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Malacca Straits: Initial Risk Assessment

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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 the Marine Pollution in the East Asian Seas is to support the efforts of the 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, 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 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 environmental 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 the 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

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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 Community
EU - European Union
GESAMP - Group of Experts on the Scientific Aspects of Marine Pollution
GM - geometric mean
HELCOM - Helsinki Commission
MAFF - Ministry of Agriculture, Fisheries and Food
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
RQ - risk quotient: MEC(or PEC)/STD (or Threshold)
STD - standard
STW - sewage treatment works
TBT - tributyltin
TDI - tolerable daily intake
TSS - total suspended solids
US FDA - United States Food and Drug Administration

FOREWORD

The Straits of Malacca is one of the busiest shipping lanes in the world, serving as a natural shipping route linking the Indian Ocean via the Andaman Sea with the South China Sea, to the Pacific Ocean. But the Straits is more than a shipping lane. It is a unique tropical estuarine environment, rich in renewable and non-renewable natural resources. For local communities along its coastlines, the Straits supports a large marine fishery as well as numerous aquaculture and mariculture activities. At the national level, the people and governments of Indonesia, Malaysia and Singapore rely on the natural resources provided by the Straits as a source of food, raw materials, employment, trade and commerce, transportation, recreation and social well-being.

As a consequence of rapid economic growth and development in the region, the marine and coastal resources and environment of the Straits are under increasing stress. Deterioration of the marine environment is occurring as a result of expanding human activities on land and at sea, with physical destruction of habitats, coastal erosion, overfishing and pollution inputs from both land and sea-based sources being some of the more obvious threats to the sustainable development of this most treasured resource. The littoral States well recognize the exisitng and potential threats to the marine and coastal environment of the Straits and, over the years, have embarked on individual and joint programs and activities to prevent, mitigate and respond to such threats. The Government of Japan has also made significant contributions to support maritime safety and marine pollution prevention programs in the Straits. However, increasing pressure on the resources of the Straits implies that the balance between human activities and a healthy environment is changing. The need to strengthen existing capacities and programs to meet these challenges at a regional level is apparent.

The initial risk assessment of the Malacca Straits was implemented as part of the GEF/UNDP/IMO Regional Programme on the Prevention and Management of Marine Pollution in the East Asian Seas. The assessment was primarily based upon information which had been collected from the littoral States and compiled into a Programme document, entitled the Malacca Straits Environmental Profile (October 1996). The purpose of this initial risk assessement is two-fold:

1. to systematically identify the main elements of risk, namely the different categories of targets or endpoints in the Straits, any significant adverse changes to those targets, possible causes of such changes and possible consequences of the changes for the Straits' ecosystem, human welfare and society as a whole;

2. to estimate the likelihood of adverse effects on appropriate targets as a result of environmental conditions that exist, or might exist in the future, within the Straits.

An obvious shortcoming of the initial risk assessment is that it is based mainly upon information available in the *Malacca Straits Environmental Profile*. Therefore, any data gaps, limitations and uncertainties that occur in the *Profile* impact on the comprehensiveness and reliability of the risk assessment. To address this concern, the report includes detailed descriptions of the rationale and procedures that have been employed during the assessment, in order to ensure transparency and to provide opportunity for replication as more information becomes available. In additon, efforts have been made to explain the weaknesses of existing information and analyses, and to suggest ways for improving the situation through short-term and longer-term actions.

The ultimate objective of the Regional Programme is to develop and verify a risk assessment-risk management framework that can be transferred to other subregional sea areas in the East Asian Seas region, based upon the Malacca Straits experience. This initial risk assessment report is an important step in the development of the framework, but by no means is it a final product. Over the next few months, the Programme will be working with the littoral States of the Straits to refine the assessment component of the framework, and to operationalize aspects of the framework dealing with marine pollution preventative options and benefit-cost appraisal. By the end of the Programme, it is envisaged that a practical risk assessment-risk management framework will have been developed and proven, and ready for application in sea areas throughout the region.

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The initial risk assessment of the Malacca Straits was completed under the direction and coordination of Mr. S. Adrian Ross, Senior Programme Officer, Regional Programme for the Prevention and Management of Marine Pollution in the East Asian Seas, Manila, Philippines.

Professor Peter Calow, The University of Sheffield, Sheffield, U.K., and Dr. Valery E. Forbes, Roskilde University, Roskilde, Denmark prepared the risk analysis and compiled the final report.

Views and comments on the draft report were made by representatives of the three littoral States at a consultative meeting in Cebu, Philipppines in March 1997. The following individuals are recognized for their inputs:

Indonesia:	Capt. Henky Lumentah, Directorate General of Sea Communications
	Mr. Alam Syah Mapparessa, BAPEDAL
Malaysia:	Hajah Rosnani Ibarahim, Department of Environment
	Mr. Hasbulah Zakaria, Department of Environment
Singapore:	Mr. Bin Chee Kwan, Ministry of the Environment
	Capt. Chan Heng Lum, Maritime Port Authority of Singapore
	Ms. Lee Mui Ngah, Maritime Port Authority of Singapore

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EXECUTIVE SUMMARY

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. *Cost-benefit 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].

1. TERMS OF REFERENCE

1.1 Introduction

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 contaminative 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. A number of states 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 (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.

Here we have been given the *Malacca Straits Environmental Profile* (GEF/UNDP/IMO, 1996), as a 'snapshot' of the state of the environment of the Straits and invited to carry out an initial risk assessment on this basis. 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, to which we shall allude in this report, recognize that environmental protection measures while bringing benefits to the environment can bring costs to the economy, and present the possibility of taking this into account in establishing appropriate control measures.

These risk and risk /benefit approaches also require more precision in defining what it is we want to assess risk 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 we want and are prepared to protect (Forbes and Forbes 1994).

In the report that follows, Sections 2 to 6 provide background, with Section 4 translating the principles that are discussed above into the approach that we have used in the initial assessment of information in the Malacca Straits Environmental Profile. The initial risk assessments are carried out in Sections 7 to 10. Conclusions and recommendations are given in Section 11. In carrying out this initial exercise we have had 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. We have attempted to make all our analyses as systematic and as transparent as possible to facilitate future amendments. The rest of this section summarizes the formal objectives and terms of reference.

1.2 Objective

1.2.1 Formal Specifications

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.

1.2.2 Work Programme Outline

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

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

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

   iii. spatial and temporal scales of the assessment;

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

   v. combined effects of multiple and diverse stresses on the ecology of the Straits; and

   vi. 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 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 programme for the subregion.

2. SOURCES OF INFORMATION

The material upon which we have based this initial risk assessment has been largely from the Malacca Straits Environmental Profile, hereafter referred to as the *Profile*. When we refer to material from this source we shall specify it routinely as *Profile* Table, *Profile* Figure and *Profile* p. In general we shall not cite source references again when they are listed in the *Profile*. Other literature that we used in the initial risk assessment has been cited in the normal way.

As specified in the Terms of Reference we have relied on the data, and to some extent standards, compiled within the *Profile*. The presumption has therefore been that the data that we have 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?). Future risk assessments will need to address these issues more systematically and rigorously. Because of our own methodology, this initial risk assessment is very dependent upon the reliability of standards, and we shall return to this again below.

3. THE STRAITS

This section provides a brief description of the geography, ecology and socioeconomic aspects of the Straits as background to the initial 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 c. 300 vessels passing through per day (*Profile* Table 4-6). At the same time their 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* p. 370).

Many of these natural biological resources are exploited along both coasts of the Straits. Chief amongst 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 (c. 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.

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 (c. 11 mil. along the west, c. 10 mil. along the east and c. 3 mil. 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, 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 p.a. is quoted for the Indonesian side; *Profile* Table 2-3). 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.

We calculate the total volume of the Straits as c. 10^{12} m^3 (see section 8.1), 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* p. 14) 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* p. 10) 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 ppt (*Profile* p. 355) 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., a greater fraction of dissolved Cd is in the bioavailable free ion form at lower salinity, and hence the toxicity of Cd increases with decreasing salinity; 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 the water body 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 we define further these complex interactions in section 6 before proceeding to the detailed risk assessments in sections 7 to 10.

4. 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.

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 assessment - Comparison of the intrinsic ability of a substance to cause harm (i.e., to have adverse effects for humans or the environment) with its expected environmental concentration, often a comparison of PEC and PNEC. Sometimes referred to as risk assessment.

Hazard identification - Identification of the adverse effects which 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 the environment 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 which 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).

Risk characterization - The step in the risk assessment process where the results of the exposure assessment (e.g., PEC, daily intake) and the effects assessment (e.g., PNEC, NOAEL) are compared. If possible, an uncertainty analysis is carried out, which, if it results in a quantifiable overall uncertainty, produces an estimation of the risk.

Risk classification - The weighting of risks in order to decide whether risk reduction is required. It includes the study of risk perception and the balancing of perceived risks and perceived benefits.

N.B. We have tended to use the general term "risk assessment" throughout where others might have used "characterization" or "classification". It will be clear, however, what we intend from the text.

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

1. What evidence is there for 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.

2. What problems might conditions that exist now or in the future cause for human health, habitats and species? This is known as the **Prospective Approach**.

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).

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.

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. & Streatfeild, C. 1995. *DIY Environmental Risk Profile.* Sheffield Regional Green Business Club; Sheffield, UK - ISBN 0 9524211 00).

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 - but precise probabilities cannot be specified. We use this approach in our initial risk assessment of the Malacca Straits.

3. PROBABILITIES

If we 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. & Baldwin, L.A. 1993. *Performing Ecological Risk Assessments.* Lewis Publishers, Chelsea, MI, USA.

Assessments of risk provide a likelihood of occurrence of some harm on the basis of an understanding of all the variables involved. Rarely, however, do we have 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).

5. THE APPROACH HERE

Based largely on available information in the *Profile*, we shall carry out both retrospective and prospective analyses 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 conditions known to exist in the Straits (or expected in the future) cause for human health and ecological systems?

The main categories of targets in these contexts will be:

a. Human health
b. Habitats (i.e., mangroves, peat swamps, seagrass beds, coral reefs, soft-bottom habitats).
c. Species (i.e., commercial and non-commercial marine species).

We shall identify appropriate assessment and measurement endpoints. Our general philosophy will be to identify systematically each of the two main elements of risk: potential harm (H) and likelihood of exposure to potential (E), such that Risk = f(H)(E), where f means "function of".

We shall distinguish between different scales of risk. In particular, we will consider risks to the Straits as a whole (in which we will treat the Straits as a single compartment and estimate a single average exposure concentration for the entire Straits) and, for selected contaminants, risks to local areas within the Straits (in which, for example, we will estimate local exposure concentrations in the vicinity of individual rivers).

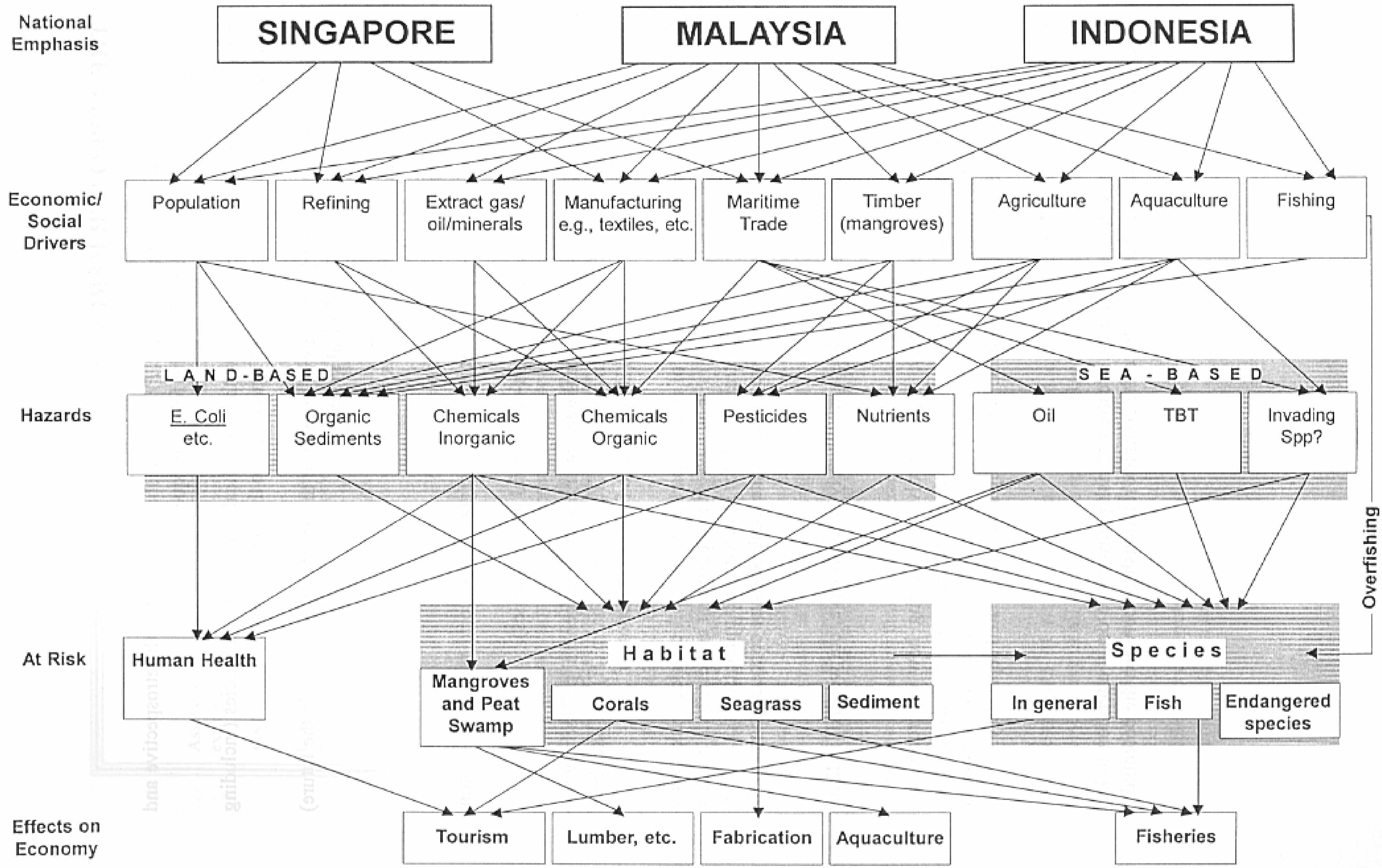
We shall carry out uncertainty assessments. These will largely be qualitative, but we shall indicate how they might be made more quantitative.

