SCIENCE
RESEARCH
DEVELOPMENT

EUROPEAN
COMMISSION

Practical information and programmes

Marine science and technology

Environmental risks from large-scale ecological research in the deep sea: a desk study

EUROPEAN COMMISSION

Edith CRESSON, Member of the Commission
responsible for research, innovation, education, training and youth

DG XII/D.3 — RTD actions: Environment —
Marine sciences and technologies

Contact: Mr Klaus-Günther Barthel
Address: European Commission, rue de la Loi 200 (SDME 7/83),
B-1049 Brussels — Tel. (32-2) 29-51242; fax (32-2) 29-63024

ENVIRONMENTAL RISKS
FROM
LARGE-SCALE ECOLOGICAL RESEARCH IN THE DEEP SEA

- A DESK STUDY -

BY

HJALMAR THIEL, MARTIN V. ANGEL, ERIC J. FOELL,
ANTHONY L. RICE AND GERD SCHRIEVER

WITH CONTRIBUTIONS FROM

ANDREAS BELOW (PHOSPHORITES)
PETER M. HAUGAN (CARBON DIOXIDE)
HOWARD S.J. ROE (NUCLEAR WASTES)
ROBERT TALLACK (DISPOSAL TECHNIQUES)

PREPARED FOR

COMMISSION OF THE EUROPEAN COMMUNITIES
DIRECTORATE-GENERAL FOR SCIENCE, RESEARCH AND DEVELOPMENT
UNDER CONTRACT NO. MAS2-CT94-0086

BREMERHAVEN 1997

A great deal of additional information on the European Union is available on the Internet.
It can be accessed through the Europa server (http://europa.eu.int).

Cataloguing data can be found at the end of this publication.

Luxembourg: Office for Official Publications of the European Communities, 1998

ISBN 92-828-3517-0

Printed in Belgium

PRINTED ON WHITE CHLORINE-FREE PAPER

TABLE OF CONTENTS

Contents I

List of Figures IV

List of Tables VI

List of Acronyms used VII

Excecutive Summary X

1. Impact of Research in the Deep Sea: An Introduction 1

2. Ores in the Deep Sea 5

2.1 Manganese Nodules 7

2.1.1. Resource Description 7
2.1.2. Description of Mining Techniques 13

2.1.2.1. Continuous Line Bucket (CLB) System 13
2.1.2.2. Autonomous Robot Submersible Shuttles 15
2.1.2.3. Collector Systems 15

2.1.2.3.1. Mechanical Collectors 17
2.1.2.3.2. Hydraulic and Hybrid Collectors 18

2.1.3. Potential Effects of Mining 19
2.1.4. Research Required to evaluate the Effects of Mining 24

2.2. Manganese Crusts 31

2.2.1. Resource Description 31
2.2.2. Description of Mining Techniques 33

2.2.2.1. Continuous Line Bucket (CLB) Crust Mining System 34
2.2.2.2. Combined Mechanical Fragmentation/Hydraulic Lift Systems 35
2.2.2.3. Submarine Solution Mining System 36

2.2.3. Potential Effects of Mining 37
2.2.4. Research Required to Evaluate the Effects of Mining 39

2.3. Polymetallic Sulphides: Metalliferous Muds 41

2.3.1. Resource Description 41
2.3.2. Description of Mining Techniques 45
2.3.3. Potential Effects of Mining 48
2.3.4. Research Required to Evaluate the Effects of Mining 53

2.4. Polymetallic Sulphides: Massive Consolidated Sulphides 57

2.4.1. Resouce Description 57
2.4.2. Description of Mining Techniques 61
2.4.3. Potential Effects of Mining 64
2.4.4. Research Required to Evaluate the Effects of Mining 65

2.5. Phosphorites 67

2.5.1. Resource Description 67

2.5.1.1. East Atlantic Continental Margin (Namibia and South Africa,Morocco, Spain) 69
2.5.1.2. West Atlantic Continental Margin (the Carolinas, Georgia and Florida) 72
2.5.1.3. East Pacific Continental Margin (Peru, Chile, California and Baja California) 73
2.5.1.4. East Australian Shelf and the Chatham Rise East of New Zealand 75
2.5.1.5. Economic Potential 77

