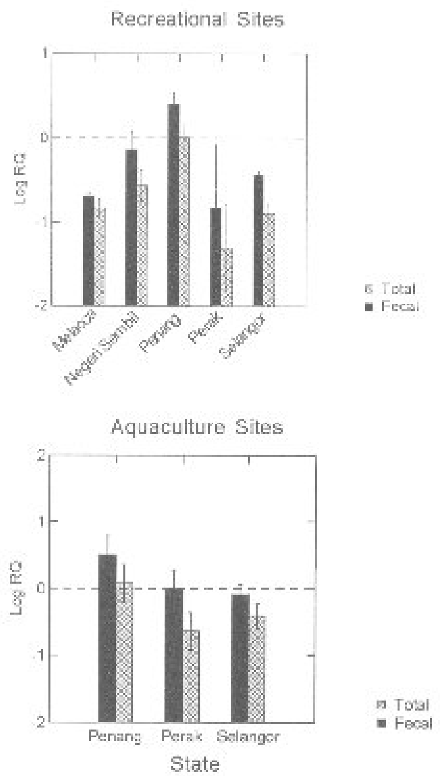


Error bars represent 95% confidence limits. A dashed line is drawn through the critical log RQ value of zero. 'Coastal' refers to coastal sites reported in *Profile* Table 7-18; 'Aqua' refers to aquaculture sites on the west coast of Peninsular Malaysia reported in *Profile* Table 7-19; 'Rec' refers to recreational sites on the west coast of Peninsular Malaysia reported in *Profile* Table 7-20.

It is surprising that the risk quotients for coastal marine waters were all appreciably above the critical log RQ=0. Even noting that the percentage exceedance of standards in individual samples ranged between 36 and 76, presumably indicating some variability in samples taken at different times and hence introducing stochastic uncertainty, there is still, in general, considerable cause for concern. The average log RQ for both total and fecal coliforms at aquaculture sites did not differ significantly from zero, whereas average log RQ for total coliforms, but not fecal coliforms, at recreational sites was less than zero.

Average coliform concentrations at recreational and aquaculture sites in different Malaysian states are shown in Figure 10. Applying analysis of variance to coliform counts among states as recorded in *Profile* Table 7-20 for recreational sites, significant differences were found for both total coliforms and fecal coliforms (total coliforms: ANOVA; $F=4.77$; $P=0.015$; fecal coliforms: $F=6.17$; $P=0.006$) among sites. In both cases, RQs for the Penang beaches were significantly higher than Melacca; for fecal coliforms, Penang beaches were significantly higher than all other beaches (Tukey post-hoc comparisons; $P \leq 0.05$). Omitting the Penang data and recalculating average log RQs for the other states,

Figure 10. Comparison among Different Malaysian States of log RQs for Total and Fecal Coliforms at Recreational and Aquaculture Sites.



Measured concentrations of coliforms are from *Profile* Tables 7-19 and 7-20. Error bars represent SEM and a dashed line is drawn through the critical log RQ value of zero.

a value for total coliforms of -0.832 (95% CL=-1.07 - -0.60) and a value for fecal coliforms of -0.469 (95% CL = -0.73 - -0.20) was found; i.e., both now significantly less than the critical threshold value of log RQ=0. On the other hand, there were no significant differences among aquaculture sites in the different states in terms of either total (ANOVA; F=2.36; P=0.16) or fecal coliforms (ANOVA; F=2.11; P=0.18).

Counts for Singapore waters (Strait of Johor and Strait of Singapore) were below the critical count of 1,000 MPN/100 ml for more than 70% of the sites (*Appendix Document* IX).

Oil, Grease, Petroleum Hydrocarbons and Tar Balls

All these substances comprise very complex mixtures of thousands of organic compounds with different behaviors and hence different possible effects on marine life and ultimately human health. Once released into the environment, all of these compounds are subject to continuous and variable changes due to biological degradation, photooxidation, etc. There are also background levels of hydrocarbons, for example arising from natural biosynthetic processes. Hence, both the summary data given in *Profile* Tables 7-13 to 7-16 and the standards quoted in *Profile* chap. 7 from FAO and Marchand et al. (1982, cited in Abdullah and Samah, 1996) (see *Appendix Document* II) have to be treated with some caution. For the refined risk assessment, the focus is on the most up-to-date data available from *Appendix Document* II for water column and sediments in both the Malacca Straits and the Strait of Johor. Here, a distinction is made between 'oil and grease' representing both biogenic and petrogenic substances and specifically 'petrogenic hydrocarbons' measured in terms of both crysene equivalents and Seligi crude oil equivalents.

Critical concentrations

Table 35 summarizes critical concentrations obtained from various sources and of relevance for assessing risks from hydrocarbon contamination within the Malacca Straits.

The standards used are more conservative than most and have been estimated in the following way. As quoted in *Profile* chap. 7, mesocosm studies have shown that sublethal effects can be observed at hydrocarbon concentrations as low as 20 µg/L in *Mytilus edulis*. Reductions in diversity and changes in size structure of phytoplankton and zooplankton were observed at concentrations down to 75 µg/L. Also, as far as the standards are concerned, it can be argued that many of the hydrocarbons of concern for marine ecosystems will be polycyclic aromatic hydrocarbons (PAHs). These are relatively resistant to degradation and so are likely to accumulate relative to other fractions, especially in sediments. Much of the oil entering the marine environment (especially that from land-based sources) consists of used oils which are enriched in PAH

Table 35. Critical Water Concentrations for Hydrocarbons.		
Standard	mg/L	Source
Malaysian water quality criteria for class I waters, very sensitive aquatic species	Background	*Appendix Document* II
Malaysian water quality criteria for class II waters, mineral oil, sensitive aquatic species	0.04	*Appendix Document* II
Malaysian water quality criteria for class II waters, emulsifiable & edible oil, sensitive aquatic species	7	*Appendix Document* II
Aquaculture, hatchery, mineral oil, sensitive aquatic species	Less than 0.01	*Appendix Document* II
Oil & grease, recommended standard for Malaysia	1	*Appendix Document* II
GESAMP criteria to prevent tainting and sublethal effects to embryos and larval invertebrates, oil components	0.01	*Appendix Document* XII
LOEC dissolved & emulsified oil products, benthic crustaceans, fish, phytoplankton (Patin, 1982)	0.01	*Appendix Document* XII
Florida Dept. of Natural Resources (1992) for protection of recreation, fish & wildlife	< 5	*Appendix Document* XII
ASEAN marine water criteria for oil & grease (water soluble fraction)	0.14	*Appendix Document* XII
For most sensitive species, for oil & grease	0.01	*Appendix Document* XII
FAO hydrocarbon content in 'unpolluted' waters	0.0025	*Appendix Document* II
Lowest 96h LC_{50} (Betton, 1994) divided by 1,000	0.001	Calow and Forbes, 1997

(GESAMP, 1993). These compounds tend to be extremely toxic, and many are known carcinogens (Neff, 1979). On this basis, it is noted from Kennish (1994) that a lowest LC_{50} value for PAHs is between 50 and 300 µg/L for marine fauna. Applying a safety factor of 1,000 would therefore give a PNEC of approximately 0.1 µg/L (cf. FAO standard of 2.5 µg/L). Now, taking a log K_{ow} of 5 (which is representative of the larger, more persistent PAH, Neff, 1979) and applying equation 8 gives a PNEC for sediments of 0.15 mg/kg dry wt (cf. the value of 100 mg/kg attributed to Marchand et al., 1982 cited in Abdullah and Samah, 1996). Moreover, the latter has to be put-in-the context of reported values in the literature of 100 mg/kg total PAH for heavily polluted sites and background measures for the abyssal plain of 0.055 mg/kg and for Alaskan marine sediments of 0.005-0.113 mg/kg (Kennish, 1994).

Table 36. Critical Sediment Concentrations for Hydrocarbons.		
Standard	mg/kg	Source
FAO hydrocarbon content in 'unpolluted' sediment	100	*Appendix Document* II
Background, abyssal plain (Kennish, 1994)	0.055	Forbes and Calow (1997)
Background, Alaskan marine (Kennish, 1994)	0.005	Forbes and Calow (1997)
PNEC for oil and hydrocarbon	1.5	Forbes and Calow (1997)

Trying to calculate a PNEC for oils is even more problematic. But it is noted from Betton (1994) that 45% of the data on oil ecotoxicity in the literature indicates 96h LC_{50} values between 1 and 10 mg/L. Taking a precautionary approach and applying an application factor of 1,000 to the lowest value gives a PNEC of 1 μg/L.

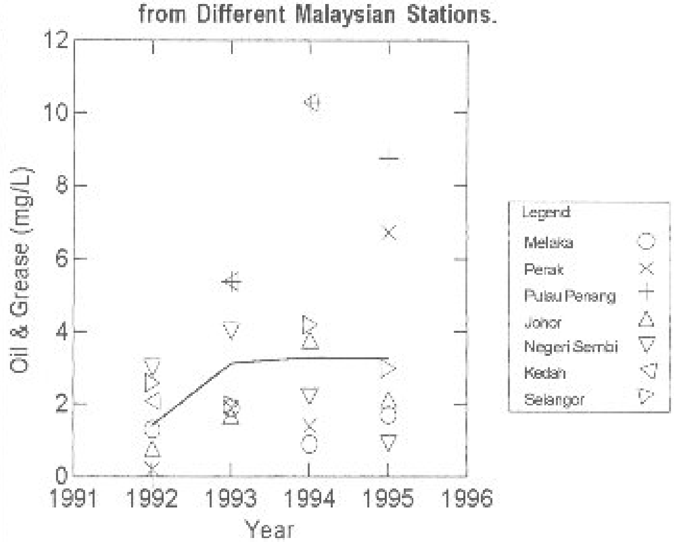
In the light of all this variability in PNECs, for the initial risk assessment a value of 1 μg/L was chosen for both oil and hydrocarbons and a value of 3 mg/kg for sediments. However, following *Appendix Document* II, it seems more appropriate for the Refined Risk Assessment to use a standard of 1 mg/L for total oil and grease with the lower standard being used only for hydrocarbons.

MECs

First of all, comparing the critical concentrations referred to above with all the reported data in the *Profile* indicates substantial pollution in terms of both oil and hydrocarbons. This appears to be true not only for sites specifically selected around offshore oil fields and refineries (*Profile* Tables 7-14 and 7-15), but also for coastal waters off the west coast of Peninsular Malaysia (*Profile* Table 7-13) and other selected sites in the Malacca Straits (*Profile* Table 7-16). From *Profile* Table 7-13, there would appear to be no obvious time trends in hydrocarbon pollution of water column and sediments between 1992 and 1995. Although average levels appeared lower for 1995 as compared to 1992 (mean water column 1995: 23.3 μg/L (±21.0=SD, n=11), omitting the outlier for Kukup; mean water column 1992: 60.7 ±108.9=SD, n=9; mean sediment 1995: 67.8 ±64.98=SD, n=12; mean sediment 1992: 139.0 ±80.29=SD, n=9), there was considerable variation in the data and differences were not significant ($P>0.05$ for all comparisons). Dividing these average MECs by the critical water (1 μg/L) and sediment (3 μg/g) concentrations, respectively, gives an RQ of 23 for both water and sediment (using average MECs for 1995). Also, it is noted that in *Profile* chap. 7 hydrocarbon concentrations for the Riau Archipelago have been reported to reach as high as 1,000 or even 11,500 μg/L. The maximum RQs for the Riau Archipelago would thus be greater than 1,000.