We shall deal with societal risks separately. These involve considering how environmental degradation, and its control, impact the economy. This involves 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. Hence they should be treated separately.

6. RISK PATHWAYS

For perspective we begin with a qualitative indication of risk pathways to draw attention to key issues. The risk pathways in Figure 1 illustrate the complex relationships between the potential

Figure 1. Risk pathways illustrating relationships between potential causes of human health and environmental problems and their consequences in the Straits.



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 amongst the littoral states. The consequences of pollution will have knock-on effects to the economy, again not evenly amongst 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 judgements 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.

7. RETROSPECTIVE ASSESSMENT

7.1 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;
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, we have applied this discipline to information provided in the *Profile* for habitats and species. For each we summarize: evidence for decline; attributed causes; likely consequences. We emphasize 'attributed' because in this section we have presented those views on causation that are expressed in the *Profile*. With retrospective assessment, views about causation are always based more on expert judgement 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; see Box 2), but due to the lack of detail in the *Profile* these have not been applied here. We, therefore, treat the attributed causes 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 we have maintained a distinction between habitats and biodiversity for convenience. Clearly they are closely interrelated.

7.2 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. nos. of patches) and areal. 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 we consider each of the main habitats listed in the *Profile* systematically in terms of: evidence for decline; attributed causes; 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.

Table 1. Summary of retrospective analysis of declines in key habitats for the Straits as a whole. 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; our judgements 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.

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	**	**

7.2.1 Mangroves

7.2.1.1 Evidence for decline

Presently, the area occupied by mangrove forests along the Straits is 386,100 hectares in Indonesia (=77.5% of total mangrove area), 111,409 hectares in Malaysia (=22.4% of total mangrove area), and 600 hectares 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* p.125). For Malaysia 17% of the mangrove area was lost between 1965 and 1985 (*Profile* p.128); another estimate of loss is around 35% (*Profile* p. 369). For Singapore the percentage of the coastline occupied by mangroves has declined from 10-13% (*Profile* p. 129), to

0.1-1% by the most recent estimates. Approximately 81% of the area occupied by mangrove forests in Singapore was lost during the last 20 years (*Profile* p. 31).

Several species of mangrove are listed in *Profile* Table 3-18, but there is little information (and no quantitative estimates) in the *Profile* of any changes in quality other than mention that many of the remaining mangroves in Singapore are "fragmented or degraded to a certain extent" (*Profile* p. 369).

7.2.1.2 Attributed causes

On the Indonesian side of the Straits, the main cause of mangrove decline is clearance for brackish water ponds (tambaks) (*Profile* p.108 &126). On the Malaysian side and for Singapore, the main cause of mangrove loss has been clearance for development (*Profile* p. 31 & 128). Other major causes of mangrove loss are: overexploitation (i.e., of the wood resources); 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.

7.2.1.3 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* Fig 3.20 & Fig 3.21); 4) a loss of critical habitat for endangered species and for conserving biodiversity (*Profile* Table 2-12 & p. 128); 5) possibly economic consequences for the timber industry (though 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 programmes to date appear to be relatively ineffective (e.g., compared to Malaysia; *Profile* p. 147).

7.2.2 Peat Swamp Forests

7.2.2.1 Evidence for decline

The area occupied by peat swamps in Sumatra has decreased from an original area of 7.3 - 9.7 million hectares (c. 25% of all tropical peat lands; *Profile* p. 30) to approximately 3.6 million hectares (or approximately a 50% reduction; *Profile* p. 126). 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* p. 30). There were no exact figures provided in the *Profile* of the area occupied by peat swamps in Singapore (but see *Profile* Figure 2-9). However since they typically occur in connection with mangroves, this area is relatively small.

7.2.2.2 Attributed causes

At least in Indonesia, losses of peat swamps have occurred largely from logging (there are many commercially valuable tree species), transmigration programmes and land conversion to rice, palm and coconut (*Profile* p. 126).

7.2.2.3 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* p. 37).

7.2.3 Coral Reefs

7.2.3.1 Evidence for decline

Coral reefs are found in smaller patches than in other areas in the ASEAN Region (*Profile* p.37). 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 (*Profile* p.39). For Malaysian coral reefs in the Straits, c. 26 -46% were rated as "fair" and none as "poor" in terms of percent live coral cover (*Profile* Table 2-8). Singapore's coral reefs were described as "amongst the most stressed in Asia" (*Profile* p. 40). Species diversities in the coral reefs were not reported.

7.2.3.2 Attributed causes

For Malaysia, sedimentation was rated as the greatest cause of coral reef decline, followed by fishing 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* p. 40). Pollution from metals, oil spills, and pesticides can have adverse effects on corals (Peters, et al. 1997).

7.2.3.3 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).

7.2.4 Seagrass Beds

7.2.4.1 Evidence for decline

The distribution of seagrass beds along the Malacca Straits is reported as less extensive than in other ASEAN waters (*Profile* p. 43), but no quantitative data on areal coverage (or losses thereof) were provided in the *Profile*. Of a worldwide total of c. 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 90s (*Profile* p. 370).

7.2.4.2 Attributed causes

The primary cause of seagrass decline appears to be from intentional destruction for conversion to coastal aquaculture (*Profile* p. 46). 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* p. 45 - 46 & Table 3-26). Pollution from metals, oil spills, and pesticides can have adverse effects on seagrass meadows (Peters, et al. 1997).

7.2.4.3 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, loss of nursery grounds for fishes including some of commercial importance.

7.2.5 Soft Bottom Habitats

7.2.5.1 Evidence for decline

The area of the Straits covered by sandy and muddy bottoms is reported as "extensive" (*Profile* p. 49), 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*. 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).

7.2.5.2 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* p. 122); and 2) contamination of sediments from pollutants from various sources (see Prospective Analysis).

7.2.5.3 Consequences

A decline in the quality of soft-bottom habitats has had economic consequences in terms of contamination of marine food products (*Profile* p. 371) and may be a contributing factor in the observed decline in catch-per-unit-effort (CPUE) for demersal fisheries (*Profile* p. 371). 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 (Clark 1992; HELCOM 1990) 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*.

7.3 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, though 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.

7.3.1 Non-Commercial Species

7.3.1.1 Evidence for decline

For **non-commercial species** there are few quantitative data in the *Profile*, 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 abundant pre-1950s are now cited as either rare or absent. Sting rays have decreased and dugongs that were once common in the Straits are now scarce (*Profile* p.371 - 372). 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* p.42 - 43). 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* p. 372).

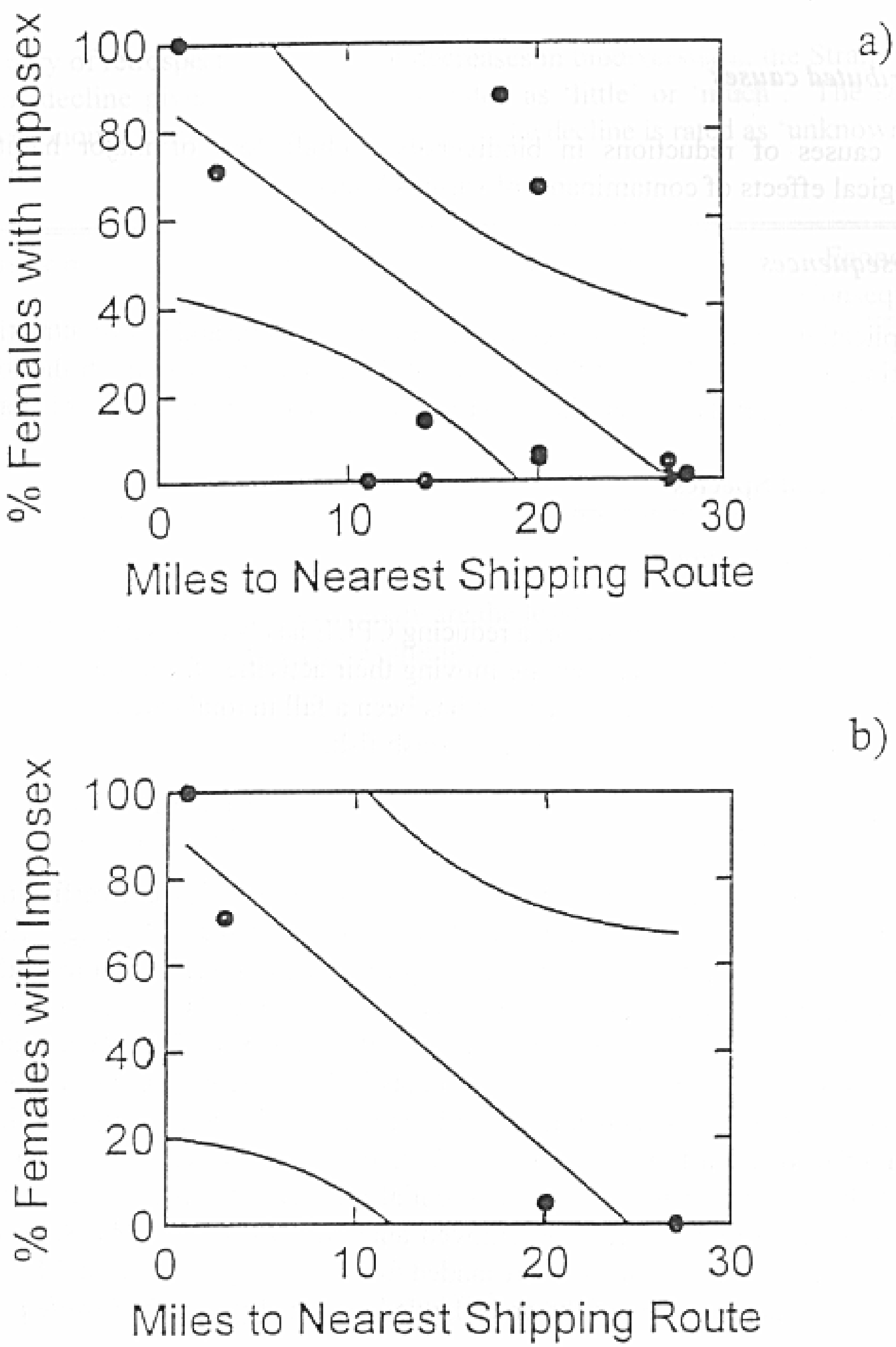


Figure 2. Incidence of imposex in relation to distance from the nearest shipping channel. a) for all species shown in *Profile* Table 7-26; Spearman Rank Correlation Coefficient (r_s) $=0.641$, $P<0.05$, df=12; Least-Squares Regression: $y=86.9 - 3.2x$; $n=14$; $r^2=0.525$; $p=0.003$. b) for *Murex occa*; $r_s \approx 1$, $P<<0.05$, df=2; $y=91.8 - 3.75x$; $n=4$, $r^2=0.937$; $p=0.032$.

7.3.1.2 Attributed causes

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

7.3.1.3 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).

7.3.2 Commercial Species

7.3.2.1 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* p.121). 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.