2.5.2. Description of Mining Techniques 78
2.5.3. Potential Effects of Mining 81
2.5.4. Research Required to Evaluate the Effects of Mining 83

3. Wastes in the Deep Sea 85

3.1. Munitions 87

3.1.1. Description of Waste Materials 87
3.1.2. Description of Disposal Techniques 92
3.1.3. Impacts of Disposal 92
3.1.4. Research Required to Evaluate Impacts 94

3.2. Radioactive Wastes 97

3.2.1. Description of Waste Materials 97
3.2.2. Description of Disposal Techniques 106

3.2.2.1. Low/Intermediate Level Wastes 107
3.2.2.2. High Level Wastes 108

3.2.3. Impacts of Disposal 112

3.2.3.1. Low Level Wastes 113
3.2.3.2. High Level Wastes 115

3.2.4. Research Required to Evaluate Impacts 116

3.3. Offshore Installations 119

3.3.1. Description of Waste Materials 119
3.3.2. Description of Disposal Techniques 120
3.3.3. Impacts of Disposal 121
3.3.4. Research Required to Evaluate Impacts 122

3.4. Sewage Sludge 125

3.4.1. Description of Waste Materials 125
3.4.2. Description of Disposal Techniques 128
3.4.3. Impacts of Disposal 138
3.4.4. Research Required to Evaluate Impacts 145

3.5. Dredge Spoils 147

3.5.1. Description of Waste Materials 147
3.5.2. Description of Disposal Techniques 148
3.5.3. Impacts of Disposal 148
3.5.4. Research Required to Evaluate Impacts 152

3.6. Carbon Dioxide 155

3.6.1. Description of Waste Materials 155
3.6.2. Description of Disposal Techniques 159
3.6.3. Impacts of Disposal 162
3.6.4. Research Required to Evaluate Impacts 167

4. The Acceptability of Large-Scale Research in the Deep Sea 169

4.1. The Changing Scale of Deep-Sea Research 169
4.2. The Deep Sea as a Pristine Environment? 170
4.3. Criteria for Acceptability 171
4.4. Proposed Research 175
4.5. Acceptability of Proposed Research 176

5. Literature 179

LIST OF FIGURES

Figure 1.1	Relationship between scale of impact and degree of acceptability
Figure 2.1.1	Distribution of manganese nodules
Figure 2.1.2	Diagrams of CLB and PLA mining systems
Figure 2.1.3	Diagram of generalized mining systems
Figure 2.1.4	Diagram of collector pick-up principles
Figure 2.3.1	Diagram of marine sulphide deposit formation
Figure 2.3.2	Chart of the Red Sea metalliferous mud deposits
Figure 2.3.3	Daily intake and discharge requirements of a hypothetical metalliferous mud mining operation
Figure 2.5.1	Distribution of marine phosphorites
Figure 2.5.2	Phosphorites off southern Africa
Figure 2.5.3	Phosphorites off western South America
Figure 2.5.4	Chatham Rise phosphorites
Figure 3.2.1	Quantities of radioactive wastes dumped by year
Figure 3.2.2	North-east Atlantic radioactive waste disposal sites
Figure 3.2.3a, b	Types of monolitihic and void-containing packages for low level radioactive wastes
Figure 3.2.4	Fractional release rates of radioactive substances from package types
Figure 3.2.5	Representative emplacement methods for radioactive wastes
Figure 3.4.1	Map of the North Atlantic
Figure 3.4.2	Sewage sludge and dredge spoil disposal. Option 1: the shipborne riser concept
Figure 3.4.3	Sewage sludge and dredge spoil disposal. Option 2: the floating riser concept

Figure 3.4.4	Sewage sludge and dredge spoil disposal. Option 4: the surface emplacement concept
Figure 3.4.5	Sewage sludge and dredge spoil disposal. Option 5: the ROV glider concept
Figure 3.4.6	Sewage sludge and dredge spoil disposal. Option 6: the direct descent disk concept
Figure 3.4.7	Concentrations of benthos at, and around the Garroch Head sewage sludge dumpsite
Figure 3.6.1	Total annual carbon dioxide emissions
Figure 3.6.2	Annual per capita emissions of four countries
Figure 3.6.3	Phase diagram for carbon dioxide in water
Figure 3.6.4	Diagrams of possible carbon dioxide disposal options
Figure 3.6.5	Conceptualisation of droplet plume dynamics for carbon dioxide release at great depths