A further consideration of the risk assessment was made using the most up-to-date data summarized in Table 37. Applying the oil and grease standard of 1 mg/L to the means gives a range of RQs from 0.0032 to 3.8. However, inspection of the ranges indicates that the RQ can rise appreciably above one. Figure 11 plots

Figure 11. Time Trends in Oil and Grease Concentrations in Water from Different Malaysian Stations.



Data are from *Appendix Document* I-M. The solid line is a locally-weighted scatterplot smooth (LOWESS) and is a robust indicator of trends.

Table 37. Water Column Oil and Grease for Different Areas of the Malacca Straits.

Location & Year	Water Column Oil & Grease (mg/L) (mean ± range)	*Appendix Document*
Malacca Straits coastal waters & estuaries along Malaysian side, 1990s	0.68 (0.1-2.4)	II
Straits of Johor, 1990s	1.07 (0.3-3.5)	II
Malaysian States along Malacca Straits, 1992-1995	3.28 (ND-10.3)	I-M
Siak River, Indonesia, 1995	(0.6-25.2)	I-I
Siak Estuary, Indonesia, 1995	(0.5-9.6)	I-I
Straits of Singapore, 1996	0.0032 (0.0025-0.0043)	I-S

MECs for different Malaysian States between 1992 and 1995 (from *Appendix Document* I-M). This further illustrates the extent of both temporal and spatial variability. Robust trend analysis, shown by the solid line in Figure 11, suggests a possible increase in time in average MEC between 1992 and 1993 and a leveling off after 1993. Further consideration of this variability will be given in the analysis of uncertainty below.

From Table 37, it is interesting to note that the concentrations within the Siak River are generally much greater than those in the estuary itself, and this strongly suggests that the land-based sources are important contributors to contamination within the Straits.

The most detailed analysis in terms of oil constituents is given in *Appendix Document* II in terms of total oil and grease and petrogenic hydrocarbons. Here, data were also given on sediment concentrations, as well as water column concentrations. This survey included the Strait of Johor, as well as the Malacca Straits, and in all cases the former were more contaminated than the latter. However, correlation analysis indicated that there was little or no correlation between water column and sediment contamination or (at least for the Malacca Straits) between total oil and grease and petrogenic hydrocarbons (Table 38). The two measures of sediment petrogenic hydrocarbon contamination did correlate significantly, but there was still a considerable amount of scatter in the relationships. Mean concentrations for petrogenic hydrocarbons from this survey are summarized in Table 39.

Applying the most stringent standard of 1 µg/L to the mean water column MECs gives RQs all in excess of one. But there was considerable variation indicated by the ranges, and less stringent standards could have been applied. Similarly, for sediments applying the most stringent standard of 3 mg/kg to sediment MECs gives RQs equal to or greater than one. Again, the ranges suggest considerable variation and a less stringent standard could have been applied. These issues will be considered further in the uncertainty analysis below.

Uncertainty analysis

RQs based upon means generally give cause for concern but this approach hides a considerable amount of apparently unsystematic uncertainty between sites and through time. Such variability is not unexpected for these kinds of data given that they can be influenced by local events (such as the disposal of oil from a passing vessel shortly before a sample was taken) and/or the complexity of dispersal of oil through water currents. On the presumption that variability in the data sets reflects this variability in some average kind of way,

Table 38. Spearman Rank Correlation Coefficients for Different Measures of Hydrocarbon Contamination in the Straits of Malacca and the Straits of Johor.

Correlation	Malacca Straits n, r_s, p	Straits of Johor n, r_s, p
Crysene: Water vs. Sediment	62, 0.012, >0.05	21, 0.201, >0.05
Seligi Crude: Water vs. Sediment	62, 0.015, >0.05	21, 0.305, >0.05
Water: Oil & Grease vs. Crysene	79, 0.073, >0.05	31, 0.454, <0.05
Water: Oil & Grease vs. Seligi Crude	79,-0.087, >0.05	31, 0.522, <0.01
Sediment Crysene vs. Seligi Crude	62, 0.408, <0.01	21, 0.696, <0.01

Data are from *Appendix Document* II.
n=sample size; r_s=Spearman rank correlation coefficient; p=probability level

Table 39. Petrogenic Hydrocarbons for the Malacca Straits and Straits of Johor for Water Column and Sediment.

Sample Type	Average Concentration (Range); Crysene Equivalents	Average Concentration (Range) Seligi Crude Oil
Malacca Straits, water	1.53 (0.1-7.0)	30.90 (1.5-186.4)
Johor Straits, water	27.70 (0-135)	449.10 (5-2795)
Malacca Straits, sediment	3.04 (0.05-9.51)	69.67 (5.48-703.89)
Johor Straits, sediment	8.41 (0.71-36.7)	112.01 (14.26-182.82)

Data are from *Appendix Document* II. Units are µg/L for water and mg/kg for sediment.

implications for risk assessment are now explored using Monte Carlo simulations. The output of these will be a probability statement that a sample taken from any point within the Straits will be likely to exceed one and hence lead to ecological harm.

This technique is first applied to the water column oil and grease MECs measured between 1992 and 1995 in selected Malaysian States (*Appendix Document* I-M). This incorporates the spatial and temporal variability illustrated in Figure 11. As in the previous section, a single standard of 1 mg/L is applied. The MECs were fit to a log-normal distribution (mean 3.28 mg/L; sd=2.85), and the distribution was sampled 1,000 times with replacement. On this basis, the probability of RQ being greater than one was 90%. It was noted that there was a possible difference between samples taken in 1992 and later years. The 1992 MECs may have lowered the estimates of RQ for other years. Hence, rerunning the analysis excluding the 1992 MECs, giving an adjusted mean MEC of 3.93 (sd=2.99) results in an adjusted probability of RQ being greater than one of 95%. Therefore, in both cases the likelihood of any water column sample causing harm is considerable.

The Monte Carlo uncertainty analysis was also applied to the data on petrogenic hydrocarbons for water and sediments in the Straits of Malacca and the Strait of Johor from Table 39. Here, again a log-normal distribution of MECs was presumed and for the water column account was taken of possible variability in standards by applying a range between the low of 1 µg/L and the Malaysian water quality criterion for class II waters of 40 µg/L. Similarly, for sediments a range of standards was considered from a low of 1 mg/kg and the FAO standard of 100 mg/kg. For both water and sediment, it was not possible to assign relative likelihood of applicability to different values for the standards and so as a default, it was presumed equal likelihood and fit to a uniform distribution for the Monte Carlo simulation. The results as probabilities of RQs being greater than one are summarized in Table 40. The likelihoods of RQ being greater than one were always greater for Johor Strait than for Malacca Straits. Concentrating on Malacca Straits, the probability of the water column RQ exceeding one was 60% for crude oil equivalents but 2% for crysene equivalents. Similarly, for sediment the probability was 51% in terms of crude oil equivalents and 2% in terms of crysene equivalents. So, on this basis, the risks of any sample of either water or sediment having an RQ of greater than one, and hence causing harm are considerably less than for total oil and grease. This is a significant point because notwithstanding the choice of standards, some would argue that the petrogenic components of oil are appreciably more important than total oil from an ecotoxicological point of view. It is also important to note that some would argue that sediment contamination, because of its persistence, should be taken more seriously than water column contamination. The results

Table 40. Assumptions and Results of Monte Carlo Simulation to Quantify the Probability that RQ Exceeds One for Different Measures of Petrogenic Hydrocarbon Contamination in Water and Sediment.

Type of MECs	MEC Avg (sd) (µg/L or mg/kg)	PNEC (µg/L or mg/kg)	Probability that RQ exceeds one (%)
Malacca water, Crysene units	1.53 (1.43)	1-40	2
Malacca water, Crude oil units	30.92 (29.23)	1-40	60
Johor water, Crysene units	27.71 (42.70)	1-40	46
Johor water, Crude oil units	449.13 (822.71)	1-40	94
Malacca sediment, Crysene units	3.04 (2.59)	1-100	2
Malacca sediment, Crude oil units	69.67 (101.45)	1-100	51
Johor sediment, Crysene units	8.41 (8.13)	1-100	8
Johor sediment, Crude oil units	112.01 (57.59)	1-100	86

here would suggest that risks from water column and sediment are in similar ranges. It is possible, though, that the water column risks could be reduced more easily and rapidly than the sediment risks.

Conclusions including consideration of human health risks

Notwithstanding the uncertainty referred to in the above section, the risk assessment would indicate a prima facie cause for concern. This would therefore argue for the development of more rigorous and thorough monitoring programs for the future that take appropriate account of the complexity of oil-related compounds.

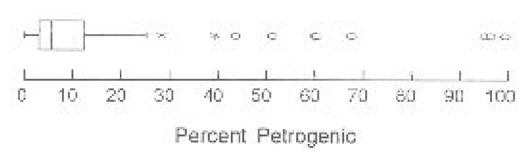
Many of the 'oily substances', their derivatives and breakdown products are hazardous to human health as carcinogens and general poisons. In addition, tainting of fish has been reported to occur at oil exposures as low as 10 µg/L and may occur within hours of exposure (GESAMP, 1993). This can have economic consequences by restricting areas or species suitable for commercial fishing. The high MECs for the Straits in general and particular places suggest possible risks to humans from dermal and dietary exposures. However, one is unable to carry out more detailed assessments of the kind applied for metals and pesticides until more is known about the identity, distributions and levels of specific oil compounds. Knowledge of the compositional characteristics of a hydrocarbon mixture can aid in identifying the kind and degree of hazard for human health (e.g., larger PAHs such as benzo(a)pyrene are known to be potent carcinogens), as well as in identifying the source of the contamination (e.g., it is possible to distinguish raw oil from used engine oil on the basis of the PAH content). Used engine oil comprises a primary component of the total hydrocarbons entering marine areas from ships, fishing boats and land-based discharges. So, if these sources of hydrocarbon contamination are high, combustion products, such as PAH, will comprise a relatively high fraction of the total amount of hydrocarbons in marine samples (GESAMP, 1993).

Given the uniformly higher oil contamination in the Strait of Johor compared to the Malacca Straits, it is important to note that *Appendix Document* I-S indicates that aquaculture production by Singapore is mainly in the Strait of Johor and that 3% of seafood consumed is from aquaculture species.