7.3.2.2 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* p.121), 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* p. 49). White prawn, tiger prawn and greasyback prawns are fished. Prawn fisheries exceed potential yield (*Profile* p.136). Approximately 14,000 tonnes of sergestid shrimp (*Acete*) are removed annually (*Profile* p.136). 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" (*Profile* p. 136).

Table 2. Summary of retrospective analysis of decreases in biodiversity in the Straits. 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'.

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

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).

7.3.2.3 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).

7.4 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.

7.4.1 Evidence for Decline

The *Profile* contains no specific quantitative information on the levels of morbidity or mortality for human populations in the littoral states.

7.5 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).

8. PROSPECTIVE ANALYSIS

8.1 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.

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 we shall often take quoted outflows of rivers as sources and estimate PECs on the following basis. To predict environmental concentrations in the Straits ($PEC_{Straits}$) from information on land-derived contaminant loadings we have used an extremely simple, one-compartment model of the system 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. We have calculated total volume of the Straits by assuming a symmetrical geometrical configuration with triangular cross-section, having 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 (=1853 m/hour) specified on *Profile* p.16 we calculate a flushing rate of once per 500 hours, or approximately once per month, but conservatively have rounded down to 10 times per year. In consequence we have taken conservative estimates 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. We have 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, or 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 we shall either use information given in the *Profile* (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). 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). We use this approach in considering the likelihood of accidents to shipping within the Straits. Note that the terms in [ ] are the same as before.

For humans our 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 we shall sometimes use this approach.

As already noted (see Retrospective Analysis, above) 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. We therefore rely on general assessments of likelihood of effects from concentrations of likely effectors. These are either derived from standards (STDs) often taken from the *Profile* 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 judgements 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 tolerable (daily) intakes (TDIs). The basis of assessment factors used here is discussed in ECETOC (1995).

For the simplified ecological risk assessment we compare MECs and/or PECs with PNECs and/or STDs. We use ratios known as risk quotients (RQs), where

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

For human health risk assessment:

$$RQ = \frac{(MEL \ or \ PEL)}{(PNEL \ or \ TDI)} \qquad (2)$$

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 in the *Profile* are generally not detailed or 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 we 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). For certain types of composite quotients, less computer-intensive techniques may be appropriate (e.g., Slob 1994).

Otherwise we examine variability in RQs and use this to make judgements 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 we need 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 we presumed that they were from unpolluted areas and calculated a measure of contamination (defined by GESAMP as "raised levels of the chemical compared with natural background levels", Olsgard and Gray 1995) 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.

8.2 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 initial risk assessment we assume that these levels are representative of the Straits in general. Hence we refer to them 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. The standards are summarized in Table 3.

8.2.1 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), we also calculated a measure of contamination 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.

Results of the risk quotient analysis 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.

We compared the relative risks of heavy metal pollution 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 section 8.2.11), as indicated by the fact that their 95% confidence limits did not overlap with zero. From these RQ_{Local}s copper consistently is associated with the highest environmental risks with RQs always exceeding 100 and in the Port of Singapore greater than 1000 (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. The RQs for Mn and Fe could not be determined due to a lack of available standards for these metals. No single site ranked consistently highest or lowest for all of the measured metals, although Sentosa & Marina Bay appeared overall to be the least polluted of all of the sites.

8.2.1.1 Uncertainty analysis

There are two levels of uncertainty in these data: a) based on standards; and b) based on variability in MECs. Here we examine variability in MECs and return to an analysis of variability in the standards below.

Variability across samples in Table 5 gives some impression of the variability 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 we simply look at the distributions of RQs relative to the critical value. Presuming a lognormal distribution of measured concentrations and hence RQs (which appears plausible on both theoretical grounds (Slob 1994) and from inspection of the raw data), we have transformed the data and present them as mean log RQs ± 95% confidence limits in Figure 3. 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 RQs for all metals are greater than zero.

8.2.1.2 PECs

Using data from *Profile* Table 5-8 for the west coast of Peninsular Malaysia for river inputs of heavy metals, and using our one-compartment model, we calculate a $PEC_{Straits}$ (in μg/L) of 7 x 10^{-4} for Hg (all coming from Kelang), 404 x 10^{-4} for Pb (most coming from Melaka), 2.6 for Cu (all from Kelang), and 334 x 10^{-4} for Zn (all from Kelang). 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 Kelang and Melaka. Using average outflow data from *Profile* Table 2-3, we calculate a total annual outflow of approximately 10^{12} L/yr. Using this figure and applying metal loadings from *Profile* Table 5-11, gives the following PEC_{Locals} for the outflow from the river Kelang: 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 river Kelang. *Profile* Table 5-9 shows that the Kelang 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. We note also that a 1996 survey (*Profile* p. 359) "showed that, in general, heavy metal contamination in coastal waters was limited to certain areas close to industrial sites and estuaries". We were unable to find flow rates for the river Melaka and therefore have not been able to carry out similar calculations for that system.

Table 3. Environmental standards for heavy metals from various sources. Values are given in μg/L. 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. NS indicates that no standard was provided for this metal.

Metal	Table 7-3 Std.	Table 7-4 Std.	Danish (EU) Std.
Pb	100	0.2 (10)	5.6
Hg	1	0.1 (0.3)	0.3 (SW)
Cu	100	1 (60)	2.9
Cd	10	0.2 (10)	2.5 (SW)
As	100	NS	NS
Cr	500	NS	1
Ni	NS	NS	8.3
Zn	NS	NS	86
Mn	NS	NS	NS
Fe	NS	NS	NS
Sn	NS	NS	NS

Table 4. Metal concentrations in water. 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. 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 ✔. BG=background values obtained from Laane (1992). BQ=highest mean value/background value.

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

Table 5. RQ_{Local}s for metal concentrations in water. Local RQs are based upon MECs given in *Profile* Table 7-7 (presuming data are in mg/L) 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.

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
Johore Straits East	169	793	24	71
Johore 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

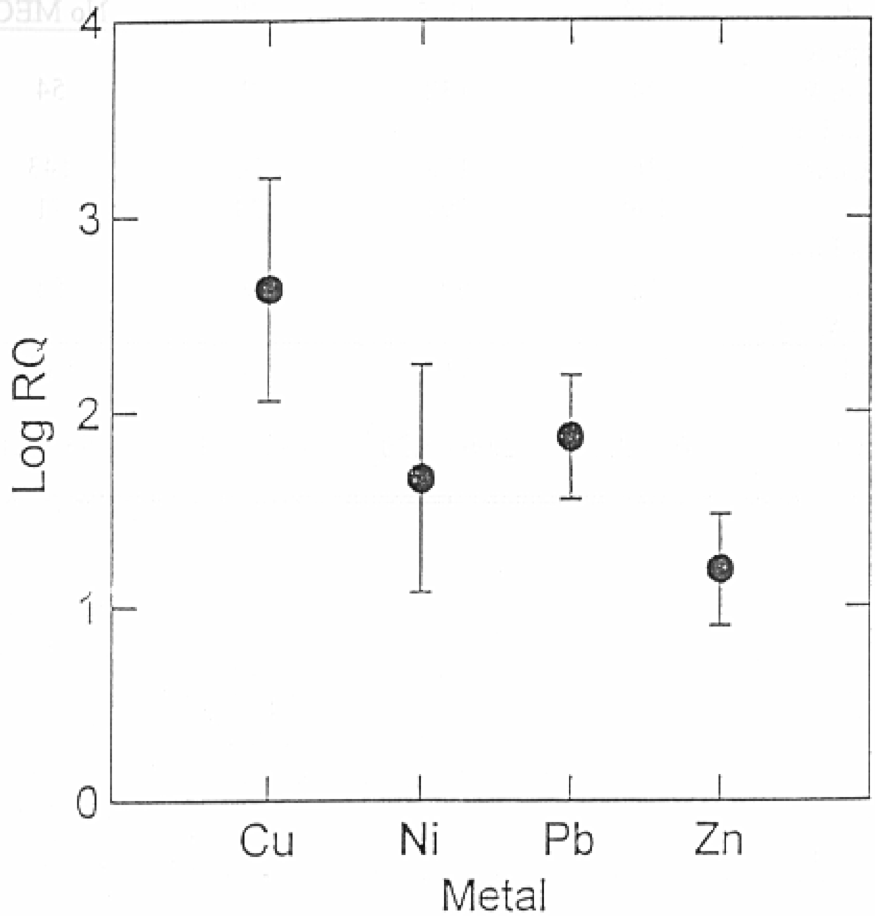


Figure 3. Average risk quotients (transformed to logarithms) ± 95% confidence limits for metals in water. All metals exceed the critical value of log RQ=0.

8.2.2 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). To date, there are no generally accepted sediment quality standards, and instead we based our RQ estimates on water quality standards following Van Der Kooij et al. (1991). Briefly, threshold water concentrations were converted to critical sediment concentrations using the formula:

$$C_{Sed} = \frac{(C_W \times K_{SW})}{r} \quad (3)$$

where C_w is the threshold concentration of metal in water (mg/L; here the water STD)
C_{sed} is the critical concentration of metal in sediment (mg/kg)
K_{sw} is the solids-water partition coefficient (L/kg)
r is 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 we used to estimate C_{sed} values from the standards given in our Table 3.

Table 7 calculates 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. It will be clear that there is a considerable amount of variability in these data and we shall return to this below. For the purposes of this initial risk assessment, however, it is convenient to focus 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 these were always greater than one except when *Profile* Table 7-3 standards were used. Lead and cadmium had the lowest RQs (using Danish (DK) STDs), 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 & 7-8) and different metals were included the two types of analysis (e.g., *Profile* Table 7-7 omits Cr, As and

Table 6. Critical sediment concentrations (mg/kg) based on water quality criteria for selected heavy metals. 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 (above).

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

Table 7. Metals in sediments. The highest MEC values from *Profile* Tables 7-5, 7-6, and 7-8 are shown for each metal. RQs are calculated as the highest MEC 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).

I. Using Median K_{sw}	Highest MEC	RQ (Table 7-3)	RQ (Table 7-4)	RQ (Danish Standard)
Pb	134	0.003	1.6	0.06
Hg	No MEC			
Cu	229	0.07	6.9	2.4
Cd	5.5	0.006	0.3	0.03
As	26	0.04		
Cr	69	0.0007		0.36
Ni	89			2.1
Zn	428			0.07
II. Using Lowest K_{sw}	**Highest MEC**	**RQ (Table 7-3)**	**RQ (Table 7-4)**	**RQ (Danish Standard)**
Pb	134	0.005	2.3	0.08
Hg	No MEC			
Cu	229	0.3	28.6	9.9
Cd	5.5	0.02	0.8	0.07
As	26	0.06		
Cr	69	0.002		0.8
Ni	89			3.6
Zn	428			0.1

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.

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 4. Measured environmental concentrations exceed standards for Cu and Ni at several sites giving RQ_{Local}s greater than one. For Cu the highest MEC was in the Port of Singapore with an RQ of c. 6, and for Ni highest values were found in the Riau stations with RQs ranging from c. 1 to 3.6.

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.

8.2.2.1 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 we characterize the relative importance of variability in each of these three elements by summarizing the factor difference between minimum and maximum values of these for each metal.

From equation 3 we can write the RQ as

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

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. Hence, this requires further clarification in terms of which standards are most appropriate for the Straits.

Were we able to more precisely describe the distributions of the variability for each of the elements of the RQ calculation we 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 we 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* we have only limited information on the extent and form of these variability distributions and so have to be content with the first stage uncertainty analysis that is summarized in Table 9. Again if this aspect of the study turns out to be of importance, then a more detailed uncertainty analysis would need to be contemplated.