LIST OF TABLES

Table 2.1.1	Sample calculation of sediment volume mobilized during nodule minig operations
Table 3.1.1	Explosive articles made safe by U.K. Royal Navy teams, 1982 - 1983
Table 3.2.1	Dumping in the North-east Atlantic, Belgium, 1967 - 1982
Table 3.2.2	Dumping in the North-east Atlantic, France, Germany, Italy, Sweden, 1967 - 1969
Table 3.2.3	Dumping in the North-east Atlantic, Netherlands, 1967 - 1982
Table 3.2.4	Dumping in the North-east Atlantic, Switzerland, 1969 - 1982
Table 3.2.5	Dumping in the North-east Atlantic, UK and Belgium, 1949 - 1970
Table 3.2.6	Dumping in the North-east Atlantic, UK, 1967 - 1982
Table 3.2.7	Locations of dumping areas and quantities of waste dumped
Table 3.4.1	Chemical composition of sewage sludge dumped off New York
Table 3.4.2	Dry weight quantities of sewage sludge produced annually by the countries contracted to the Oslo Convention
Table 3.5.1	Operational and systems performance criteria for deep ocean disposal

LIST OF ACRONYMS USED

AFERNOD	Association Francaise pour l'Etude et al Recherche des Nodules (France)
AM	Amorphous-silicate zone
BIE	Benthic Impact Experiment (USA-CIS, Japan, IOM, India)
BP	Befor present
BPEO	Best Practical Environmental Option
CCFZ	Clipperton-Clarion Fracture Zone (North Pacific Ocean)
CFA	Carbonate-fluorapatite
CFC	Chlorinated fluorocarbon
CIS	Commonwealth of Independent States
CLB	Continuous Line Bucket
CO	Central oxide zone
COD	Chemical oxygen demand
CRC	Containment and regulation cover
CRESP	Coordinated Research and Environmental Surveillance Programme
DISCOL	Disturbance and Recolonization experiment in a manganese nodule area of the deep south-eastern Pacific Ocean (Germany)
DOE	Department of the Environment (UK)
DOMES	Deep Ocean Mining Environmental Study (USA)
DOP	Detrital-oxidic-pyritic zone
DSDP	Deep Sea Drilling Project
EC	European Community
EEZ	Exclusive Economic Zone
ENEA	European Nuclear Energy Agency

ESTG	Engineering Studies Task Group
EU	European Union
GDR	Former German Democratic Republik
GESAMP	Group of Experts on the Scientific Aspects of Marine Pollution
HMSO	Her Majesty's Stationery Office (UK)
IAEA	International Atomic Energy Agency
ICRP	International Commission on Radiological Protection
IFREMER	Institut Francais de Recherche pour l'Exploitation de la Mer (France)
IMO	International Maritime Organisation
IRA	Impact Reference Area
JET	Japan Deep Sea Impact Experiment
KCON	Kennecott Consortium
LDC	London Dumping Convention
LSA	Low specific activity
MAST	Marine Science and Technology
MESEDA	Metalliferours Sediments Atlantis II Deep
MMS	Minerals Management Service (USA)
MOR	Mid-ocean Ridge
NEA	Nuclear Energy Agency
NERC	Natural Environment Research Council (UK)
OECD	Organisation for Economic Co-operation and Development
OMA	Ocean Mining Associates
OMCO	Ocean Minerals Co.
OMI	Ocean Management Inc.
OSPARCOM	Oslo and Paris Commission (for the Prevention of Marine Pollution)

OTEC	Ocean Thermal Energy Conversion
PCB	Poly-chlorinated biphenyl
PDO	Pilot Disposal Operation
PLA	Preleveur Libre Autonome (French robot deep sea mining system)
PMO	Pilot Mining Operation
PPMT	Pre-Pilot Mining Test
PRA	Preservational Reference Area
ROI	Return on investment
ROV	Remotely Operated Vehicle
RTD	Research, Technological Development and Demonstration
RWMC	Radioactive Waste Management Committee
SEBA	Sea-bed Activities (a Working Group under OSPARCOM)
SOAEFD	Scottish Office for Agriculture, Environment and Fisheries Departement
SRA	Stable Reference Area
SU_1	Lower sulphide zone
SU_2	Upper sulphide zone
SWG	Seabed Working Group
TAG	Trans - Atlantic Geotraverse
TUSCH	Tiefsee-Umweltschutz (Deep Sea Environment Protection) (Germany)
UK	United Kingdom
UKOOA	UK Offshore Operators Association
ULCC	Ultra Large Crude Carrier
USA	United States of America
VLCC	Very Large Crude Carrier