An analysis of relative contributions from land- and sea-based sources

A consideration of water column MECs recorded for oil and grease from different Malaysian States between 1993 and 1995 (*Appendix Document* I-M) suggests an average level of approximately 4 mg/L (Figure 11). Comparison of total oil and grease with petrogenic hydrocarbon concentration from the same sample (*Appendix Document* II) suggests that the fraction of total oil and grease composed of petrogenic hydrocarbons can vary widely from 0.15 to 99% (Figure 12). This is not surprising given that samples from different sites will have received contamination from widely different sources. The distribution summarized in Figure 12 is obviously skewed and the best descriptor is the median. This gives a value of 6% total oil and grease comprising petrogenic hydrocarbons.

Figure 12. Percent of Total Oil and Grease Composed of Petrogenic Hydrocarbons.



Data are from *Appendix Document* II and are for stations in the Malacca Straits and Johor Strait.

On average, 94% of oil and grease contamination within the Straits is likely to be from non-petrogenic sources, e.g., from natural sources and the palm oil industry. It is more likely that the non-petrogenic sources are more biodegradable than the petrogenic sources and are therefore of less concern. In what follows, the petrogenic oil components are the focus in considering relative contributions from different sources. In this analysis, it is presumed that 6% of the average total oil and grease contamination of 4 mg/L provides a reasonable average MEC for the petrogenic hydrocarbons in the waters of the Malacca Straits, i.e., this is equivalent to about 0.2 mg/L. In considering possible sources of this material, one can distinguish between land-based, sea-based and offshore activities.

Appendix Document II provides a very rough estimate of discharges per year from these sources that are of relevance for the Straits (Table 41). This totals approximately 8,480 tonnes/yr which, for convenience, is rounded up to 10,000 tonnes/yr. According to these figures, most of the contamination comes from land-based sources (approximately 60%) with the largest single source being used lubrication oils (i.e., from petrol stations and industrial machinery). The sea-based sources generate approximately 30% and here according to this analysis shipping accidents are a dominant source. Offshore activities were of lesser importance over the period of the analysis.

Taking the rounded total of 10,000 tonnes/yr and applying the one-compartment model gives a predicted environmental concentration from these sources in the water column of 100 μg/L (i.e., 50% of the total average MEC), and hence a substantial contribution to the overall contamination.

Another major potential source of contamination not apparently included in the analysis is due to operational losses from petrogenic industries and in particular refineries and production installations. One can calculate a contribution from refining by using data in *Profile* chap. 4 together with standard emission factors taken from the *EU Technical Guidance Notes for New and Existing Chemical Substances* (European Commission, 1996). The total volumes of oil involved in the refining process are quoted as follows: for Indonesia, 127,300 barrels per day (bpd); for Malaysia, about 300,000 bpd; for Singapore, 1.11×10^6 bpd. This is equivalent to a total of 5.1 million tonnes/day or about 2×10^{18} mg/yr. The *EU Guidance Notes* suggest that 0.05% of the total volume refined is likely to be lost as emission to water. Information in the literature suggests that 50% of that in the emission is likely to be lost to evaporation and decomposition within 7 days of release (Jensen, 1983), and it is presumed, conservatively, that 50% of the emission will persist in the water column. It is noted that similar factors have been used in *Appendix Document* XII. Applying these standard factors and using the one-compartment model, a prediction of an environmental concentration from refining losses is 0.05 mg/L. Given that most of the refining industry is concentrated in Singapore at the eastern end of the Straits, it becomes problematic as to how much of the emissions from there are likely to enter the Straits or pass into open waters. Understanding the distribution of oil pollution from these kinds of sources requires a detailed understanding of the hydrodynamics of the Straits.

Most of the oil production seems to be from Indonesia (*Profile* Table 4-24), with about 580 million barrels/yr and about 20 million barrels/yr from offshore activities–giving a total of about 600 million barrels/yr. These amounts are similar to those involved in refining (total of 550 million barrels/yr) and so applying a similar rationale would give approximately the same predicted environmental concentration of approximately 0.05 mg/L.

Table 41. Sources of Petrogenic Oil to the Malacca Straits as Estimated in *Appendix Document* II.

Source	Approximate Estimated Quantity (tonnes/yr)
Sea-based	
Oily waste water from vessels	400
Fishing boats	700
Shipping accidents	1,600
Offshore	
Exploration accidents	300
Exploration discharge	750
Land-based	
Crude oil terminals	400
Refineries deballasting	500
Used lubrication	2,500
Atmospheric	800
Natural	800
Total	8,480

These data apply largely to the first half of the 1990s.

The total contribution to contamination from refining and production is 100 μg/L. Adding this to the estimated inputs from other sources (Table 41) gives an overall total of 200 μg/L and hence accounts for the average MEC for the Straits. Table 42 summarizes relative contributions from different sources and indicates that by far the most important sources are land-based. Refineries and production installations and their contributions are likely to increase as both these industries expand.

Accidental discharges

According to the analysis in Table 42, the overall contribution of contamination from accidental spillages

Table 42. Estimated Relative Contributions of Oil Contamination to the Straits from Land-based and Sea-based Sources.

Source	Approximate Percentage Contribution to Average MEC of 200 µg/L	
Sea-based		16
Operational	6.5	
Accidents	9.5	
Land-based		78
Used Oil	15	
Refineries Deballasting	3	
Refineries Operations	25	
Oil Production	25	
Rest	10	
Offshore		6

to the average MEC for the Malacca Straits is probably less than 10%/yr. However, because the local impacts can be considerable and also because of the high profile nature of these events, risk assessment associated with these sources were considered in more detail. For mangroves in particular, it has been shown that oil from tanker spills can remain in mangrove sediment and cause ecological damage for many years after the spill (e.g., Burns et al., 1993).

In this analysis, it is important to distinguish between two kinds of risk: one is the probability of an accident occurring and the other is the probability of that accident causing harm to ecological systems. In a sense, these represent respectively probability of exposure to contamination from accident and probability of harmful effect and so at least in principle these two probabilities could be combined to give an overall assessment of risk arising from these sources. The key elements in each kind of risk can be summarized as follows:

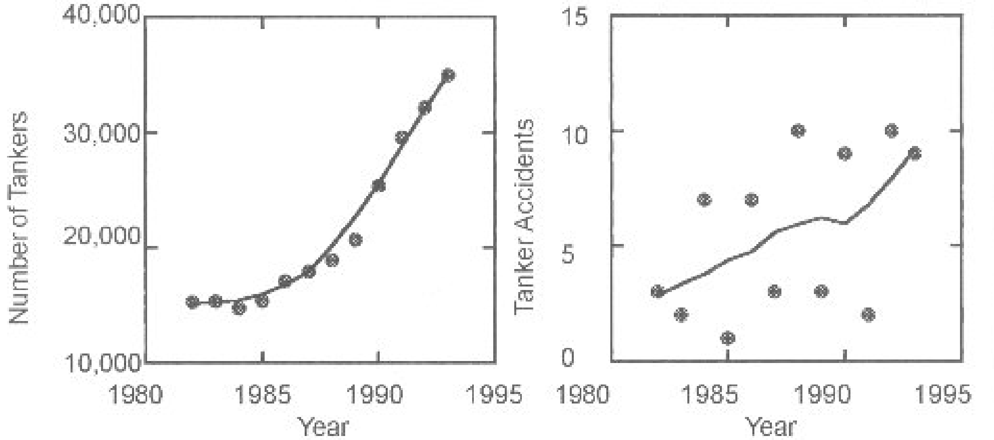
1) Probability of an accident occurring = f(number of vessels, age of vessels, type of vessel, time of day, weather conditions, width of shipping channel, presence of navigational aides, human error, etc.)

2) Probability of an accident causing ecological damage = f(amount of oil spilled, likelihood of oil reaching critical habitats, composition of the oil when it reaches its target, acute threshold for the target system, chronic threshold for the target system, exposure time, etc.)

In principle, it should be possible to calculate the probability of an accident occurring prospectively from knowledge of all the functional relationships summarized in the above equation. However, this kind of complete understanding of the situation is remote and hence probabilities derived from historical analyses of actual rates of accidents happening within the Straits are the only option.

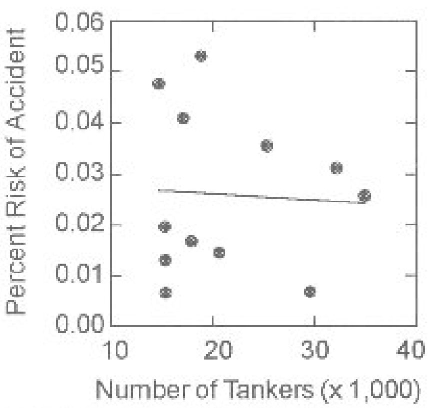
The amount of total shipping traffic through the Straits has increased over recent years (*Profile* Table 4-6). Presuming that a constant percentage of this (about 35%) comprises tankers (*Profile* Figure 4-3), the number of tankers passing per year from 1982 to 1993 has been calculated. Then using data in *Profile* Table 6-5 the relationship between tanker traffic and number of casualties is analyzed. Both the number of tankers and the number of tanker accidents increased with time in the period from 1982 to 1993 (Figure 13), but because of the understandable scatter in the data on accidents, there is only a weak correlation between tanker traffic and number of tanker accidents ($r=0.46$, $n=12$, $P=0.13$). In other words, 21% of the variability in tanker accidents can be explained from the variability in number of ships alone. Nevertheless, computing a risk of accident as number of tanker accidents/total number of tankers passing through the Straits and plotting this against number of tankers passing through the Straits (Figure 14), it is found that risk as a percentage of the number of tankers passing through the Straits per year is relatively constant at 0.029% (±0.03=95% CL). One could also have calculated these risk values from *Profile* Table 4-8, but this only includes data starting from 1987. However, the mean risk based on this smaller data set was 0.026% and hence close to the first estimate. These data represent total casualties and appear to take no account of size or seriousness. Between 1982 and 1993, there was an average of 4 casualties/yr (95% CL=1.9-6.1). From *Profile* Table 6-12, it is noted that between 1975 and 1993, there were 0.8 major oil spills/yr (>1,500 tonnes or 5,000 barrels) in the Straits. However, between 1982 and 1993, the frequency reduced to 0.4 major spills/yr. Taking this latter figure, this means that of the total casualties/yr over this period, approximately 0.4/4=0.10 or around 10% were major oil spills. This means that, fairly roughly, the likelihood of a major oil spill is 0.0029% of tankers passing through the Straits. These data are summarized in Table 43. Worldwide rates of serious casualties to oil/chemical tankers (6,000 gross tons and above) was

Figure 13. Total Number of Tankers and Number of Tanker Accidents in the Straits between 1982 and 1993.