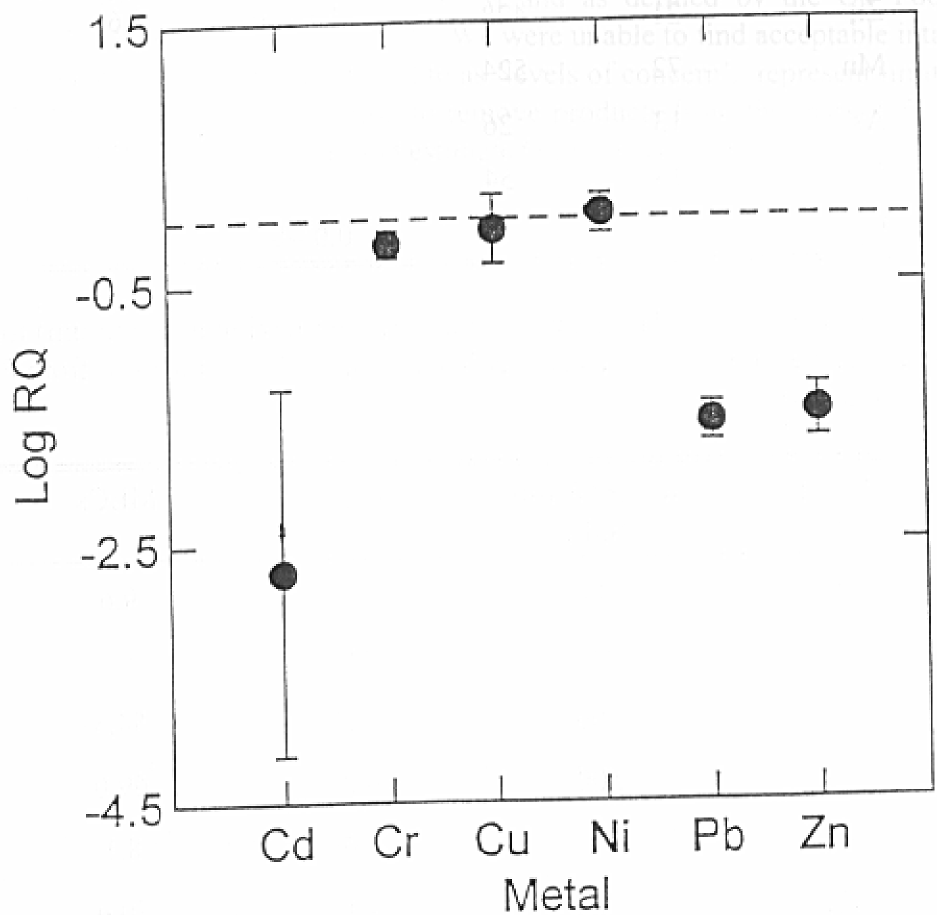


Figure 4. Average risk quotients (transformed to logarithms) ± 95% confidence limits for metals in sediments. Cu and Ni are the only metals having confidence limits that overlap the critical value of log RQ=0.

Table 8. Metal concentrations in sediments. MECs are from 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.

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

Table 9 . Metals in sediments. Maximum factor difference (maximum/minimum) for water quality standards (Table 1), K_{sw}s (from Van Der Kooij et al. 1991) and MECs (from Profile Tables 7-5, 7-6, and 7-8).

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

8.2.3 Heavy Metals and Human Health

Levels of heavy metals in fish and shellfish from the west coast of Peninsular Malaysia are given in *Profile* Table 7-9. The highest values for all metals were found in shellfish, so we will focus the initial risk assessment on metal intake from shellfish consumption. Daily intake levels of heavy metals can be calculated on the basis of estimated daily shellfish consumption and metal concentration in shellfish tissue, under the assumption that shellfish is the only source of metal intake, as follows:

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

Tolerable intake levels for Cd, Pb, Hg, Cr, Ni and as defined by the US Food and Drug Administration (FDA) are shown in Table 10. We were unable to find acceptable intake levels for Cu or Zn. Action levels, sometimes referred to as 'levels of concern', 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, the US FDA has defined action levels as follows:

$$\frac{\text{Tolerable Daily Intake } (\mu g/person/day)}{\text{Shellfish Consumption } (g \text{ shellfish/person/day})} = \text{Action Level } (\mu g/g \text{ shellfish tissue}) \qquad (6)$$

and

$$\frac{\text{Daily Metal Intake } (\mu g/person/day)}{\text{Tolerable Daily Intake } (\mu g/person/day)} = RQ \qquad (7)$$

In order to derive levels of concern for the above metals, the FDA used estimates of shellfish (molluscan bivalves and crustaceans) consumption based on a Market Research Corporation of America 14-day survey (MRCA 1988). The average and 90th percentile daily intakes of molluscan bivalves by adults (18-44 yrs) were given as 12 and 18 g/person/day, respectively. Average and 90th percentile intakes for crustaceans were 9 and 19 g/person/day, respectively. The US FDA has also estimated action levels (µg contaminant/g shellfish) for selected heavy metals in edible shellfish (see below), which when multiplied by the tolerable daily intake rates for the metals, give a consumption rate of ca. 16 g shellfish tissue/day.

Table 10. Metals in fish and shellfish. Tolerable or acceptable levels of intake for selected heavy metals as defined by the US FDA (http://vm.cfsan.fda.gov/). The tolerable daily intake level for Hg was estimated from the FDA action level of 1 ppm assuming an average seafood intake of 16 g/person/day (see Table 11). Except for Pb the figures are assumed to have been estimated for a 60 kg adult.

Metal	FDA Definition	Intake Level
Cd	Maximum tolerable daily intake	55 μg/person/day
Pb	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
Hg	Tolerable daily intake estimated from FDA Action Level of 1 ppm.	16 μg/g fish tissue
Cr	Safe and adequate dietary intake	200 μg/person/day
Ni	Provisional maximum tolerable daily intake	1.2 mg/person/day
As	Tolerable daily intake	130 μg/person/day

Estimates of fish consumption for China are in the range 0-119 g/person/day (http://www.human.cornell.edu/DNS/ChinaProject/Ranges.HTML), giving a geometric mean (using 1 g/day as the minimum) of 11 g/day, which is close to the US average for shellfish consumption. Per capita fish consumption in Indonesia is given as 16 kg/year (*Profile* p. 116), which is equivalent to 44 g/day. We assume that both of these estimates include all types of 'fish' (i.e., vertebrate fish, molluscs and crustaceans). However, for the purposes of this initial risk assessment we have assumed that 'fish' consumption is equivalent to shellfish consumption (or alternatively that the maximum concentrations in all types of seafood are similar) and used both the maximum (119 g/day) and Indonesian fish consumption values (44 g/day) to calculate levels of concern for shellfish.

Profile Table 7-9 gives 1.11 μg/g tissue as the highest level detected for Cd. Using the maximum fish consumption estimate of 119 g/day gives a Cd intake of 132 μg Cd/person/day, an intake that is 2.4 times the tolerable daily intake for Cd (Table 10), i.e., RQ=2.4 (eqn. 7). Using the rate of fish consumption for Indonesia of 44 g/day gives a daily Cd intake of 49 μg Cd/person/day, which is 89% of the tolerable daily intake for Cd (RQ=0.89).

Profile Table 7-9 gives 1.63 μg/g tissue as the highest level detected for Pb. Using the maximum fish consumption estimate of 119 g/day gives a Pb intake of 194 μg Pb/person/day, an intake that is 32 times the tolerable daily intake of Pb for small children and 2.6 times the tolerable daily intake for non-pregnant adults (Table 10). Using the Indonesian figure for fish consumption of 44 g/day gives a Pb intake of 72 µg Pb/person/day, which is still well above the tolerable intake level for children and pregnant women (RQ=12, 0-6 yrs; RQ=4.8, 7 yrs - adult; RQ=2.9, pregnant women) and approximately equal to the tolerable level for (non-pregnant) adults.

Profile Table 7-9 gives 25.2 µg/g tissue as the highest level detected for Hg. The US FDA has set an action level for Hg in fish (all types) of 1 µg/g (Tables 10 & 11). Concentrations of Hg in shellfish reported by Jothy et al. (1983) and given in *Profile* 7-9 exceed this level by 9-25 times. We have converted the US FDA action level to a tolerable daily intake for Hg, assuming that shellfish consumption averages 16 g/person/day (Table 11). This gives an estimated tolerable daily intake of 16 µg Hg/person/day. Those in the high consumption group (i.e., 119 g fish/day) consuming shellfish with 25.2 µg Hg/g tissue would have an estimated ingestion of ca. 3000 µg Hg/day, a value nearly 200 times the estimated tolerable daily intake (i.e., RQ=200). Individuals consuming 44 g fish/day would have an estimated ingestion of 1109 µg Hg/day, a value approximately 70 times the estimated tolerable daily intake (RQ=70).

Table 11. Concern levels for metal content of seafood tissue. US FDA action levels are based on tolerable daily intakes (see Table 10) and assume a consumption of 15 g bivalves/person/day or 17 g crustaceans/person/day. Values for Pb assume a tolerable daily intake of 25 µg/day (pregnant women, see Table 10). Levels of concern for those consuming large amounts of seafood were calculated using the US FDA tolerable daily intakes and a consumption rate of 119 g fish tissue/person/day.

Metal	US FDA Action Level (µg/g fish tissue)	Level of Concern for High Consumption Group (µg/g fish tissue)
Cd	3 (crustaceans) 4 (bivalves)	0.46
Pb	1.5 (crustaceans) 1.7 (bivalves)	0.21
Hg	1.0 (all fish)	0.13
Cr	12 (crustaceans) 13 (bivalves)	1.7
Ni	70 (crustaceans) 80 (bivalves)	10.1
As	7.6 (crustaceans) 8.6 (bivalves)	1.1

8.2.3.1 Uncertainty analysis

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 information becomes available regarding the toxic effects of metals in humans. With the exception of Pb, all TDIs in Table 10 are based on an 'average' adult; tolerances may vary widely as a function of age, weight, sex, etc. Our analysis assumes that shellfish 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 fish consumed is highly variable, but a critical factor in determining tolerable levels of metals in fish tissue.

FDA action levels for selected heavy metals are shown in Table 11. For comparison with FDA levels and with shellfish tissue concentrations in *Profile* Table 7-9 we calculated levels of concern assuming (rather than the US averages) a maximum shellfish consumption rate of 119 g/person/day, and these are also shown in Table 11.

Table 11 provides some idea of the effect of differences in shellfish consumption on the tolerable levels of heavy metals in shellfish tissue. Maximum detected concentrations of Hg, Pb, and Cd all exceed the action(concern) levels. However, since *Profile* Table 7-9 only provides ranges, it is not possible in this initial risk assessment to determine how frequently fish tissue exceeds tolerable levels of heavy metals for average and high fish consumers. We shall consider this question in more detail in section 8.3.3. We note 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 economic impact on the cockle industry in Malaysia (*Profile* p. 359).

8.2.3.2 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.

8.3 Pesticides

8.3.1 Concentrations in Water

Levels of organochlorine pesticides in selected Malaysian rivers were reported in *Profile* Table 7-10 and for Indonesia on *Profile* p. 361. 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).

This analysis is based upon the worst case assumption of little dilution of waters passing from rivers into the Straits. We take both highest and median values 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 5), but with considerable variability as indicated by the CVs which for four of the five pesticides exceed 100% (Table 12). The highest MECs therefore represent very worst-case scenarios. We also note that levels quoted on *Profile* p. 359 from a 1991 survey for endosulfan and heptachlor in the Kelang River were less than the median values quoted in Table 12. 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 12 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 c. 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 c. 10 lower than those based on maximum MECs. Using highest MECs all risk quotients exceed one.

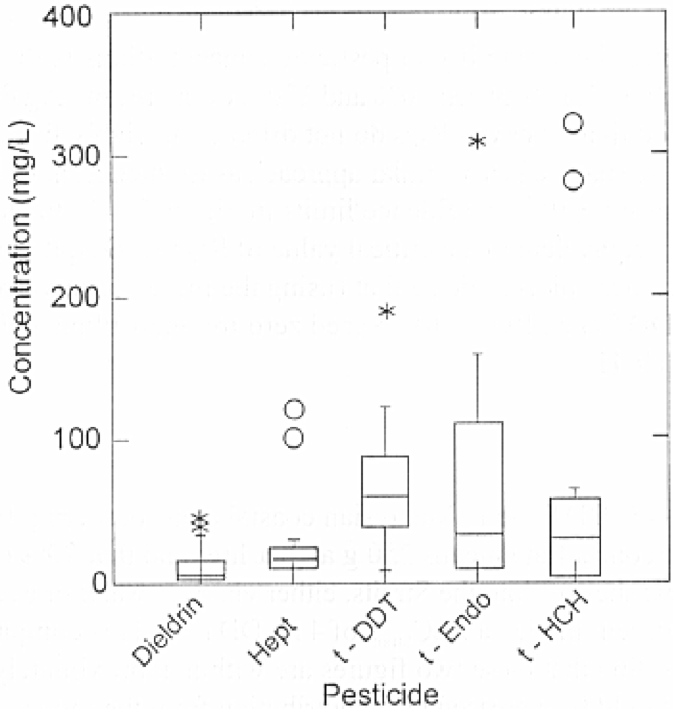


Figure 5. Box diagrams of measured environmental concentrations for pesticides in water from Malaysian rivers. The horizontal lines within the boxes represent medians; the lower and upper box borders represent 25% and 75% quartiles, respectively; whiskers represent the non-outlier range; outside values are shown as asterisks, far outside values as circles.

The concentrations of several of the pesticides were significantly correlated (Figure 6) indicating that certain rivers were contaminated with more than one organochlorine compound. In particular, the River Bernam contained the highest concentrations of t-HCH, heptachlor, and t-DDT, and the River Selangor contained the highest concentrations of t-endosulfan and dieldrin.

For the Indonesian side, there are no comprehensive estimates of pesticide residues in coastal waters (*Profile* p. 361). However, measured pesticide concentrations in water from the Siak River (quoted on *Profile* p. 361) and our risk quotient estimates are shown in Table 13. 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 programme.