EXCECUTIVE SUMMARY

The RISKER (Environmental **Risk**s from Large-Scale **E**cological **R**esearch in the Deep Sea - A Desk Study) report was initiated through the European Commission and addresses the question of whether marine ecological research could in itself be unacceptable from an environmental standpoint. The international study group specifically considered the environmental acceptability of conducting the research needed to assess:

- The impacts of past uses of the deep ocean for various types of waste diposal.

- The sustainability of ocean resources in the face of any future exploitation of sea-bed minerals.

- The environmental acceptability of any disposal of waste into the deep ocean for which alternative management approaches (e.g. waste minimisation, re-cycling and land-fill) become insufficient to circumvent environmental problems as a result of the growth of human populations and the concomitant increases in industrialisation.

Recently it has become evident that results from traditional scale sampling experiments and observations cannot be extrapolated to asses the impact of full industrial extraction of deep-sea minerals, and so recent oceanographic investigations, perforce, have had to conduct much larger-scaled experiments. The question then arises as to the environmental acceptability of these experiments.

Chapter 2 is devoted to the extraction of deep-sea minerals. After a brief introduction to the mineral resources of the deep sea, a series of Subchapters examine each of the major potential mineral ores which may be mined in the future. Subchapters 2.1 and 2.2, respectively, address manganese nodules and crusts, the major oxide minerals found in the deep ocean. Subchapters 2.3 and 2.4 review the two dominant forms of sulphide mineralisations that may become future sources of metals, namely metalliferous muds and massive consolidated sulphides, respectively. Another important mineral resource found in some shallower deep-sea areas are the phosphorites which are the subject of Subchapter 2.5. Each of these five Subchapters provides a resource description, reviews possible mining techniques, examines potential mining effects and lists the research necessary to evaluate these possible effects.

Chapter 3 addresses the past and potential use the deep ocean for waste disposal. Several categories of waste have already been disposed of into the oceans, but seldom in a manner that would meet modern criteria of acceptability. Nearly all such dumping is currently severely curtailed by international law, because it is generally considered that too little is known about the impacts, and indeed some argue it never will be justified. In spite of the current moratorium, options for using the deep ocean for certain types of waste disposal are being evaluated because of our realisation that society's ability to cope with its ever-growing waste management problems is approaching a crisis point. After a brief introduction to the disposal of wastes in the sea, the Chapter is organised into a series of Subchapters. The first two discuss wastes that have been dumped in the sea in the past, but are unlikely ever to be so again (munitions (3.1) and radioactive wastes (3.2)). However, monitoring of their dumpsites are considered to be necessary, not only to reassure the public that these sites threaten neither Mankind nor the long-term integrity of oceanic environments, but also to provide useful insights into what long-term impacts may result from any future ocean disposal of other types of waste. The third Subchapter (3.3) examines the environmental impacts associated with the accidental and intentional disposal of large structures into the deep ocean. The Subchapters 3.4 and 3.5 discuss the possible disposal of sewage sludge and dredge spoils, both of which can still be disposed of legitimately into deep water. It is envisaged that in the future deep ocean disposal of such wastes may prove to be the optimum waste management approach compared to land disposal, both in terms of regional and global impacts. The final Subchapter (3.6) deals with the even more thorny question as to whether the deep ocean can (or should) be used as a means to reduce the atmospheric carbon dioxide which will result from the burning of fossil fuel, and which will have predictably serious environmental impacts. Each Subchapter includes a description of the waste material, considers possible disposal techniques and examines the environmental impacts expected to accompany the disposal activities. Some methods that may be appropriate for the disposal and/or containment of sewage sludges and dredge spoils on the deep-sea floor are introduced and discussed in the sewage sludge Chapter. The choice of disposal technique will make a considerable difference to the experimental evaluation that would necessarily be undertaken before any such disposal could be contemplated on an industrial scale. The final section of each Subchapter therefore summarizes the research activities deemed to be necessary for the evaluation of the potential effects imposed on the environment by commercial-scale waste disposal.