Curves represent locally-weighted scatterplot smooths (LOWESS) of the data points and are robust indicators of trends.

Figure 14. Estimated Risk of Tanker Accidents as a Function of Total Number of Tankers Passing through the Straits.



As indicated by the non-significant linear regression through the data points, risk is independent of tanker number and averages approximately 0.029% (±0.015=SEM).

an average of around 2.6%/yr until 1980 and from 1980 to 1988 approximately 2.2% (IMO, 1989), and these are therefore considerably higher than the figures. For tankers of the size 100 to 5,999 gross tons, the average rate since 1978 is 1.26%/yr with virtually no changes over the years 1983-1989 (IMO, 1989). However, these data are not directly comparable with this calculation because they represent a percentage of tankers and not a percentage of tanker voyages.

Extrapolating from this time period, one can say that the absolute number of accidents is likely to increase with time as the level of traffic increases. On the other hand, this is problematic because it is clear that the level of accidents depends upon age and design of vessels, and these are likely to change as management practices change (*Profile* Figures 6-2 and 6-3). The likelihood of an accident at sea is essentially equivalent to the likelihood of the marine environment being exposed to oil pollution through this source.

SMEIS (*Appendix Document* XIV & XV) contains a subroutine for predicting the fate and effect of a spill from a vessel, given information on the rate of oil discharge, time and position of the spill and wind speed and direction. It estimates the impact of an oil spill on coastline habitats by calculating the pattern of spreading, losses due to evaporation, dissolution of the oil slick into water and hence the area coverage, surface concentration and time of impact along a stretch of shore along the coast. By combining this information with critical effect concentrations and information on the economic value of coastline habitats, it is possible to estimate the degree of ecological impact in monetary units. The critical effect concentrations are based upon long-term, chronic exposure and may therefore be conservative. On the other hand, with short-term, acute exposure, there can be post-exposure responses that would need to be taken into account, and in any event

Table 43. Average Risks of Accidents to Tanker Traffic in the Straits of Malacca, 1982-1993.

	Numbers/Year	Per Hundred Tankers
Risks of Total Casualties	4 (95% CL=1.9 - 6.1)	0.029
Risks of Major Spills	0.4	0.0029

there can be long-term exposure consequences from oil spills in particular habitats (e.g., mangroves; Burns et al., 1993). In developing the risk assessment models, these further complexities need to be incorporated, but that will require further research effort into the nature of the exposure and the ecological responses associated with particular spillage scenarios.

Risk management procedures, although well developed within the Straits (*Profile* chap. 6) are somewhat reactive, being intended to deal with accidents once they have occurred. However, SMEIS contains an inventory of oil combat equipment and its containment capacity. It also contains locations of different agencies equipped with oil spill facilities and their capacities for containment (*Appendix Document* XV). Using this and additional information, it is possible to envisage more proactive systems based upon risk assessment. For example, routing controls might be based upon an attempt to keep risky vessels (=f(size)(cargo)(age)) away from vulnerable sites (Marine Environmental High Risk Areas, sensu Donaldson, 1994). Using the oil spill trajectory model, it should be possible to explore the potential ecological damage from different oil spill scenarios associated with particular routing options.

It is noted in *Profile* chap. 7 that there are no published records of dangerous goods being shipped through the Straits. Thus, it is not possible to assess the potential significance of chemical spills due to shipping accidents.

Oil releases from other sources

For the marine environment in general, the primary inputs of oil are believed to occur from land-based sources, and in particular from refineries, municipal wastes and urban runoff (GESAMP, 1993). The refineries are addressed above, but there were no data in the *Profile* on the other sources. These, therefore, deserve further consideration.

COMPARATIVE RISK (AND UNCERTAINTY) ASSESSMENT

Introduction

Comparing risks across different contaminants on the basis of risk quotients has to be carried out with some caution for at least four reasons:

1) Relationships between the differences in threshold levels and exposures, and effects to both ecological systems and human health are unlikely to be either linear or independent of contaminant.

2) Relationships between the differences in threshold levels and exposures and ecological effects, even within families of contaminants are unlikely to be linear or standard from one ecological entity to another (i.e., the same RQ for different contaminants could have different meanings).

3) The RQ analyses are based on chronic responses and do not take account of the effect of episodic incidents at particular places.

4) The relative priority of effects and hence of the agents causing them is not just a matter for science, but also raises broader societal issues and perceptions. This is why comparative risk assessment often involves judgments from panels of experts and other stakeholders (SETAC, 1996).

Yet, the RQ analysis can provide some initial insights into relative risks. A procedure suggests itself from the conventions emerging from the EU new and existing substances assessment and management legislation (European Commission, 1996). Thus, if the RQs for any substance are less than 1, there are no immediate causes for concern. On the other hand, if RQs are greater than 1,000, immediate risk reduction measures are suggested. Between these extremes risks require more consideration, possibly with a more detailed risk assessment, and with increasing urgency as values increase in order-of-magnitude bands. On this basis, comparative risk profiles for the contaminants in terms of ecological entities and human health have been constructed in Tables 44 to 46.

In these tables, lines represent general conditions in the Straits with their extent reflecting either different values for different species of contaminants and/or uncertainties. Points represent highest values at particular places.

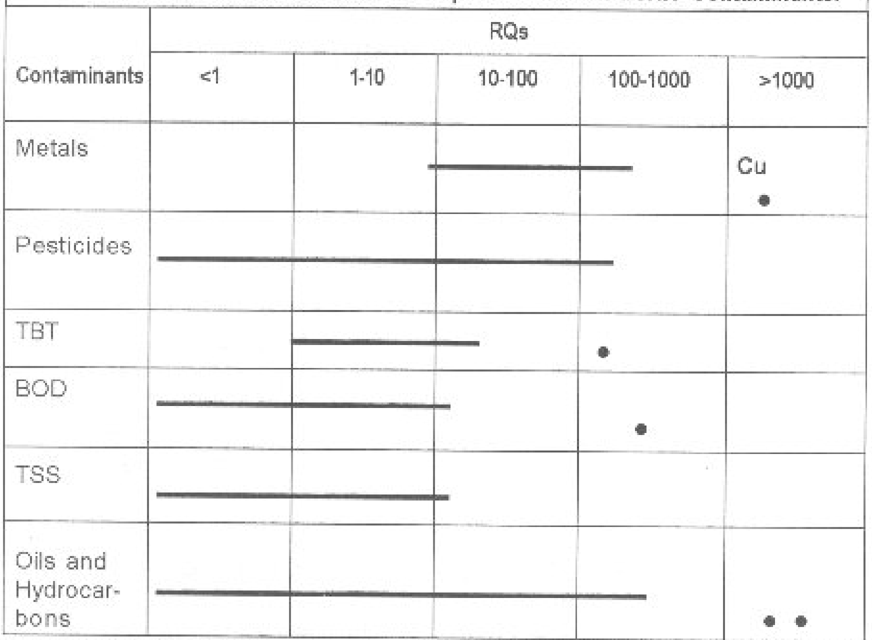
Comparative Assessment of Risks to the Environment from Water-borne and Sediment-borne Substances

These are summarized in Tables 44 and 45.

For water-borne substances, all categories have RQs that exceed 1 and so all generate cause for concern. Highest values in the Straits, indicating most pressing need for further attention are associated with some metals, pesticides and oils. Local "hot spots" inviting immediate action involve copper, TBT, BOD and oils.

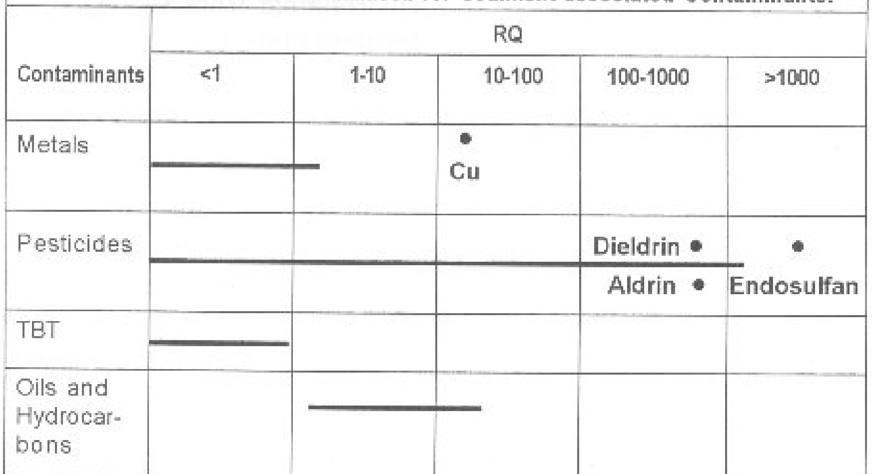
For sediment-borne substances, the situation is somewhat different with the pesticides and especially endosulfan in need of most immediate attention and action. As noted above, there is skepticism about the values for TBT (because biological effects likely to result from TBT exposure (i.e., imposex) have been recorded in the Straits) and it is believed that more monitoring is necessary here.

Table 44. Comparative Risk and Uncertainty Assessments for Ecological Entities Within the Straits of Malacca Exposed to Water-borne Contaminants.



Lines show the range of RQs determined in the prospective analysis and based on MECs given in the *Profile*. Selected compounds or sites having particularly high RQs are indicated with filled circles.

Table 45. Comparative Risk and Uncertainty Assessments for Ecological Entities Within the Straits of Malacca for Sediment-associated Contaminants.



Lines show the range of RQs determined in the prospective analysis and based on MECs given in the *Profile*. Selected compounds or sites having particularly high RQs are indicated with filled circles.

However, all these conclusions need to be judged against the background of considerable uncertainty. Uncertainty analyses have been performed taking into account variability in both MECs/PECs and standards to the extent possible. In general, these analyses demonstrate the need for better measured environmental concentrations, more robust exposure models and a more thorough understanding of critical effects levels for tropical marine ecosystems. Where the variability has been quantifiable, the probabilities have been calculated of RQs exceeding the critical value. In other circumstances, worst-case scenarios have been adopted.

Comparative Assessment of Risks to Human Health

These are summarized in Table 46.

All suggest that exposures for certain groups in certain places can give cause for concern and require further attention, with some urgency.