8.3.1.1 Uncertainty analysis

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

With regard to standards, the differences here were not as great as for heavy metals (see above), 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 12).

One measure of among-site variability in pesticide concentrations is the coefficient of variation, which as shown in Table 12, 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. We have taken a similar approach as for metals, have log-transformed the data, and present them as means $\pm$ 95% confidence limits in Figure 7. 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 STDs) the log RQs exceed one for DDT (i.e., RQ > 10), exceed zero for endosulfan and heptachlor, and overlap zero for dieldrin and HCH.

8.3.1.2 PECs

From *Profile* Table 5-15 (DDT use in Malaysian coastal areas bordering the Straits) and assuming that 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, we calculate, using our one-compartment model, a $PEC_{Straits}$ of 4 ng DDT/L. 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 12. Pesticide concentrations in water. 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 Std. (*Profile* Table 7-11)=RQ_{AQ} or the Danish Environmental Std.(Table 3)=RQ_{DK}. MECs and STDs are in units of ng/L.

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)

Table 13. Pesticide concentrations in water from the Siak River, Indonesia, in ng/L. GM MEC=the geometric mean of the measured environmental concentrations calculated from the ranges cited on *Profile* p. 361. 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.

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
λ-HCH	0.21	2.14	-3.26	-2.25

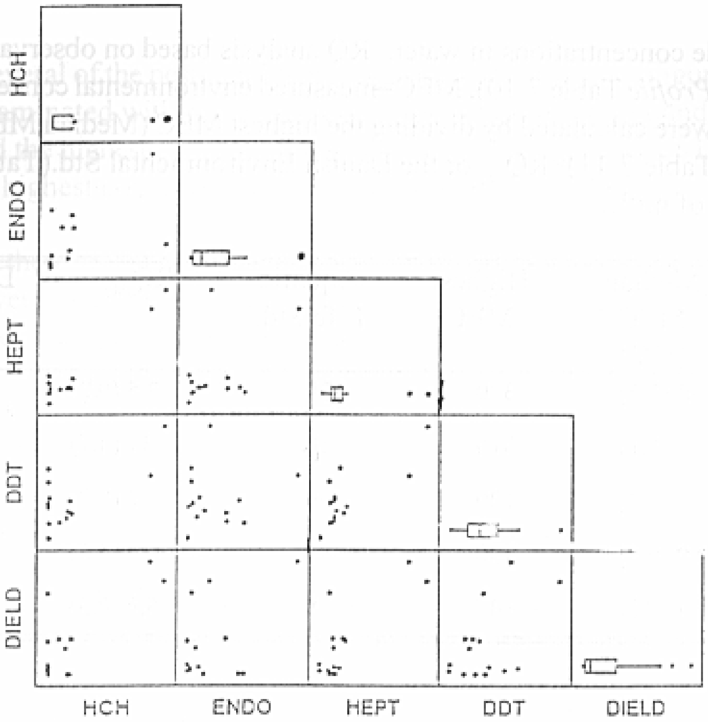


Figure 6. Scatterplot matrix showing correlations among pairs of pesticides in water from Malaysian rivers. The box diagrams shown along the diagonal represent concentration distributions identical to those shown in Figure 5.

8.3.2 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) we can calculate sediment threshold concentrations from published water quality criteria according to the equation:

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

where

C_{sed}=the critical pesticide concentration in sediment (mg/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)

On the basis of the aquatic life standards reported in *Profile* Table 7-11, we calculated critical sediment concentrations of organochlorine pesticides (Table 14). These critical concentrations were compared to the highest and geometric mean (GM) MECs from *Profile* Table 7-12. As indicated in Table 14, 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 > 1000. 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.

8.3.2.1 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. We chose the aquatic life standards from *Profile* 7-11 for our calculations, however as Table 12 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 3.692-6.2; 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), we selected the geometric mean of the reported Log K_{ow}. For some compounds (i.e., aldrin, DDT, and lindane) only a list of K_{oc}s was reported. For these substances we took the average K_{oc} and multiplied it by 0.6 to estimate K_{ow} (Van der Kooij, et al. 1991). We used a standard f_{oc} of 0.05 to calculate C_{sed}. Sandy sediments with less than 5% organic carbon will have lower critical sediment concentrations than estimated in Table 14. Finally there is uncertainty in the RQs arising from variability in the MECs. As

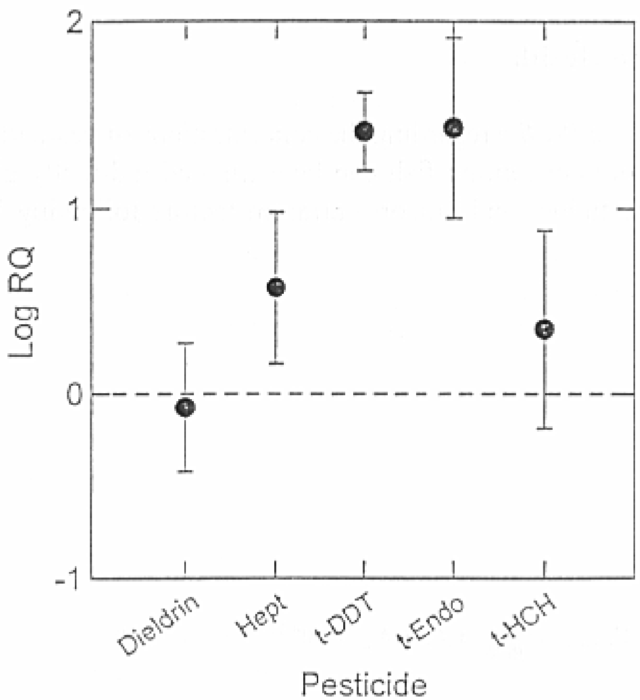


Figure 7. Means ± 95% confidence limits of log RQs for pesticides in water from Malaysian rivers. Log RQs for all pesticides are greater than or equal to the critical value of zero.

Table 14. Pesticide concentrations in sediment. 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.

Pesticide	Log K_{ow}	C_w	Highest MEC (GM MEC) (μg/kg)	Critical Sediment Concentration (μg/kg)	$RQ_{highest}$ (RQ_{GM})
pp-DDT	6.11	4	5392 (275)	158.9	33.9 (1.7)
Endrin	4.19	14	143 (31.6)	6.7	21.3 (4.7)
Dieldrin	4.78	8	4374 (187)	14.9	293.6 (12.6)
Aldrin	4.84	8	4374 (477)	17.1	255.8 (27.9)
Heptachlor	4.92	60	243 (15.6)	153.9	1.6 (0.1)
Endosulfan-I	3.13	10	1448 (309)	0.4	3620 (772.5)
λ-HCH	3.58	130	1378 (52.5)	15.2	90.7 (3.5)

only ranges were reported in *Profile* Table 7-12, we cannot assess the frequency with which critical sediment concentrations were exceeded.

8.3.3 Pesticides and Human Health

There was no information in the *Profile* regarding the concentrations of pesticides in fish or shellfish tissue. 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):

$$C_{water} = \frac{C_{fish}}{BCF} \qquad (9)$$

or

$$C_{sediment} = \frac{C_{fish}}{BCF} \; x \; f_{oc} \; x \; K_{ow} \; x \; 10^{-0.21} \qquad (10)$$

where

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

$C_{sediment}$=measured concentration of pesticide in sediment (μg/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/g)

Using these equations, the concentrations of pesticides in fish tissues can be predicted from water or sediment concentrations and compared to action levels or tolerable daily intake rates (which are estimated as the product of tolerable daily intake of the contaminant and fish consumption rate following eqn. 6). Alternatively, action levels can be used as a measure of C_{fish} to calculate critical pesticide concentrations in water and sediment, which can be compared to measured concentrations.

The US FDA has set an action level in fish of 0.3 mg/kg for dieldrin, aldrin and heptachlor. The action level for DDT is 5 mg/kg in fish, and the action level for λ-HCH ranges between 0.1-0.5 for a variety of foods (no level specifically for fish) (http://vm.cfsan.fda.gov/~lrd/fdaact.txt). As we could not find action levels for endrin and endosulfan, we used a value of 0.3 μg/g for these substances.

Table 15 shows the predicted concentrations of pesticides in fish tissue based on measured water concentrations from Table 12 and estimated BCFs for each pesticide (estimated as K_{ow} x 0.05, Van der Kooij, et al. 1991) calculated using eqn 9. RQ in this case can be calculated as the ratio of the predicted concentration of pesticide in fish tissue/action level (which is of course dependent upon assumptions made about the amount of fish consumed). Fish consumed from areas of the highest MECs would exceed the US FDA action levels for heptachlor and DDT, whereas fish consumed from areas of median MECs are predicted to have tissue concentrations below the US FDA action level for all pesticides.

Table 16 shows the predicted concentrations of pesticides in fish tissue based on measured sediment concentrations from Table 14 and calculated using eqn. 10. RQs are again calculated as the ratio of the predicted concentration of pesticide in fish tissue/action level for each pesticide. Fish consumed from areas of the highest MECs would exceed the US FDA action levels for all pesticides with the exception of endrin. Fish consumed from areas of GM MECs would equal or exceed the action levels for dieldrin, aldrin and endosulfan.

8.3.3.1 Uncertainty analysis

The US FDA action levels are based on US average fish consumption rates (c. 16 g/day), which give tolerable daily intakes of c. 80 μg/day for DDT, 1.6-8 μg/day for λ-HCH and 4.8 μg/day for the remaining pesticides. If these values are recalculated on the basis of Indonesian fish consumption rates (c. 44 g/day), they give the following levels of concern: 1.8 μg/g for DDT, 0.04-1.8 μg/g for λ-HCH and 0.1 μg/g for the remaining pesticides. For individuals consuming large amounts of fish (i.e., 119 g/day, see above), estimated levels of concern would be 0.04 μg/g for DDT, 0.0008-0.004 μg/g for λ-HCH and 0.003 μg/g for the remaining pesticides. 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 (see section 8.2.3.1) as well as uncertainties involved in estimating water and sediment concentrations (see sections 8.3.1.1 and 8.3.2.1).

The general approach that we have 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. Simulation techniques can be used to take this variability into account. As already noted, the traditional method involves Monte Carlo techniques (section 8.1). 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, though 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).

8.3.3.2 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, λ-HCH and dieldrin are to be specified, but to date this has not been done.

Table 15. Predicted pesticide concentrations for fish tissue based on water concentrations (see eqn. 9). Human health RQs were calculated as the ratio of fish tissue concentration/action level for each pesticide. Calculations were made using MECs from Table 12. Median and Highest C_{fish} values were calculated using equation 8 with median and highest water MECs, respectively.

Pesticide	Median C_{fish} (mg/kg)	RQ_{Median}	Highest C_{fish} (mg/kg)	RQ_{High}
λ-HCH	0.006	0.012 - 0.06	0.061	0.12 - 0.61
Endosulfan	0.003	0.01	0.021	0.07
Heptachlor	0.066	0.22	0.499	1.66
DDT	3.800	0.76	12.238	2.45
Dieldrin	0.012	0.04	0.142	0.47

Table 16. Predicted pesticide concentrations and human health RQs (see eqn. 7) in fish tissue based on sediment concentrations. MECs are from Table 14. Geometric mean (GM) and Highest C_{fish} values were calculated using equation 9 for GM and highest sediment MECs, respectively. BCF values were estimated as K_{ow} x 0.05 (Van der Kooij, et al. 1991).

Pesticide	Median C_{fish} (mg/kg)	RQ_{Median}	Highest C_{fish} (mg/kg)	RQ_{High}
DDT	0.446	0.09	8.745	1.75
Endrin	0.051	0.17	0.232	0.77
Dieldrin	0.303	1.01	7.094	23.65
Aldrin	0.774	2.58	7.093	23.64
Heptachlor	0.025	0.08	0.394	1.31
Endosulfan	0.498	1.66	2.333	7.78
λ-HCH	0.085	0.17 - 0.85	2.234	4.47 - 22.34

8.4 Tributyltin (TBT)

8.4.1 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 TBT in water samples from the west coast of Peninsular Malaysia as recorded in *Profile* Table 7-23 (p. 378) are in excess of this; and RQ values range between approximately one for Sg. Buloh to 140 for Port Kelang (South Port Area). The distributions of concentrations and Log RQs appeared normal with Port Kelang detected as an outlier (log RQ=2.15). A surprising observation is that levels for open waters (four sites recorded in the Table), though on average higher, were not significantly different from levels for Port Areas (six sites recorded in the Table): mean for open water=40.2 ($\pm$ 40.7=SD) ng/L; mean for Ports=75.5 ($\pm$ 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 Kelang (South Port Area) and the lowest from Pilau Jemor, but nevertheless with considerable overlap between both kinds of area. Excluding the Port Kelang outlier, the mean log RQ was 1.28 (95% CL=1.14 - 1.43) for the remaining stations.