The acceptability of conducting large-scale ecological research and experiments in the deep sea is discussed in Chapter 4. In the absence of internationally agreed criteria, the following have been adopted for this report:

- The objectives of any experiment can be reasonably expected to contribute towards the environmental evaluation of full-scale industrial activity.

- The impacts must be monitorable.

- There must be no persistent adverse effects on diversity at regional scales, i.e. no species will be driven to extinction, no exotic species shall be introduced, the area of destructive impacts shall be >0.1% of the area (or volume) inhabited by any community.

- The experiment must not impair any ecological processes.

- No living resources shall be adversely effected.

- The experiment must not interfere with other legitimate uses of the ocean.

- There must be no critical or mass transport pathways to convey significant quantities of deleterious substances or organisms back to Man.

- Risks to human health and safety must not be increased.

The research proposed in the various Subchapters of Chapters 2 and 3 are concurrently assessed against these criteria.

The scales and nature of the research necessary to improve the assessments of environmental impacts of the full industrial scale activity for each use of the deep sea are also examined. These range from traditional laboratory and field observations (including monitoring) of existing impacts and ocean conditions, to novel large-scale experiments linked to industrial feasibility trials. The criteria of acceptability are used to assess the various scientific studies identified as being necessary for the environmental impact studies of each activity. It is concluded that the traditional scales of investigations fall well within the limits of acceptability. Impacts associated with such studies occur over a relatively small area and are negligible. For several types of waste (radioactive material, offshore installations and munitions) monitoring of existing disposal sites will provide significant information on these and other uses of the ocean, and should be given a high priority even if no further disposal is ever conducted. These monitoring activities can be carried out with traditional methods and no significant environmental effects are expected.

Before any mining of the mineral resources discussed and any disposal of wastes such as sewage sludge, dredge spoils or carbon dioxide can be approved, the authors consider that it will be essential to conduct large-scale experiments well in advance of the development of full-scale operations. These would need to ascertain that the scale of the impacts will not exceed the limits of sustainability, to develop monitoring protocols for the industry, and to reassure the general public that the operations will carry a tolerable level of risk both for the human population and the ocean ecosystems. It will also be important to conduct base-line studies to allow adequate monitoring of the operations to ensure that the actual impacts do not exceed those predicted. Criteria of acceptability remain poorly developed and their much greater sophistication will be essential for good environmental management and public acceptance.

Mining operations will involve extensive destruction of seafloor communities during the extraction of the minerals, and potentially wider impacts on the water column and sea-bed by discharges of tailings resulting from the initial processing on the mining platform. Experimental studies must be conducted at scales that can be extrapolated confidently to the scale of impact, both in space and in time, of full industrial operations. As long as the precise details of the mining technology that will be employed remain undecided, general precautionary, large-scale experiments are the means to learn about the reactions of the community to various disturbances and to be prepared for subsequent more targeted research approaches. The experiments will have to encompass studies of both the sea-floor and the water column. Since some of the open questions about impacts are identical no matter which ore is being mined (such as the behaviour of plumes and the sedimentation of tailings after discharge), many of the experiments and subsequent verifications will be common to all mining operations and should be conducted in advance. The final study level to evaluate mining impacts will be the monitoring of pilot mining operations (PMOs).

In a similar manner, large-scale experimentation will have to be conducted prior to any disposal of wastes, freely or contained, into the ocean. In addition, better baseline knowledge of the distributions of deep-ocean biota and the factors controlling them will be required if site selection, both for the experiments and any eventual disposal, is to be carried out according to any objective scientific rationale. The experimental design will need to involve nested scales of impact ranging from laboratory experiments to *in situ* experimental manipulations involving stepped increases in the volumes and delivery rates of each type of waste. Prior to each increase in experimental scale, the acceptability of the impacts will need to be re-evaluated and the decision taken whether or not to scale up the experimentation. The final experimental level will be the pilot dumping operation (PDO).