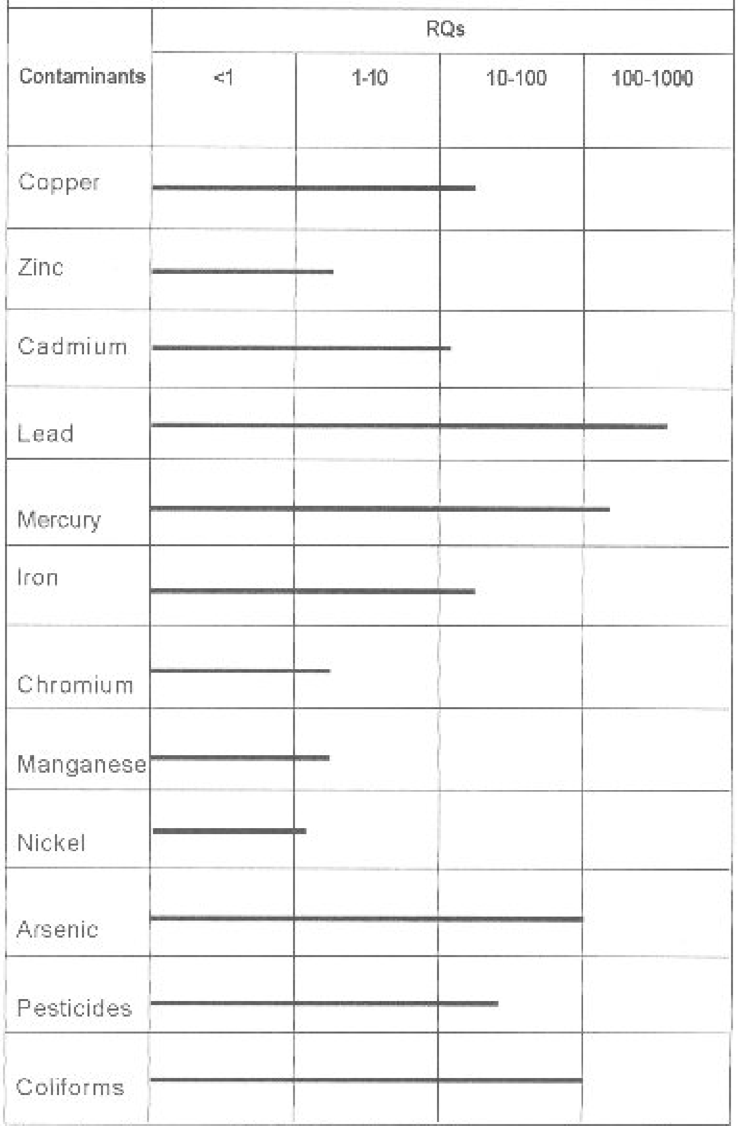
There are uncertainties with all of these conclusions in terms of both the extent and variability of contamination of the shellfish, and the extent to which they form part of the local diet–something that is likely to vary with both geography, age class and socioeconomic group. All these sources of uncertainty deserve urgent attention. As a first step, some of these sources of uncertainty were explored using Monte Carlo simulation. These analyses indicate the importance of seafood consumption rates and age-specific standards as important elements of uncertainty.

There were insufficient data to estimate the risks to human health from TBT and oily substances. However, the situation with regard to oil products is problematic. It is believed that the high levels in general, and the especially high levels in some places ought to give grounds for concern about possible implications for human health through both dermal and dietary exposure. Uncertainty here is largely due to ignorance about the kinds of substances that might be involved and hence their levels. Again, it is felt that this deserves urgent attention.

Comparative Assessment of Risks from Land- and Sea-based Sources

In the *Profile*, it is stated that 60-70% of the marine pollution is derived from the land. However, it is hard to be this precise because the relative contributions of each source can be measured in more than one way: as loadings; from the degree of contamination above background; in terms of risk contributions; in terms of relative effects on marine ecosystems.

Table 46. Comparative Risks and Uncertainty Assessments for Human Health from Various Contaminants.



Lines show the range of RQs determined in the prospective analysis and based on PECs.

Judged in terms of the comparative risk profiles, though, it is clear that land-based activities are the most important source of problems for the Straits' ecosystems. Thus, contaminants with (almost) exclusively land-based sources are the metals, pesticides and BOD. The only contaminant that is almost exclusively of sea-based origin is TBT. Oils and hydrocarbons might come from both sources but the analysis (see section on "Oil Releases from Refining and Production") suggests that the land-based sources are dominant. Much of the TSS comes from land-based activities, but most is associated with littoral activities, especially involving the clearance of mangroves.

Combined Effects of Multiple and Diverse Sources

In principle, it ought to be possible to consider combined effects on targets from diverse stressors by combining risk quotients, but this raises two not unrelated issues that have yet to be resolved. The first is concerned with how effects from different sources are likely to combine: additively or by more complex positive or negative synergism. The second, already alluded to above, is in terms of the form of the relationship between RQs and relevant effects in targets. On the presumption of additivity (for which there is growing evidence of generality; Doi, 1994) and linearity between RQs and effects, then combining RQs would simply involve summation:

$$\text{Combined RQ (Index of impact)} = RQx + RQy + RQz \text{ etc.}$$

And there are some examples of combined impact indices being calculated in this way; for example, for purposes of integrated pollution control in the UK (HMIP, 1993). Clearly, the assumption of simple combination (embodied in the plus signs of the above equation) is more likely to be reasonable for some groups of substances than others (Donkin, 1994), and within rather than between species of chemicals.

However, on the basis of the simplified assumptions, it is clear that the overall risks from combined sources, even within metals and pesticides could be very considerable. This is especially true of particular sites near to the outflows of contaminated rivers, industrial sites and harbors. Similarly, combined risks from various contaminants to the health of people living in various parts of the littoral States could be considerable; for example, from the combined effects of metals or pesticides in the diet.

It is also worth noting that, as for humans, other target species or habitats could be singled out for treatment if something about their specific sensitivities is known with regard to contaminants and their exposure. It would then be possible to calculate specific RQs, say for mangroves or fishes, and consider the separate and combined effects of different contaminants. In general, this kind of detail is not available, which is why RQs based on standard ecotoxicological tests were used and are presumed to reflect effects in ecological systems in general. However, more specific approaches are beginning to be made.

Implications for Risk Management

This comparative assessment suggests some immediate implications for risk management that are summarized for convenience.

1) Immediate action is suggested for the metals–but sources will have to be identified for action, and this is not immediately obvious for Hg.

2) TSS presents a problem for ecological systems and from the retrospective analysis, it would seem that the most obvious need for control here is with respect to mangrove clearance.

3) Oils and hydrocarbons are a cause for concern–and the analysis suggests a need for controls on refineries, which should include details of their location and operation (see SMEIS, *Appendix Document* XV). It should be possible to attain a better understanding of the relative importance of various sources of oil and hydrocarbon contamination from a more detailed analysis of the specific compounds present (GESAMP, 1993).

4) Pesticides are a worry because they may be persistent (e.g., in sediments) and are often applied by nonskilled operatives. Diffuse sources are always more difficult to manage than point sources.

5) Assessment of the potential health effects from contaminated shellfish and fish requires that more attention be given to diets and their control.

6) Coliform counts suggest that sewage pollution is a problem in many places. The ultimate solution will be in terms of the provision of better sewage treatment, but this is costly and long term. More immediate consideration ought to be given to bathing restrictions and controls on collection of marine life for food.

ASSESSING SOCIETAL RISKS

Societal risks refer to the likelihood of impairment of aspects of social welfare and the economy arising out of environmental conditions within the Straits.

The risk pathways in Figure 1 make it clear that deterioration in environmental conditions within the Straits can have important impacts on human health and wealth generation through, for example, impacts on fisheries, exploitation of other ecological resources, such as mangroves and tourism. The subsequent risk analyses, both retrospective and prospective, have demonstrated impairment of fisheries and mangroves and the possibility of serious risks to habitats and biodiversity in general from various contaminants, with likely implications for productivity and yield of the ecological resources and negative effects on tourism.

Even more generally, the likely risks to human health indicated by the prospective risk assessments from a number of contaminants can lead to a deterioration in the quality of lives of peoples of the Straits, loss of output and increasing pressures on the health care and welfare systems. Again, there are also serious implications for tourism.

All these risks have either been assessed in terms of population density and species diversity measures or as risk quotients that are presumed to relate to likelihoods of impacts on these. To gauge the seriousness of each and their relative importance in societal terms, it will be necessary to translate them into units that reflect societal impacts. These are usually taken to be monetary units. Thus,

Societal risk = f (likely loss or impairment of entity)(economic value)

where 'value' is not intended as an 'absolute', but as a measure of societal needs and preferences in a situation where resources are limited. It is usually judged by willingness to pay for the entity at risk in real or imaginary market places.

On this basis, values have been put on human lives (Quasim, 1988) and good health (Krupnick and Cropper, 1989), and on species, such as fish and lumber, that are traded commercially (Pearce and Moran, 1994). It is more difficult to put values on ecosystems and biodiversity in general, but techniques are being developed to get indications of willingness to pay for these kinds of entities and their protection from 'game playing' techniques involving imaginary market places (Pearce and Moran, 1994).

Against all these negatives to the economy arising out of the deterioration in environmental conditions in the Straits has to be set the positive contribution to the economy of the actions and activities causing the environmental problems. These, like the negative effects, can be valued, but in fact by more classical economics, involving both producer and consumer surpluses and supply and demand considerations. Thus, in principle, it is possible to value both costs (to the environment) and benefits (to aspects of the economy), and to consider the balance between the two in coming to policy decisions (UK Govt./Industry Working Group, 1995). And indeed this can be carried out at a number of levels, for example:

1) Involving particular risk pathways - e.g., balancing the ecological value of banning a particular pesticide against the economic costs in terms of food production and employment;

2) Involving particular projects - e.g., balancing the ecological gains of not allowing the construction of a refinery or tourist development in a particular place against the cost to the local economy from lost employment and revenue.

3) Making national accounts reflect not only standard relationships between income and expenditure in GDP/GNP (*Profile* Table 7-1), but also the using up of man-made capital (i.e., that which is usually referred to as 'capital') in net accounts (NDP & NNP) and, more radically, the using up of natural capital (i.e., valued ecological resources) as 'green' national accounts (gNDP & gNNP). Various governments and international bodies are currently experimenting with these ideas and methodologies (Pearce, 1993).

It is also important to note that the central presumptions of the sustainable development approach are: (a) that these costs and benefits can be made explicit at all levels; and (b) that modes of economic development can be identified that maintain a balance between environmental costs and economic benefits over the long term.

There is not sufficient economics information in the *Profile* for these ideas to be developed further here, and it will also be obvious that many of them are not without controversy. Nevertheless, the following recommendations are made:

1) Careful consideration needs to be given by all the major players in the Straits, governments and agencies, to the issues asociated with societal risks.

2) At the heart of assessing societal risks is the need for appropriate valuations. Most of those that exist have been developed from North American and Western European perspectives. Their relevance for the Straits needs careful appraisal, and it is very likely that adjustments will be needed.

3) Models need to be developed that flesh out the economics links in the risk pathways indicated in Figure 1, and from which rigorous and transparent cost-benefit analyses can be carried out of the kinds suggested above.

4) Consideration should be given to the extent to which the using up of natural capital within the Straits can, and indeed should, be reflected in the national accounts of the littoral States.