8.4.2 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 sed 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 c. 1-60 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). Our 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.

8.4.3 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 (Willows 1994; Waldock 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). Even presuming a high water:tissue ratio of 10:1, the TBT concentrations recorded 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 235 ng/g dry wt, and such concentrations are presumably not hazardous for human consumption. However, we note that the *Profile* (p. 379) 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'.

8.4.4 Caveat

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 (see Retrospective Analysis, above). Therefore future risk assessments should address this substance more thoroughly.

8.5 Nutrients and Oxygen Demand

8.5.1 Nitrogen and Phosphorous

Using loads quoted for Malaysia in *Profile* Table 5-8 and for Singapore in *Profile* Table 5-5 (with no information available for Indonesia) we calculated a $PEC_{Straits}$ of 1.8 μg/L for nitrogen and 2.6 μg/L for phosphorous using our one-compartment model. MECs for these nutrients from *Profile* p. 15 are 0.98 μg/L for nitrogen and 0.42 μg/L for phosphorous (both maxima for surface water). It is encouraging that these are within the same orders of magnitude. 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 70-560 μg/L for NO_3-N with most in open water tending to 100, and from 12-28 µg/L for PO_4-P with most in open waters less than 20 (Eisma 1987).

The initial 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* p. 376), and so this suggests that a more detailed risk assessment should be carried out.

8.5.2 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. 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* p. 355).

8.5.2.1 PECs

To calculate a $PEC_{Straits}$ we have summed all BOD loads, and included all sources (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 we calculate a total input of 3×10^5 tonnes/year to the Straits which from our 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, 1.5×10^7 mg/person/year, as taken from *Profile* Table 5-2, which also gives a $PEC_{Straits}$ of 0.03 mg/L. In making this calculation we have used the data quoted for Indonesia but note that these were for a coastal population of 110.76 million. We have presumed that this refers to the coastal population for Indonesia as a whole, rather than for the coastal population for the Malacca Straits (which we take to be c. 10.9 million; *Profile* p. 69). We note from *Profile* p. 251 that in 1989 the total coastal loading from sewage discharge for Malaysia, Indonesia and Singapore was c. 5,000 tonnes per day, and was expected to increase to 6,000 tonnes by the year 2000.

From our calculations from *Profile* Table 5-2, we estimate an average BOD production per million of c. 40 tonnes, which multiplying by our estimate of total littoral population (20 million) gives 800 tonnes per day. We presume, therefore, that the levels quoted on *Profile* p. 251 for 1989 and 2000 represent coastal areas in general and not specifically the Malacca Straits. Clearly were this presumption to be incorrect our predicted BOD would increase by about 5 or 6 times.

We have not been able to find a PNEC/STD for BODs in estuarine/marine circumstances. However, for rivers we note that those with a BOD of < 2 mg/L are considered not polluted (Clark 1992). Hence taking 2 mg/L as an interim standard we have calculated the RQs shown in Table 17.

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* p. 355).

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.

8.6 Total Suspended Solids (TSS)

The interim standard for TSS adopted by the DOE in Malaysia is 50 mg/L. We have applied this to the data quoted from routine surveys conducted by the DOE in Malaysia (quoted on *Profile* p. 353) and the data in *Profile* Table 7-2.

To calculate $PEC_{Straits}$ we used total inputs from Malaysia (given in *Profile* Table 5-8) and Singapore (*Profile* Table 5-5) to give a value of 288,554 tonnes/year. There were no recorded data for Indonesia. However, presuming that Indonesian sources generate as much again (if not more due to extensive piggeries along the east coast of Indonesia) gives a total of c. 6×10^5 tonnes/year. Using our one-compartment model this translates into a $PEC_{Straits}$ of 0.1 mg/L.

Log RQs are summarized in Table 18. 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 our $PEC_{Straits}$ 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, we note a considerable amount of uncertainty associated with the MEC data from *Profile Table* 7-2 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 1395 mg/L giving log RQs as indicated in Table 18. However, we note that Dow 1995 (cited on *Profile* p.353) reported that most stations in the Malaysian routine survey were close to the interim standard.

We were 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). Silt particles also trap toxicants and so enter food chains of importance to humans (*Profile* p. 354).

Table 17. Measured and predicted BOD values for various regions along the Straits. For all MECs a geometric mean BOD was calculated from the ranges given (*Profile* p. 355). The $PEC_{Straits}$ was estimated using a simple, one-compartment model of the Straits. RQs were calculated using a standard of 2 mg/L.

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

Table 18. 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* p. 353)	c. 0.3-1.5
Based on *Profile* Table 7-2	mean 0.21 (±0.285=95%CL)
Based on $PEC_{Straits}$	-2.7

Table 19. 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

8.6.1 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 c. 20 mil. 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 we can also calculate a PEC_{Local} for selected rivers. Highest PECs were from the Kelang and Perak rivers, calculated by taking loadings (i.e., mass to river) and dividing by the average annual flow rates, which were estimated from data given in *Profile* Table 2-3. The PEC for Kelang 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.

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, we calculate a production level of 39 x 10^{12} mg for Sumatra and 90 x 10^{12} mg for Malaysia giving approximately 129 x 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, we can say that aquaculture is contributing a maximum of c. 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 c. 35,000 tonnes/year 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 tonnes. From *Profile* Table 5-13 and p. 271 we have a total pig population for the west coast of Malaysia of 2.5 million, 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 9. Applying analysis of variance to coliform counts among states as recorded in *Profile* Table 7-20 for recreational sites, we found significant differences for both total coliforms and fecal coliforms (total coliforms: 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 we find 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); 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).

8.8 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 through to 7-16 and the standards quoted at the bottom of *Profile* p. 362 from FAO and Marshand et al. as cited by Abdullah et al. (1994) have to be treated with some caution.

In assessing the data summarized in the tables we were unclear as to the analytical techniques used and whether corrections had been made for possible interference from other compounds. For example, measured fluorescence in seawater samples can originate from polar, i.e. oxygen or nitrogen-containing compounds, even though a part of these may have been derived from nonpolar, even petroleum precursors, e.g., by photo-oxidation or bacterial degradation (HELCOM 1990).

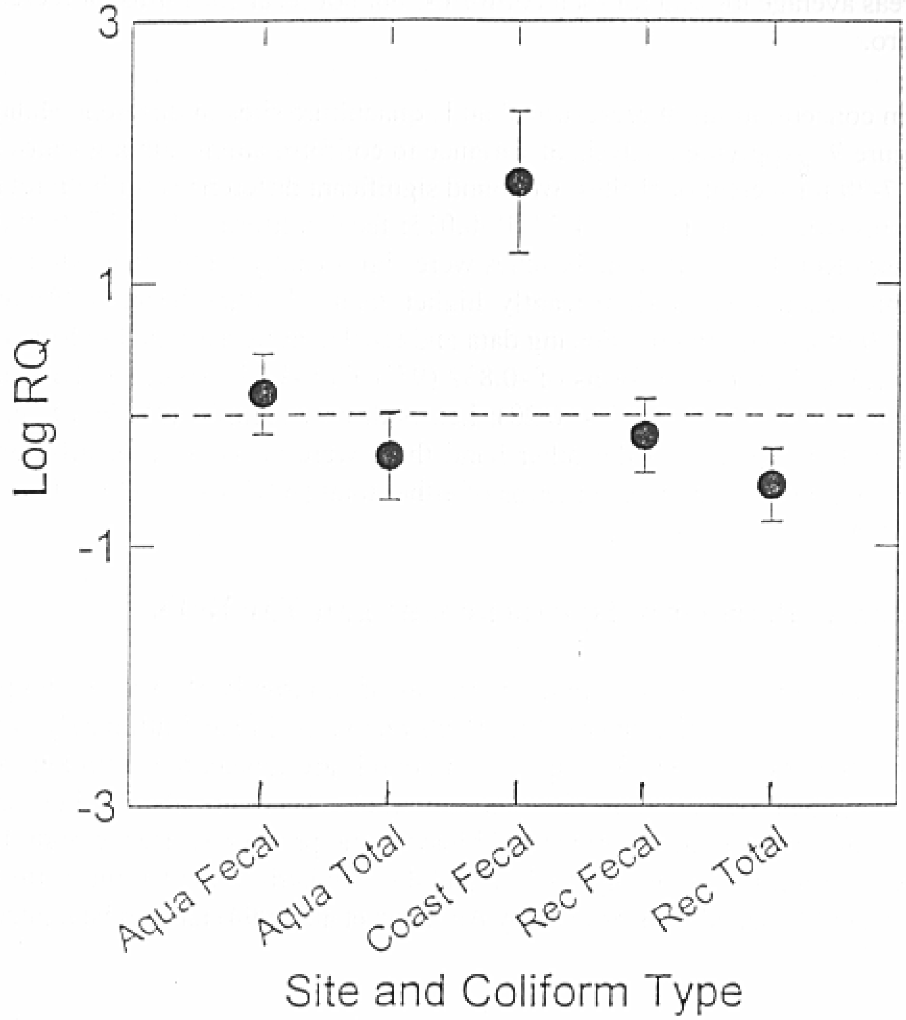
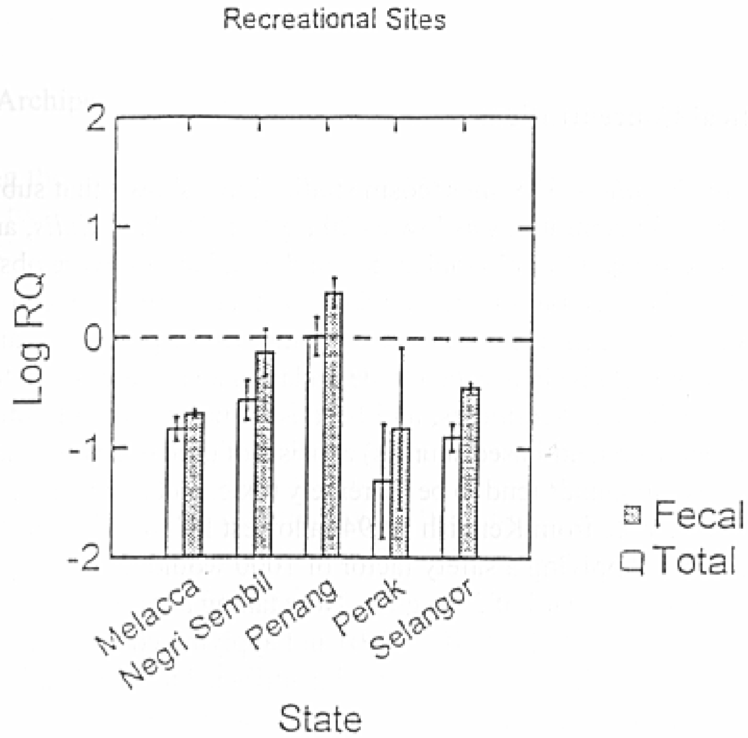


Figure 8. Comparison of mean log RQs for total and fecal coliforms among different types of stations in the Straits. Error bars represent 95% confidence limits. A dashed line is drawn through the critical Log RQ value of zero. 'Coastal' refers to coastal sites reported in *Profile* Table 7-18; 'Aqua' 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.



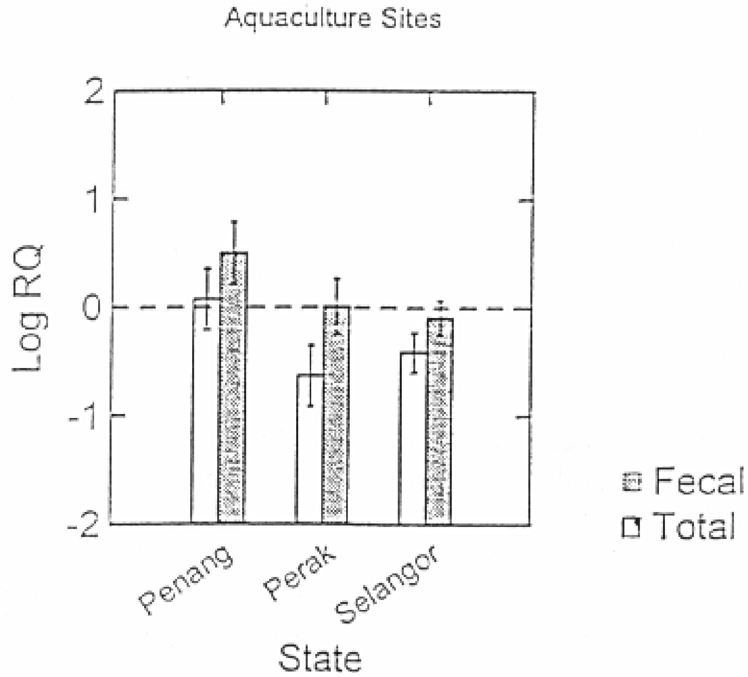


Figure 9. Comparison among different Malaysian states of log RQs for total and fecal coliforms at recreational and aquaculture sites. Measured concentrations of coliforms are from *Profile* Tables 7-19 and 7-20. Error bars represent SEM and a dashed line is drawn through the critical log RQ value of zero.