A significant start has been made with the development of these approaches in the GEF/UNDP/IMO Regional Programme for the Prevention and Management of Marine Pollution in the East Asian Seas (see MPP-EAS, 1999b).

FORMULATION OF AN ACTION PLAN AND OTHER RECOMMENDATIONS

The purpose of the *Profile* has been to provide an inventory of resources in the Malacca Straits, with special reference to pollution risks and to identify information sources which could serve as a basis for the SMEIS (*Appendix Document* XV). Risk assessment takes the process from a state of the environment inventory to a more detailed analysis of pollution risks and suggests possible needs for management action.

The approach herein has been based upon risk assessment principles and practices, and in making recommendations, the presumption is made that further developments will be similarly risk based. As is made clear throughout the report, the key elements of such an approach are that targets (and hence endpoints) are clearly defined, that appropriate sensitivity thresholds are identified and defined, and that exposure scenarios are specified. Each of these issues is treated in turn, first for ecological systems and then for human health indicating where the priorities for action are believed to be. As in the body of the text, issues for societal risks are treated separately. Finally, although it has not been part of the remit to make recommendations on management matters, a number of management "signposts" have emerged and are summarized as well.

Need for Definition of Ecological Targets and Endpoints

Throughout the exercise, the points are made that defining targets for protection depends both on scientific and societal issues; i.e., in terms of what conditions should be protected to maintain vital resources (e.g., how much mangrove forest is needed to maintain a sustainable fishery?) and in terms of where societal preferences and priorities are (e.g., in terms of conserving particular species or habitats because of their appeal for tourism). These issues need consideration, discussion and debate within the context of the Straits.

Obvious targets for attention were identified in the retrospective risk assessment, but there it is noted that more attention needs to be given to how impacts are judged in terms of both quantitative and qualitative aspects of the targets. Views about decline in ecological resources were often based on anecdotal evidence, and causation was often attributed subjectively. In particular, species losses fall into this category.

All this leads to the obvious recommendation that a carefully designed and coordinated program of monitoring of ecological resources should be developed for the Straits. Variables for assessment need to be agreed upon and sampling programs should be clearly designed. Data from these programs need to be assessed and stored in a coordinated way.

In the context of endpoints it is worth reiterating that a wide range is possible, from ecosystem to molecular levels. It has become somewhat fashionable to use physiological and molecular ones (i.e., biomarkers) since these often improve sensitivity. However, from a risk assessment perspective, that in itself ought not to be a defining criterion. Relevance to targets is much more important and ought to be used as a critical test of appropriateness. Biomarkers might, nevertheless, be of considerable importance in exposure assessment.

Need for Definition of Thresholds (Standards & PNECs)

An important aspect of prospective risk assessment is the identification of appropriate and relevant standards and PNECs. It has been illustrated how choice of standard can importantly influence results and hence conclusions for metals and pesticides in water column and sediments. Most of these standards and PNECs are based upon literature information that is largely if not exclusively from temperate systems. These data need careful consideration with regard to their relevance for the Straits, and here due regard needs to be given to international and regional activities on the review of ecotoxicological information and methodology for application within the tropics (Peters et al., 1997). In any event, agreement of appropriate standards for the Straits needs attention and again ought to be coordinated so that in carrying out risk assessments all players are using the same standards as a basis, and that this is done transparently so that revisions in the light of developing insights are facilitated. The development of a register of agreed standards for the Straits is a possible solution that is revised and updated in a coordinated way on a regular basis.

Very generalized risk assessments have been conducted for ecological conditions within the Straits. But the same methodology could be applied to more specific targets, whether habitats or species, provided that appropriate PNECs can be defined. This will require ecotoxicological data that are more specifically related to the targets. These kinds of data are slowly being gathered (e.g., Peters et al., 1997) but more work will be required.

What has been said with regard to the ecotoxicological thresholds also applies to the partition coefficients used in this report in calculating the standards for sediments. Again, it was illustrated how uncertainty here can lead to considerable uncertainty in the risk assessment. These coefficients are sensitive to conditions and will need to be applied with care for tropical systems. Standardized guidance for application to the Straits would again be helpful. However, it would be more advantageous to derive sediment quality standards directly using ecotoxicological test systems that are appropriate for conditions within the Straits, and it is suggested that some attention be given to this by all the interested parties.

Need for More Confidence in MECs

The uncertainty analyses revealed much variability in MECs that leads to considerable uncertainty in both Straits and local risk assessments for all contaminants.

This was described as stochastic variability, but potentially, it derives from a number of sources: analytical procedures, design and implementation of sampling programs and natural variability. In measuring environmental concentrations of contaminants, it is important to use agreed state of the art techniques; to take, store and analyze samples in a standardized way according to good laboratory practice; and to collect, collate and store data in a way that is easily available to all interested parties. Once more, this hinges on agreement between major players in the Straits with regard to standardization and the sharing of effort and information. A register of agreed techniques, that can be updated, would again be helpful.

Need for Exposure Models

PECs, both for the Straits and for more local scenarios, are likely to be needed in prospective risk assessments to cross compare with MECs, for example to test their generality, and to make predictions about impacts in advance of release of a substance or a new development at whatever scale. Their use was illustrated with a simple one-compartment model of the Straits. This likely needs considerable revision.

Clearly, more realistic models are required, and these need considerable understanding of the hydrodynamics of the Straits as a whole and of particular parts of it. To quantify the impacts of land-based pollutant sources on the coastal resources of the Straits, numerical models have now been developed and incorporated into SMEIS (*Appendix Document* XV). The flow rate and concentration of different pollutants discharged by several rivers have been used as input for the modeling of pollutant dispersion. The model consists of first a momentum jet model for assessing initial entrainment and throw distance perpendicular to the coastline prior to long shore drift. In the long shore dispersion phase a simple 2-D Gaussian model is adopted. Based on established mixing parameters the pollutant centerline concentration and sideways distribution are assessed. For a given concentration of pollutant the coastline impact distance exceeding appropriate threshold values can be computed.

Likewise to assess the impacts on coastal resources of oil released after an accident at sea a subroutine has been incorporated into SMEIS (*Appendix Document* XV). This takes account of the quantity of oil released, its spreading, evaporation, and dissolution and from these predicts the length of coastal resource impacted. By taking into account the kind of resource that is exposed and appropriate ecotoxicological threshold values, the ecological impact can be assessed in terms of monetary losses.

Contamination and pollution from agriculture is a serious concern, and predicting environmental concentrations from this source will require the development of understanding and models concerning: practice, rainfall, soil properties, groundwater and river flows and a host of other features. This will be a considerable challenge but experience from it may well lead to pointers on management practices concerning, for example, where and when to spray to minimize impact.

Needs in Human Health Risk Assessment

In terms of human health, there are uncertainties with both threshold effect values and exposure information. As far as threshold effects values are concerned, for some contaminants there is a considerable amount of information covering a wide range of age classes. However, for many, the threshold values have only been established in an interim way and most of the information is on adults rather than younger age groups.

The most important sources of uncertainty, nevertheless, are for dietary composition and levels of contamination in different foods. The former requires the collection, collation and ready availability of information on average diets for different groups in different parts of the Straits. The latter requires a more extensive survey of dietary contamination, taking into account not only the average concentrations but also the likelihood of high doses in particular units of food leading to acute poisoning.

Need to Keep Under Review What Is Monitored

In carrying out this risk assessment, attention was restricted to substances listed within the *Profile* and *Appendix Documents*, while being aware that this is unlikely to represent the complete universe of substances likely to be of importance to both ecosystems and human health in the Straits. For example, the view is expressed that specific derivatives and breakdown products of oils will be important; pesticides other than organochlorines are likely to be of significance, etc. Yet it would be an impossibility, and a waste of valuable resources to consider all possible contaminants. In Box 3, an algorithm is suggested for narrowing down a priority list of substances for consideration within the Straits.

Needs for Societal Risk Assessment

Societal risks are interpreted as the likelihood of impairment of aspects of social welfare and the economy arising out of environmental conditions within the Straits. There are a number of ways that the economy can be impacted by deteriorating environmental conditions, but attention was drawn to ways that environmental protection measures can have negative effects on the economy at least in the short term. This leads to considerations of balancing benefits with costs and attempting to optimize these to achieve sustainable development. The key recommendations here are that appropriate and relevant valuations are developed, especially for human lives and ecological benefits and that these be internalized into both appropriate micro- and macroeconomic models (e.g., MPP-EAS, 1999b).

Summary of Major Areas for Further Action

From Tables 44, 45 and 46, and taking into account the data gaps identified in the sections "Need for Definition of Ecological Targets and Endpoints" to "Needs in Human Health Risk Assessment", the following recommendations are made for prioritizing the needs for giving further attention to:

Box 3. Identifying Possible Causes of Pollution in the Straits and Prioritizing Them.

1. Search international lists of hazardous substances.

2. Identify contenders for a Straits' priority list by considering if any substance from #1 is likely to arise from industrial activities in and around the Straits. **Most will be rejected as low or zero priority.**

3. Are those from #2 recorded within the Straits?
    - If **yes**: **proceed to initial risk assessment.**
    - If **no**: is this because there have been no attempts to monitor?
        - If no: **discard as low or zero priority.**
        - If **yes**: is the substance likely to be persistent?
            - If **no**: **discard as low priority.**
            - If **yes**: design monitoring program.
                - If **not detected: discard as low or zero priority.**
                - If **detected: proceed to initial risk assessment**.

4. From initial risk assessments decide on need for further action using criteria employed.

1) Water column impacts on ecological systems: metals, oils and hydrocarbons are of primary concern.

2) Sediment impacts on ecological systems: pesticides are of outstanding importance.

3) Human health impacts: metals, especially Pb, Hg and As, pesticides and coliforms are all of importance, but there is also ignorance surrounding oils and this gives cause for concern.

These priorities are, of course, expressed in terms of contaminants, but of ultimate concern is (are) their source(s), especially from a management point of view viz.:

1) The sources of metals, largely industrial presumably, need to be identified and their relative contributions to general and local conditions need to be assessed. For example, industrial outputs along the River Klang deserve attention, and the Port of Singapore is a particular concern.

2) The sources of TSS from the analysis in order of importance are: loadings associated with mangrove removal and deforestation > industrial activities > pig farming > domestic outputs > aquaculture. These priorities need further consideration on the basis of future anticipated trends in littoral deforestation, agricultural practices, population increases and the provision of more effective sewage treatment throughout the littoral States, and the development of aquaculture.

3) A major source of oils and hydrocarbons from this risk assessment is petroleum refining and this is likely to be of increasing significance if the industry expands. However, contamination from municipal wastes and urban runoff could be appreciable.

4) The sources of TBT are obvious.

5) The sources of pesticides are also obvious, but the challenges of carrying out more refined risk assessments here are likely to be considerable.

6) The sources of coliforms are also obvious.

Finally, it is important to remember that the ecological risk assessments are based on generalized RQ analyses. It may become necessary to make these more specific. Thus, the retrospective analyses indicated clear deterioration of some habitats:

1) mangroves and peat swamps
2) seagrass beds

and some species

1) commercial fisheries

These were largely ascribed to physical removal for biomass and to make way for other developments. However, the results from this risk assessment point to the possibility of chemical impacts–i.e., signaled by the generalized RQ analyses. To make these assessments more precise and to judge their importance relative to the other sources of deterioration, it will be necessary to carry out more focused and in-depth studies using effects thresholds and exposure scenarios more specifically related to the targets.

Possible Risk Management Actions

Priorities for risk management are signposted by both the retrospective and prospective assessments.

From the retrospective analysis, attention is drawn to the following needs:

1) The loss of mangroves, peat swamps and seagrass beds. Notwithstanding the need for further analysis here, it is clear that an agreed and coordinated approach to clearance would be helpful.

2) The declining fisheries. Again, it is clear that an agreed and coordinated approach to the rational implementation of controls on fishing intensity by using appropriate models to set levels and possibly quotas would be helpful.

3) Protection of other species will need to be considered in the light of more objective and systematic assessments of risks of extinction and the values put on them by public and legislators.

From the prospective analysis, the following needs urgent attention:

1) For ecological impact, RQs greater than 1,000 should invite immediate action. Tables 44 and 45 point to the problem of copper contamination in the Port of Singapore and of oils and hydrocarbons in, for example, the Siak Estuary, Riau; Rangsang Island, Sumatra; Port Klang, Malaysia; Johor Strait. TSS at Pantai Sungai Lurus in Johor and TBT at Port Klang are also contenders. Endosulfan in sediments also falls into this category.

2) For human health protection (Table 46), food contamination from metals deserves serious attention, but pesticides also require consideration. Here, monitoring for likely contamination should be more extensive, and restrictions on where food organisms are collected need to be contemplated. Similar immediate measures may need to be taken to guard against sewage pollution, with more long-term provision of improved sewage treatment in Malaysia and Indonesia.

All of the above bear primarily on the likelihood of risks arising out of chronic exposure. In addition, there will be certain situations, namely accidents, in which acute effects will be of primary importance. This has already been discussed in terms of accidental spills from oil tankers in the Straits. Management actions here are well developed but could be made more proactive by developing response strategies that formally incorporate information on, for example, the type and volume of cargo, age of the vessel, proximity of critical habitats, current regime, weather and possibly other factors. Management strategies could be designed to minimize the potential for contact between high-risk vessels and particularly vulnerable habitats.

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APPENDIX 1

Executive Summary of Initial Risk Assessment

The initial risk assessment of the Malacca Straits is focused on the effects of land-based and sea-based activities, and the contaminants deriving from them, on living and non-living resources of the Straits of Malacca, including ecological, human and societal components. The *Malacca Straits Environmental Profile* (1996) was the primary resource document for the initial risk assessment. The *Profile* was completed as part of the Malacca Straits Demonstration Project, a component of the GEF/UNDP/IMO Regional Programme for the Prevention and Management of Marine Pollution in the East Asian Seas. The *Profile* provides an inventory of natural resources in the Straits with special reference to pollution risks, compiles existing management policies, strategies, practices and initiatives taken by major players, and identifies pollution sources and activities.

In this report, environmental risk assessment is defined as follows: estimating the likelihood of harm being done to identified targets as a result of factors emanating from human activity, but reaching the targets through the environment. It therefore combines an understanding of the potential that factors have to cause harm (hazard) with an understanding of levels of exposure. Two kinds of (related) questions have been asked: what evidence is there for harm being done to appropriate targets in the Straits (referred to as retrospective assessment); and what problems might conditions known to exist, or that might exist in the future, cause for targets (prospective assessment)? Attempts are made to identify the most relevant endpoints for assessment, though very often it is necessary to use surrogates. These are referred to respectively as assessment and measurement endpoints.

The risk assessment approach implies that it is possible to use scientific techniques to specify likely consequences for targets of human influence. Usually the chemical quality of the environment is considered. Further, it is often presumed as part of this approach that there exist states of the environment arising out of human actions that are associated with low likelihoods of adverse effects, and that human activities can be managed to these levels, often without need for zero emission requirements. This therefore suggests an important distinction between contamination and pollution, which was followed during this project. As a further step, risk/benefit approaches, which are also alluded to in the context of societal risk assessment, recognize that environmental protection measures, while bringing benefits to the environment, can bring costs to the economy. The possibility of wanting to balance benefits and costs in pursuing policies of sustainable development is raised.

In the retrospective assessment, information given in the *Profile* on the state of habitats, biodiversity and human health has been systematically reviewed. Endpoints preferably should have been in terms of biomass density and dynamics, population density and dynamics and species diversity for the ecological systems, and morbidity and mortality for human health. However states were recorded more anecdotally, and causes of decline were attributed somewhat subjectively–with little information given on the state of health of peoples in the littoral areas of the Straits. As far as the ecological systems were concerned, there were clear indications of decline in mangroves, peat swamps, seagrass and commercially exploited fish species. Most of the declines were attributed, reasonably, to physical removal for biomass or to make way for other developments. Nevertheless, the prospective analysis suggests that deteriorating chemical conditions cannot be excluded as contributory causes and this needs further attention (below).

The development of a carefully designed and coordinated program of monitoring of major resources at risk is recommended, with variables for assessment being agreed and with the collection, storage and analysis of data being coordinated, possibly in a centralized way, between all contributory parties.

The aim of the prospective analysis was to estimate the probabilities with which activities, and the emanations from them, are likely to cause problems for human health and ecological systems in and around the Straits. More precise information is required on fates, exposures and effects than are available in the *Profile*. The risk quotient approach was generally used in which ratios of environmental concentrations (either measured or predicted) and effects threshold levels were compared. Quotients above one signal likely problems, whereas those below one signal that problems are unlikely–in neither case can a precise probability be assigned to these values. Measured environmental concentrations were gleaned from the *Profile*, as were some of the threshold values. Others were derived from the literature. Predicted environmental concentrations were obtained from a one-compartment model of the Straits, using simplified assumptions about volume and hydrodynamics.

Development of more sophisticated models for the Straits as a whole and its parts are recommended as a matter of urgency as models will be of considerable importance in carrying out more detailed risk assessments. In carrying out the initial prospective risk assessment using available inputs, a distinction in scale is made between assessments for the Straits as a whole and assessments for more local situations.

Risk quotients embody a considerable amount of uncertainty: in the environmental concentrations, as stochastic variability in measurements made at different places and at different times; in the predictions, based on the lack of information on the geometry and flow regimes of the Straits; and in the thresholds, from lack of relevance to the targets of interest. As far as the latter are concerned, at least for the ecological systems, a major source of uncertainty is the extent to which ecotoxicological data, derived from temperate systems, is applicable to tropical systems. It is recommended that this be reviewed, and that a register of standards be developed by the major players in the Straits to facilitate future risk assessments.

In principle, the extent to which uncertainty from all sources affects the confidence that can be put in the risk quotients can be precisely computed if the variances associated with each component can be defined. Uncertainty analyses require random sampling from these distributions and that, in turn, build up likely distributions of risk quotients. The *Profile* contains insufficient information on variances to facilitate such analyses. Instead examination of likely effects of variance in measured concentrations was conducted by inspecting the consequent effects on the distributions of risk quotients. These turned out generally to follow lognormal distributions, and so comparisons were made between log transformed quotients and zero (equivalent to a quotient of one) using standard statistical and graphical techniques. The variability of standards were taken into account by assessing the effects of differences between those quoted in the literature on quotients.

Another source of uncertainty here was with regard to standards for sediments. No standards were available, and so estimates were made from water column levels using partition coefficients for the appropriate substance. However, these are sensitive to environmental conditions and so need to be verified for appropriateness of use within the Straits.

In carrying out the initial prospective risk assessments the worst case scenarios tended to be taken: highest measured concentrations; lowest dilutions; and most sensitive threshold values. This is in the spirit of the precautionary principle and is justifiable, especially when uncertainty is considerable. It should be possible to become less cautious as understanding of the uncertainty improves and one moves from the semiquantitative approaches used in the initial risk and uncertainty assessments to more quantitative and probabilistic techniques.

It should also be said that in carrying out the initial prospective risk assessments, general ecological quality was considered, rather than the quality for specific habitats, ecosystems or species. In other words threshold effect values were used, derived from general ecotoxicological tests. To be more specific would require standards that were more precisely related to targets, and it is recommended that effort should be put into deriving such standards.

Using risk quotients, a comparative prospective assessment was carried out for both ecological and human health effects and the following recommendations are made on the basis of the assessment:

1. For water-column impacts on ecological systems, further attention is rated in decreasing order of importance as follows: a number of metals, but especially mercury as a general contaminant, and in certain locales, copper; total suspended solids; oils and hydrocarbons; tributyltin, in specific places; pesticides, biochemical oxygen demand and nutrients not being especially important.

2. For sediment impacts on ecological systems, pesticides are of outstanding importance, followed by oils and hydrocarbons, with metals in certain places.

3. For human health impacts, coliform pollution needs attention. Pesticide and metal contamination of food also needs consideration. A special effort is required to determine the extent to which shellfish and fish make up dietary intakes in the littoral States and the variability in contamination of shellfish and fish tissues from place to place and time to time. Dermal exposure to these contaminants is possible but not of great importance. Risk assessments could not be carried out on oils and hydrocarbons due to lack of detail on likely exposures to specific substances, either through the food chain or dermally. Many of the possible hydrocarbons involved are known to be hazardous. Given the high levels of contamination throughout the Straits, especially at particular locations, urgent monitoring and the development of more detailed risk assessments is recommended.

Finally it is important to remember that all risk assessments are based on information of exposure provided in the *Profile*. There are two caveats that are necessary here. First the data have been accepted as given without question. It is believed that these data, and those collected in future, ought to be judged more critically, and recommendations are made on how this might be done (e.g. in terms of good laboratory practice). Second, it is believed that the contaminants singled out for consideration do not represent the universe of contaminants that ought to be considered for risk assessment within the context of the Straits. It is suggested that a procedure for cost-effectively developing a more comprehensive priority list of contaminants for the Straits be implemented.

The recommendations listed above are expressed in terms of contaminants, but of ultimate concern, especially from a management point of view, are their sources.

1. The sources of metals are largely obvious, but mercury, that is a general cause for concern, needs more consideration.

2. The sources of total suspended solids in order of importance are estimated as–mangrove removal and land-based forestry, various industrial activities, pig farming, domestic outputs and aquaculture. A re-evaluation of these priorities is recommended in the light of more refined risk assessments and also in the light of future anticipated trends in such activities as littoral and land-based deforestation, agricultural practices, demographic changes, the provision of more extensive sewage treatment in Malaysia and Indonesia, and the development of the aquaculture sector. It is suggested that predictive risk assessment models can be employed to provide decision-makers with information on the likely impacts of such developments.