8.8.1 Critical Concentrations

As quoted on *Profile* p. 375, mesocosm studies have shown that sublethal effects can be observed at hydrocarbon concentrations as low as 20 μg/L in *Mytilus edulis*, and 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) because these are relatively resistant to degradation and so are likely to accumulate relative to other fractions, especially in sediments, and because much of the oil entering the marine environment (especially that from land-based sources) consists of used oils which are enriched in PAH (GESAMP 1993). These compounds tend to be extremely toxic, and many are known carcinogens (Neff 1979). On this basis we note from Kennish (1994) a lowest LC50 value for PAHs of between 50-300 μg/L for marine fauna. Applying a safety factor of 1000 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 2 gives a PNEC for sediments of 0.3 mg/kg dry wt (cf. the value of 100 mg/kg attributed to Marshand et al.). 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).

Trying to calculate a PNEC for oils is even more problematic. But we note from Betton (1994) that 45% of the data on oil ecotoxicity in the literature indicates 96h LC50 values between 1 and 10 mg/L. Taking a precautionary approach and applying an application factor of 1000 to the lowest value gives a PNEC of 1 μg/L.

In the light of all this variability in PNECs we have chosen to work with a value of 1 μg/L for both oil and hydrocarbons and a value of 3 mg/kg for sediments.

8.8.2 MECs

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 & 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. Though average levels appeared lower for 1995 as compared to 1992 (mean water column 1995: 23.3 μg/L ($\pm$21.0=SD, n=11), omitting the outlier for Kukup; mean water column 1992: 60.7 $\pm$108.9=SD, n=9; mean sediment 1995: 67.8 $\pm$64.98=SD, n=12; mean sediment 1992: 139.0 $\pm$80.29=SD, n=9) there was considerable variation in the data and differences were not significant ($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, we note that on *Profile* p. 375 hydrocarbon concentrations for the Riau Archipelago have been reported to reach as high as 1000 or even 11500 μg/L.). The maximum RQs for the Riau Archipelago would thus be greater than 1000.

With regard to tar balls, using the standard from UNEP (1990) of 10 g/meter, *Profile* Table 7-17 indicates problems for at least two of the beaches surveyed on the Malaysian coast and at the beaches surveyed by LEMIGAS and CENEXO at Kepulan Riau Islands, some 16 km south of Singapore.

8.8.3 Uncertainty Analysis

There are the usual sources of uncertainty involving both PNECs and MECs. In terms of the PNECs, the critical values could be an order of magnitude less than we have quoted. This would mean that RQs would be even greater than we have suggested, and so this issue should be considered in more detail and with some care. We have also drawn attention to the variability in MECs reported in *Profile* Table 7-13 through to 7-16, as one would expect from contamination deriving from various sources scattered patchily throughout the Straits area.

8.8.4 Conclusions Including Consideration of Human Health Risks

Notwithstanding the uncertainty referred to in the above section, the initial 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. Below we go on to consider possible contributions from land-based activities, particularly the refineries, and sea-based activities, especially involving shipping.

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, we are 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 polycyclic aromatic hydrocarbons (PAH) 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 rivers and domestic sewage. 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).

8.8.5 Oil Releases from Refining and Production

Here we calculate a contribution from refining by using data in *Profile* section 4.3.3 together with standard emission factors taken from the EU *Technical Guidance Notes for New and Existing Chemical Substances* (1996). The total volumes of oil involved in the refining process are quoted as follows: for Indonesia, 127300 bpd; for Malaysia, c. 300,000 bpd; for Singapore, 1.11 x 10^6 bpd. This is equivalent to a total of 5.1 million tonnes/day or c. 2 x 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 so we have presumed, conservatively, that 50% of the emission will persist in the water column. Applying these standard factors and using the one-compartment model we predict an environmental concentration from refining losses of 0.05 mg/L. Of course 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 will require a detailed understanding of the hydrodynamics of the Straits.

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

Hence the contributions of contamination from refining and production are appreciable, and likely to increase as both these industries expand. Indeed a total PEC of 0.1 mg/L is within the same order of magnitude as the MECs quoted above. Hence the contamination from refining and production seem to make an appreciable contribution to the measured levels of total hydrocarbons in the Straits.

8.8.6 Oil and Chemical Releases from Shipping

As far as oil releases are concerned, these can be of two main kinds: one due to "routine" operations such as deballasting, tank cleaning, bilge water and sludge removal; the other due to accidents. Chemical releases from shipping will occur primarily from accidents.

8.8.6.1 Operational discharges

Operational discharges of oil in the Straits are estimated as follows:

- Deballasting (*Profile* Section 6.1.1.1) - "significant"
- Tank Cleaning (*Profile* Section 6.1.1.2) - uncertain
- Bilge Water and Sludge (*Profile* Section 6.1.1.3) - 2 tonnes/day which amounts to 7.3 x 10^8 g/year
- Discharges From Small Vessels (*Profile* Section 6.1.1.4) - 2 tonnes/day from Malaysian fleet; from the Indonesian fleet approximately five times this amount - therefore in total amounting to 12 tonnes/day, which amounts to 4.4 x 10^9 g/year

Hence the total discharge due to operational activities is at least 5 x 10^9 g/year, which using the one-compartment model amounts to a $PEC_{Straits}$ of 0.5 μg/L. On the one hand this will underestimate the contribution of operational activities to the general level of oily contamination within the Straits because it makes no allowance for the deballasting contribution which though apparently taking place in the South China Sea will, because of the dominant flow patterns through the Straits, contribute to pollution there (*Profile* Section 6.1.1.1). Also omitted are incidents of illegal discharge that may be appreciable. On the other hand the calculation does not take into account the losses of oily discharges due to various biological, physical and chemical processes, and these are known to be significant (Jensen 1983). In any event, in comparison with the MEC values for oil contamination of around 20 μg/L (average of 1995 - see section 8.8.2; derived from *Profile* Table 7-13), it would appear that the operational discharges from shipping are unlikely to make a significant contribution to the general level of oily contamination within the Straits (cf. the contributions from refineries and production estimated above).

8.8.6.2 Accidental discharges

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 (c. 35%) comprises tankers (*Profile* Figure 4-3) we have calculated the number of tankers passing per year from 1982 to 1993. Then using data in *Profile* Table 6-5 we have analyzed the relationship between tanker traffic and number of casualties. Both the number of tankers and the number of tanker accidents increased with time in the period from 1982 - 1993 (Figure 10), 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 ($r=0.46$, $n=12$, $P=0.13$). 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 11) we find that risk as a percentage of the number of tankers passing through the Straits per year is relatively constant at 0.029% ($\pm$0.03=95% CL). We 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 our first estimate. These data represent total casualties and appear to take no account of size and seriousness. Between 1982 and 1993 there was an average of 4 casualties/year (95% CL=1.9 - 6.1). From *Profile* Table 6-13 we note that between 1975 and 1993 there were 0.8 major oil spills/year (> 1500 tonnes or 5000 barrels) in the Straits. However, between 1982 and 1993 the frequency reduced to 0.4 major spills/year. Taking this latter figure, this means that of the total casualties per year over this period of time, 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 20. Worldwide rates of serious casualties to oil/chemical tankers (6000 gross tons and above) was an average around 2.6% per annum until 1980 and from 1980 to 1988 approximately 2.2% (IMO 1989), and these are therefore considerably higher than our figures. For tankers of the size 100 to 5999 gross tons the average rate since 1978 is 1.26% per year with virtually no changes over the years 1983-1989 (IMO 1989). However, these data are not directly comparable with ours because they represent a percentage of tankers not a percentage of tanker voyages.

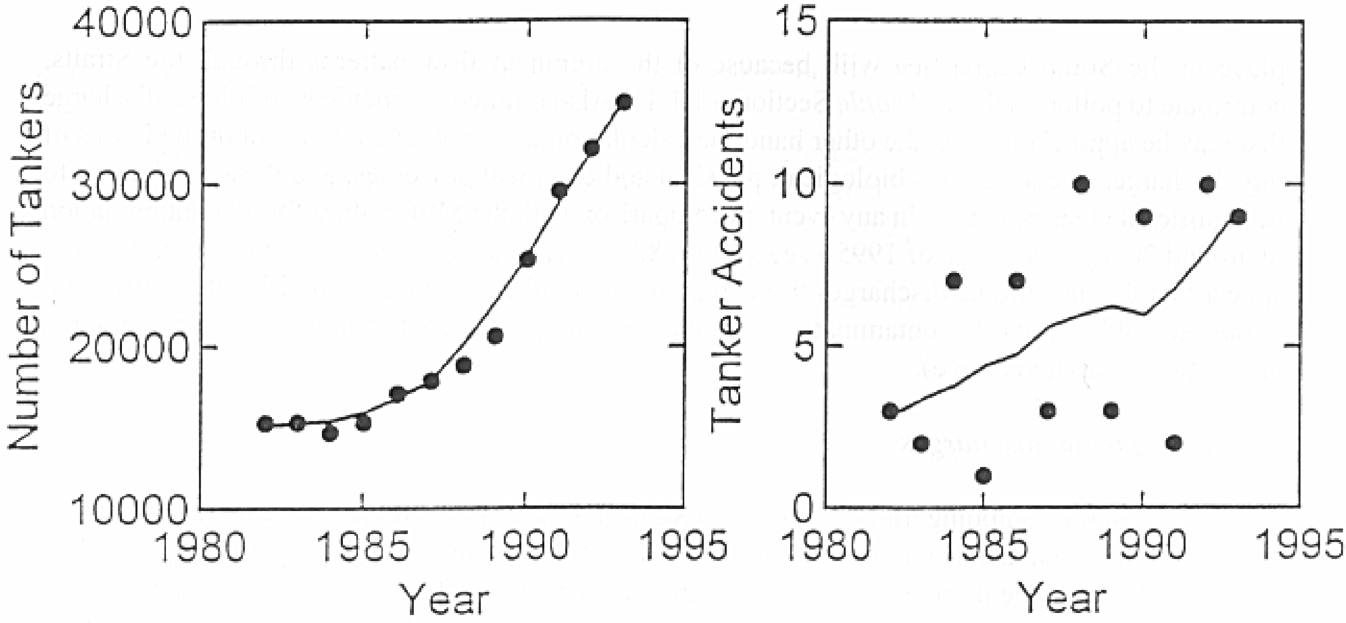


Figure 10. Total number of tankers and number of tanker accidents in the Straits between 1982-1993. Curves represent locally-weighted scatterplot smooths (LOWESS) of the data points and are robust indicators of trends.

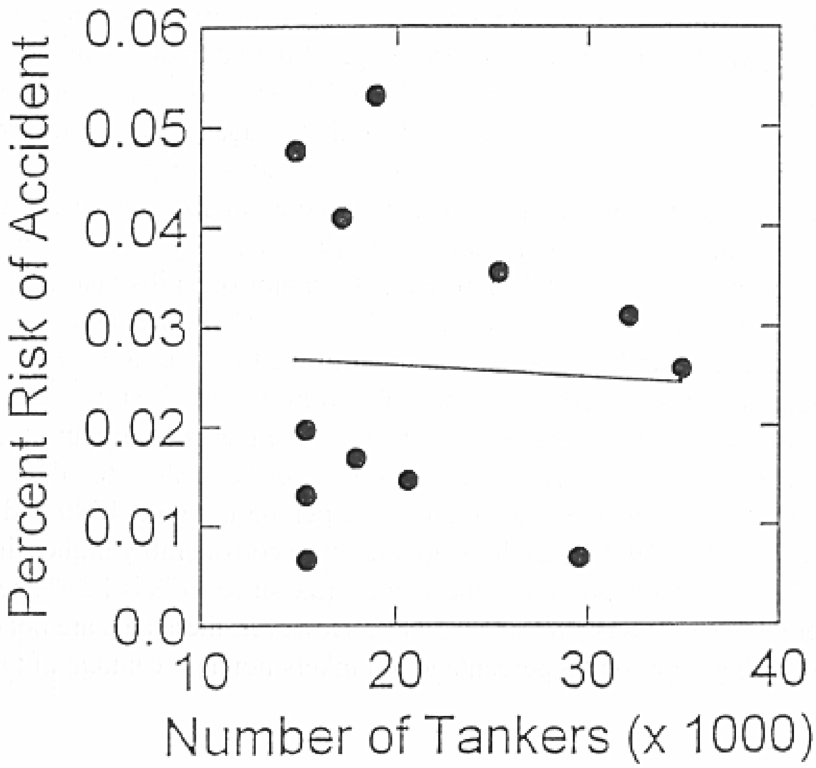


Figure 11. Estimated risk of tanker accidents as a function of total number of tankers passing through the Straits. As indicated by the non-significant linear regression through the data points, risk is independent of tanker number and averages approximately 0.029% (±0.015=SEM).