3. For oil contamination, it is estimated that a major source is the refining industry, and this is likely to be of increasing importance as the industry expands. A simple model has been formulated whereby the impacts can be predicted and managed. The model needs further development to make it more realistic. It is noted that other land-based activities could be making significant contributions, and experience elsewhere suggests that municipal wastes and land runoff are of importance. Further consideration is needed for the Straits.

4. The sources of tributyltin and of the coliform contamination are obvious.

5. The sources of pesticides are also obvious. However, making predictive models of impacts on the Straits from agricultural activities is likely to be challenging, because of the diffuse nature of the sources, and the complexities of environment between sources of pesticide application and the Straits as the ultimate sink. Advice is given on how such models might be structured and a suggestion is made that attention be given to model construction.

All the prospective risk assessments described above are concerned with long-term and chronic events. Accidents at sea can lead to acute exposures followed by chronic contamination. As far as oil tankers are concerned, it is not believed that accidental events are important with regard to contamination in the Straits as a whole. Similarly, normal vessel operations that lead to oil losses are not believed to have a Straits-wide impact. However, local exposures can be of considerable importance ecologically. The *Profile* indicates that the provisions for management of oil spills are good within the Straits. Nevertheless, it is believed that more needs to be done in terms of accident avoidance and recommendations are made in terms of route management, based upon an assessment of risks associated with carriers (e.g. on the basis of their size and age) and the proximity of marine environment high risk areas.

Most of the analyses have been carried out on substances in isolation. Nevertheless the possibilities of combined effects from multiple and diverse sources are considerable for all targets. The concern is discussed in the report, but the point is made that research will be needed to resolve issues on the nature of the synergy, if any, between toxicants, and on the relationship between the risk quotients and the extent of adverse ecological effects. This is not just a matter for the Straits but for the ecotoxicological community in general. On the basis of simplified assumptions, it is clear that the overall risk from combined sources, even within the metals and pesticides, could be very considerable for some if not all ecological systems. Similarly the combined risk from various contaminants to the health of people living in various parts of the littoral States could be considerable, e.g. from the combined effects of metals and pesticides in the shellfish diet.

No recommendations are made on risk management, but a number of areas are signposted by the analyses. The retrospective assessment indicated a need for coordinated action to control of the loss of mangroves, peat swamps and seagrass beds and of the fisheries. The prospective analyses suggested a need to address pollution in local areas from metals, total suspended solids, oils and hydrocarbons, tributyltin, and pesticides (especially endosulfan) in sediments. For human health protection, shellfish contamination from both pesticides and metals is highlighted, with the possibility of restricting permissible shellfishing areas. Similar immediate measures may need to be taken to guard against sewage pollution, with the more long-term provision of better treatment facilities being important.

Societal risks have been interpreted as the likelihood of impairment of aspects of social welfare and the economy arising out of environmental conditions within the Straits. How the economy can be adversely affected by deteriorating environmental conditions is illustrated, but attention is also drawn to ways that environmental protection measures can have negative effects on the economy, at least in the short term. The development of cost/benefit models are proposed and it is indicated how these might be used to effect at both microeconomic levels, e.g. in deciding if a particular development project should proceed, and at the macroeconomic level, e.g., in adjusting national accounts to incorporate depletions in ecological capital. All depends crucially on the development of appropriate valuations. A start has been made on valuation of resources, but is largely focused on the situation in North America and Europe. It is recommended that serious consideration be given by major players within the Straits to the relevance of environmental economics and to the development of valuations, e.g. for human health and ecological entities, that are locally appropriate.

Finally a recurrent theme running through the recommendations is the importance of agreed and co-ordinated action between all major players, e.g. in terms of monitoring; methodologies; standards; the collection, collation and storage of data; the prioritization of further programs of risk assessment and the necessary supporting research; and the prioritization of management programs. The development of a suitable forum whereby this can be facilitated will be of considerable importance.

Summary of Recommendations

1. *Development of a coordinated monitoring program for natural resources.* For the sake of a more precise inventory of the quantity and quality of natural resources, and targets and appropriate endpoints for measurement and assessment, a coordinated monitoring program to collect and collate data is recommended. The program could involve the regular updating of the *Environmental Profile* and should build on existing monitoring programs [section 11.2].

2. *Development of a coordinated monitoring program for chemical contaminants.* It is recommended that a more systematic and coordinated approach to measuring environmental concentrations of chemicals, especially with regard to sediments, be developed. Building on existing programs, this should involve the standardization of analytical procedures, the development of effective sampling programs, and the analysis of the temporal and spatial scale of variability in measured concentrations [section 11.4].

3. *Development of exposure models.* The further development of exposure models for the Straits as a whole and for particular portions and/or sources is recommended. The model for the Straits would need to incorporate an understanding of its hydrodynamics. Models for sources of contaminants would need to incorporate knowledge of production volumes, release scenarios and the distribution and decomposition within the environment [sections 8.1 & 11.5].

4. *Harmonization of critical effect concentrations.* The harmonization of critical effect concentrations (i.e., water and sediment quality standards; predicted no-effect concentrations) of likely relevance in the Straits [section 11.3] is recommended. This could be a component of the Regional Programme's proposed Environmental Atlas for the Straits.

5. *Detailed risk assessment of metals in water.* For water column contaminants it is recommended that a more refined risk assessment be carried out for metals to clarify sources and exposure levels. The initial risk assessment indicates that mercury and copper should be given special attention. Only by identifying sources will it be possible to develop effective management programs [section 8.2].

6. *Determination of sources and critical effect levels for suspended solids.* It is recommended that a more detailed analysis be made of the relative contributions of suspended solids to verify the conclusions that various land use practices and mangrove forest clearance activities are most significant, with industrial and domestic inputs of lesser significance. The critical threshold for suspended solids effects needs more consideration [section 8.6].

7. *Oil and hydrocarbon contamination.* It is recommended that more needs to be known about the composition of oil and hydrocarbon contamination in the Straits. Only with this understanding can effects levels and likely impacts be assessed. The conclusion, based on available data and using default assumptions, that land-based sources of oil and hydrocarbon contamination are more important than normal operational activities associated with shipping needs to be verified [section 8.8].

8. *A risk-based strategy for avoiding ecological impacts from oil spills.* Accidental oil spills can have both short-term and long-term consequences with considerable economic and ecological importance in local areas. A risk-based strategy that attempts to minimize exposure of sensitive habitats [section 8.8.6] is recommended for consideration.

9. *Ecological risks from nutrients.* The initial risk assessment was not able to consider nutrients in detail. Possible signs of eutrophication indicate that a more detailed risk assessment is required [section 8.5].

10. *More refined risk assessment of pesticides in sediments.* The initial risk assessment was based upon theoretical considerations of the manner in

which chemicals partition between sediment and water. These considerations need verification. There was considerable variability in measured sediment concentrations, and more extensive and carefully designed sampling programs [section 8.3] are needed.

11. *Human health effects from marine contaminants.* It is recommended that a more detailed assessment be carried out on the likely impacts of marine contaminants, especially metals, pesticides, coliforms and possibly hydrocarbons, on human health. More needs to be known about the diets of people living in the littoral States and the extent and variability of shellfish contamination [sections 8.2, 8.3, 8.7, & 8.8].

12. *Further consideration of causes of decline in commercial fisheries.* It is recommended that a review of methods for assessing fish stocks and sustainable yields be completed. The initial risk assessment indicates that overfishing is the main cause of declining stocks, but the effects of other human activities involving habitat destruction and pollution cannot be excluded. There is an urgent need for developing rational methods for managing fishing intensity [section 7.3.2].

13. *Benefit-cost analysis as an integral part of the risk management program.* It is recommended that in all instances where risk management is seen to be necessary, the costs and benefits of alternative management strategies should be considered. In assessing benefits from improved management, attention should be given to the damages avoided to resources and uses of the Straits. The initial risk assessment suggests that mangroves and fisheries should be given particular attention. The full quantification of costs and benefits and hence the development and application of appropriate valuation techniques are an appropriate objective. Although this is unlikely to be possible in many circumstances, qualitative comparisons based on listings of costs and benefits can provide helpful insights [section 10].

APPENDIX 2

Extra Documentation Provided by GEF/UNDP/IMO Regional Programme for the Prevention and Management of Marine Pollution in the East Asian Seas to supplement information in the *Profile*.

Listed in text as *Appendix Document*:

I. Preliminary information on Singapore (I-S), Malaysia (I-M), and Indonesia (I-I), supplied prior to Johor Bahru Workshop November 1997, contained in references IV to IX.

II. Abdullah, A.R. and C.W. Wang, Editors. 1996. Oil and Grease Pollution in the Malaysian Marine Environment. Institute of Advanced Studies, University of Malaya, Kuala Lumpur, Malaysia.

III. Low, K.S. and Y.N. Phua. 1998. Impact Assessment of Coastal Water Pollution in the Straits of Malacca. Draft Report.

IV. Database and GIS of Marine and Coastal Resources in West Coast Peninsular Malaysia (February 1998). Draft Report

V. Database and GIS on Marine Pollution Sources in West Coast Peninsular Malaysia (February 1998). Draft Report.

VI. Database and GIS of Marine and Coastal Resources in East Coast Sumatra (January 1998). Draft Report.

VII. Database and GIS on Marine Pollution Sources on the East Coast of Sumatra. Draft Report.

VIII. Database and GIS on Marine and Coastal Resources on Singapore, February 1998. Final Report.

IX. Database and GIS on Marine Pollution Sources, Singapore, February 1998. Final Report.

X. A 20-year forecast model for pollutant loading into the Malacca Straits from Singapore, February 1998.

XI. Report on the Training Workshop on Land-based Oil Discharges to Coastal Waters: Ecological Consequences and Management Aspects, UNEP/EAS, August 1996.

XII. ASEAN Marine Environmental Management: Quality Criteria and Monitoring for Aquatic Life and Human Health Protection, 1997.

XIII. Draft Description of the Oil Spill Trajectory and Pollutant Fate Models for the Malacca Straits, 1998.

XIV. Chong, V.C., A. Sasekumar, S.M. Phang and M.N. Jaafar. 1998. Marine and Coastal Resource Mapping for the Straits of Malacca. GEF/UNDP/IMO Regional Programme for Prevention and Management of Marine Pollution in the East Asian Seas.

XV. Low K.S., Y.N. Phua, C.K. Lee, M.N. Jaafar and W.A. Wan Zahari. 1998. Development of Regional Database System and GIS for the Straits of Malacca. GEF/UNDP/IMO Regional Programme for Prevention and Management of Marine Pollution in the East Asian Seas. Draft Report.

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