Table 20. Average risks of accidents to tanker traffic in the Straits of Malacca over the period 1982-1993.

	Numbers $Year^{-1}$	Per Hundred Tankers
Risks of Total Casualties	4 (95% CL=1.9 - 6.1)	0.029
Risks of Major Spills	0.4	0.0029

Extrapolating from this time period we 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.

The effects of a spill will depend upon a number of other factors including type and volume of cargo, proximity of critical habitats (cf. *Profile* Figure 6-9), current regime, weather, etc. The overall risk of an adverse ecological effect therefore depends upon the combination of these separate probabilities and could be modeled. Risk management procedures, though well developed within the Straits (*Profile* section 6.3.3) are somewhat reactive, intended to deal with accidents once they have occurred. It is possible to envisage more proactive systems based upon the following risk assessments. For example, routing controls might be based upon an attempt to keep risky vessels (=f(size)(cargo)(age)) away from vulnerable systems (Marine Environmental High Risk Areas, sensu Donaldson 1994). All would depend upon the identification and mapping of MEHRAs. We have previously indicated that mangrove areas appear especially sensitive to oil pollution, though aquaculture sites and bathing beaches would be of concern as well.

We note from *Profile* p. 375 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."

8.8.7 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). We have addressed the refineries above, but there were no data in the *Profile* on the other sources. These therefore deserve further consideration.

9. COMPARATIVE RISK (AND UNCERTAINTY) ASSESSMENT

9.1 Introduction

Comparing risks across different contaminants on the basis of risk quotients has to be carried out with some caution for at least 4 reasons:

i. 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.

ii. 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).

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

iv. 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 judgements from panels of experts and other stakeholders (SETAC, 1996).

Yet we believe that 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 1000 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 we have constructed comparative risk profiles for the contaminants in terms of ecological entities and human health in Tables 21 to 23.

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. We have also used RQs based on MECs rather than PECs. This has the disadvantage that the MECs are not usually unbiased with respect to conditions within the Straits. On the other hand, using MECs means that RQs are based upon recorded values and not on the output of an almost certainly oversimplified model. It is also important to note, though, that all RQs are based on conservative assumptions and often worst-case scenarios.

The tables also summarize comparative uncertainty analyses as a basis for judging the strengths of assertions based on RQs and as a way of indicating where more work is required. This analysis is based on our appraisal of importance from the quantitative assessments summarized in the appropriate sections above.

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

These are summarized in Tables 21 and 22.

For waterborne 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 (Hg (source unclear) followed by Cd and Pb), TSS and oils. Local "hot spots" inviting immediate action involve copper, TBT, TSS 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 (section 8.4) we remain skeptical about the values for TBT (because biological effects likely to result from TBT exposure (i.e., imposex) have been recorded in the Straits) and believe that more monitoring is necessary here.

However, all these conclusions need to be judged against the background of considerable uncertainty as summarized in the comparative uncertainty analysis. Apart possibly for the pesticides and TBT, that have been thoroughly worked, there is much uncertainty in the standards, and for the sediments in the partition coefficients both of which can affect the results and the conclusions drawn from them. And for all contaminants there is much variability among measurements from different places. On all counts we have tended to take worst case scenarios.

9.3 Comparative Assessment of Risks to Human Health

These are summarized in Table 23.

All, apart from TBT, suggest that exposures for certain groups in certain places can give cause for concern and require further attention, with some urgency. This is particularly so for metals. Pesticides are problematic if judged on the basis of high fish diets.

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.

The coliform counts also give cause for concern and in places require urgent attention.

The situation with regard to oil products is more problematic. We believe 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 we believe that this deserves urgent attention.

Table 21. Comparative risks and uncertainty assessments for ecological entities within the Straits for waterborne 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. Metals are based on Tables 4 and 5 using Danish water quality standards. Pesticides are based on Tables 12 and 13 using the Aquatic Life Standard. TBT data are from section 8.4.1; BOD data are from section 8.5.2; TSS data are from section 8.6, coliform data are from section 8.7; hydrocarbon data are from section 8.8.2. The largest source of uncertainty in the RQs (variability in MECs, lack of MECs, or standards) is indicated in the right-hand column.

RQs→ Contaminant ↓	<1	1-10	10-100	100-1000	>1000	Uncertainty (major sources)
Metals					Cu • Port of Singapore	Stds
Pesticides						Variability in MECs
TBT				• Port Klang		"
BOD			E.Coast • N.Sumatra			Lack of MECs
TSS						"
Oils & hydrocarbons					•• Kukup & Riau	Lack of STDs & MECs for specific HCs

Table 22. Comparative risks and uncertainty assessments for ecological entities within the Straits 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. Metals are based on Table 7 using lowest K_{sw}s and Danish water quality standards. Pesticides are based on Table 14 using the Aquatic Life Standard. TBT data are from section 8.4.2; hydrocarbon data are from section 8.8.2. The largest source of uncertainty in the RQs (variability in MECs, lack of MECs, or standards) is indicated in the right-hand column.

RQs→ Contaminant ↓	<1	1-10	10-100	100-1000	>1000	Uncertainty (major sources)
Metals		Ni	Cu			Stds
Pesticides				dield ald	endo	Stds
TBT						Lack of MECs (from closed areas)
Oils & hydrocarbons						Lack of MECs (for specific HCs)

Table 23. 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. The largest source of uncertainty in the RQs (variability in MECs, lack of MECs, standards, or lack of information on diet) is indicated in the right-hand column.

RQs→ Contaminant ↓	<1	1-10	10-100	100-1000	Uncertainty (major sources)
Metals		Cd Pb	Hg		Consumption levels? missing stds for Cu + Zn
Pesticides		High shellfish	diet		Consumption levels
TBT	<·······?				No standard
Coliform					Variability between sites
Oils & hydrocarbons	?	························>			Nature of substances. MECs. Stds

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

On *Profile* p.251 it states 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.

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 our analysis (section 8.8.5) 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.

9.5 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:

Combined RQ (Index of impact) = $RQ_x + RQ_y + RQ_z$ 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,1996). 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 were we to know something about their specific sensitivities 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 we do not have this kind of detail, which is why we have relied on RQs based on standard ecotoxicological tests that are presumed to reflect effects in ecological systems in general. However, more specific approaches are beginning to be made.

9.6 Implications for Risk Management

This comparative assessment suggests some immediate implications for risk management that we here summarize for convenience. We shall pursue these in more detail in Section 11.

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. 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 are persistent (in sediments) and in the hands of nonskilled operatives. Diffuse source are always more difficult to manage than point sources.

5. Assessment of the potential health effects from contaminated shellfish and fish requires that 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 provisions, 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.

10. ASSESSING SOCIETAL RISKS

By societal risks we mean 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 marketplaces (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. We nevertheless make the following recommendations:

i. Careful consideration needs to be given by all the major players in the Straits, governments and agencies, to the issues raised in this section.

ii. 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.

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

iv. 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.

11. FORMULATION OF AN ACTION PLAN AND OTHER RECOMMENDATIONS

11.1 Introduction

It is stated that the purpose of the *Profile* (p. 4) is "(a) to provide an inventory of resources in the Malacca Straits, with special reference to pollution risks" and "(c) identify information sources which will serve as a basis for the preparation of a management atlas, i.e. an information reference for management decision making." This initial assessment is the first step in this process: i.e. of moving from a state of the environment inventory to a more detailed analysis of pollution risks and possible needs for management action.

This initial step has been based upon risk assessment principles and practices, and in making recommendations we have presumed that further developments will be similarly risk based. As we have made clear throughout, 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. We treat each of theses issues in turn, first for ecological systems and then for human health indicating where we believe the priorities for action should be. As in the body of the text, issues for societal risks are treated separately. Finally, though it has not been part of our remit to make recommendations on management matters, a number of management "signposts" have emerged and we summarize these in this Section.

11.2 Need for Definition of Ecological Targets and Endpoints

Throughout we have made the points 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 (section 7), but there we noted that more attention needs to be given to how impacts are judged in terms of both qualitative and qualitative aspects of the targets. We were also concerned that views about decline in ecological resources were often based on anecdotal evidence, and that causation was often attributed subjectively. In particular, species losses fall into this category (section 7.2.1).

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 in which variables for assessment are agreed and in which sampling programs are clearly designed with data from them being assessed and stored in a coordinated, possibly centralized, 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 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 an important test of appropriateness. Biomarkers might, nevertheless, be of considerable importance in exposure assessment.

11.3 Need for Definition of Thresholds (Standards & PNECs)

An important aspect of prospective risk assessment is the identification of appropriate and relevant standards and PNECs. We have 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. We would propose the development of a register of agreed standards for the Straits that is revised and updated in a coordinated way on a regular basis.

We have carried out very generalized risk assessments 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 etc 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 on this.

What we have said with regard to the ecotoxicological thresholds also applies to the partition coefficients used by us in calculating the standards for sediments. We again illustrated how uncertainty here can lead to considerable uncertainty in the risk assessment. These coefficients are sensitive to conditions and will need applying with care for tropical systems. Standardized guidance for application to the Straits would again be helpful.

11.4 Need for More Confidence in MECs

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

We described this 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.

11.5 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. We illustrated their use with a simple one compartment model of the Straits. This is likely to need considerable revision.

Clearly more realistic models will be required, and these will need considerable understanding of the hydrodynamics of the Straits as a whole and of particular parts of it. We were unclear if this level of understanding already existed. Without it there is a need for research, describing the hydrodynamics and capturing the important details in user-friendly models.

We also illustrated use of release scenarios in the context of predicting environmental concentrations of oily substances from refineries. These could be developed for other sources of contamination, as they have been in the EU (European Commission 1996), but these need to be checked for appropriateness within the Straits, by discussion with the relevant industry groups. The technique can and indeed has been applied to sewage treatment facilities as well.

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.

11.6 Needs in Human Health Risk Assessment

The needs here are not so much in terms of defining threshold effects values since these have been well-worked for most substances (but we did have difficulty finding information on endrin and endosulfan). Rather largest uncertainties here are in terms exposures. The most important sources turned out to be for diet and levels of contamination. 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 not only account of average concentrations but also the likelihood of high doses in particular units of food leading to acute poisoning.

11.7 Need to Keep Under Review What is Monitored

In carrying out this initial risk assessment we have restricted our attention to substances listed within the *Profile*, 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, we are of the view 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 we therefore suggest an algorithm for narrowing down a priority list of substances for consideration within the Straits.

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 in section 9.

11.8 Needs for Societal Risk Assessment

We have interpreted societal risks as the likelihood of impairment of aspects of social welfare and the economy arising out of environmental conditions within the Straits (section 10). There are a number of ways that the economy can be impacted by deteriorating environmental conditions; but we also drew attention 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. Our 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.

11.9 Summary of Major Areas for Further Risk Assessment

From Tables 21, 22 and 23 (section 10), and using the criterion that further, more detailed risk assessments (i.e. taking into account all the needs for refinement listed in sections 11.2 to 11.6) should be carried out with priority depending on the extent to which RQ exceeds 1, then we can make the following recommendations:

(A) for water column impacts on ecological systems:

metals>TSS=oils and hydrocarbons>TBT(?)>pesticides>BOD

Though not treated in the table we would also rate nutrients of low priority (section 8.5.1).

(B) for sediment impacts on ecological systems:

Pesticides are of outstanding importance followed by oils and hydrocarbons, with metals in places.

(C) for human health impacts:

Coliforms, pesticides and metals 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.

- The sources of metals, largely industrial we presume, need to be identified and their relative contributions to general and local conditions need to be assessed. For example, industrial outputs along the River Kelang deserve attention and the Port of Singapore is a particular concern (section 8.2.1). We have already drawn attention to difficulties in identifying the source(s) of mercury pollution.
- The sources of TSS from our initial analysis (section 8.6.1), in order of importance are: loadings associated with mangrove removal and land-based forestry > industrial activities > pig farming > domestic outputs > aquaculture. These priorities need further consideration in the light of more refined risk assessments, and also 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. Indeed we envisage the kinds of models that we developed in section 8.6 acting to inform the management of these developments.
- A major source of oils and hydrocarbons from our initial risk assessment (section 8.8.5) would appear to be refining and this is likely to be of increasing significance as the industry expands. However we suggested in section 8.8.7 that contamination from municipal wastes and urban runoff can be appreciable, but there were no data on inputs from these sources in the *Profile*. We would therefore recommend the development of more sophisticated risk assessment models to anticipate both local and general impacts.
- The sources of TBT are obvious.
- The sources of pesticides are also obvious, but the challenges of carrying out more refined risk assessments here are likely to be considerable (see above).
- 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:

- mangroves and peat swamps
- seagrass beds

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