The proposal to use the deep ocean as a repository for carbon dioxide in order to keep the transient increases of atmospheric concentrations within acceptable bounds will also need experimental approaches. These should span many scales since the reactions of organisms, communities and total ecosystems to increases in ambient carbon dioxide concentrations are still very poorly understood. Two large-scale experimental approaches are envisioned: a total ecosystem experiment conducted in a semi-enclosed fjord, and an open ocean experiment involving the discharge of large quantities of solid, liquid or gaseous carbon dioxide into the deep ocean.

Few of the large-scale experimental designs considered by the authors that might be conducted to address the questions of risk assessment of industrial-scale activity approach the highly precautionary limits adopted. The basic acceptablility criteria assumed are that the scale of any impact should be restricted to <0.1% of any specific habitat and that no species should be driven even close to extinction. Even so, such are the uncertainties about deep ocean ecology that the thorough monitoring of any, and every, future pilot mining operation (PMO) and pilot dumping operation (PDO) which may eventually lead to commercial scale activity remains a high priority.

Deep-sea research and experimentation is expensive and requires a long-term commitment. Deep-ocean ecosystems are tuned to a very slow pace and so their response to, and recovery from experimental and/or industrial interventions can be expected to be slower than those of terrestrial and shallow-water ecosystems. Consequently, the research proposed, if it is to provide answers and not be just a ploy to persuade the public that "something is being done", must be conducted over long periods of time. A start must be made well before socio-economic pressures precipitate rapid development of commercial-scaled activities in the deep-sea. The time needed clearly falls well beyond the normal range (1 - 5 years) of socio-economic developments. Sound scientifically based evaluation and guidance can only be achieved if the supporting research has been intensive enough and carried out for long enough prior to the initiation of commercial activities. Such precautionary large-scale experiments will increase the chances of resolving the global-scale environmental problems which will predictably be generated by the accelerating growth of the World population and the inevitable associated increases in demands on environmental resources (both oceanic and terrestrial). These precautionary deep-sea experiments should be given high priority.

1. Impact of Research in the Deep Sea: an Introduction

All ecological work carries with it the risk of generating potentially damaging impacts on the environment being studied. However, with the exception of very fragile or spatially restricted environments, the normal scale of sampling or *in situ* experimentation involved in ecological research is such that the resulting impacts can usually be disregarded. This is particularly true in the case of the deep sea beyond the continental shelves. In comparison to the vast size of this environment, which is the largest on earth, covering more than $300 \cdot 10^6 km^2$, the impact of conventional research has been extremely small. The area directly impacted by all the sampling and experimental operations using trawls, dredges, corers, traps and landers in the 130 years or so since serious deep-sea research began must be considerably less than $100 km^2$, about 0.00003% of the total surface area. Even the inclusion of the areas indirectly impacted by, for instance, mud plumes disturbed by trawling and dredging operations would increase this figure by a factor of no more than two or three. Nevertheless, even in this vast environment there are two possible exceptions. First, the relatively small, uncommon and spatially separated hydrothermal vent communities have been the subject of intense research since their discovery in the 1970s. Some individual vents have been visited many times and may have been impacted significantly as a result. Second, there has been an increasing tendency to concentrate research on small scale spatial and time variabilities, sometimes over several years. The relatively small areas of widespread deep-sea environments, investigated in this way may also have been impacted by the cumulative effect of many individually insignificant operations. But apart from these special situations, conventional research clearly could continue indefinitely at the same, or a considerably enhanced, scale without causing any significant impact.

However, over the last two or three decades it has become increasingly clear that normal scale deep-sea research is inadequate to answer many current and future questions about this environment. For example, current estimates of the species richness of the deep-sea benthic environment, now believed by many biologists to be among the most biodiverse on earth, are based on extreme extrapolations of detailed information from very small samples totalling no more than twenty or so square metres (see Grassle and Maciolek 1992). Similarly, the fact that awareness of the existence of such dramatic phenomena as hydrothermal vents (Jones 1985, Hessler *et al.* 1988) and the seasonal sedimentation to the deep-sea floor of aggregated phytodetritus (Billett *et al.* 1983, Rice *et al.* 1986, Thiel *et al.* 1989) is very recent indicates the inadequacy of conventional research.

The shortcomings of normal research methods are particularly apparent in relation to actual or potential impacts of anthropogenic activities in the deep ocean, either resource extraction or waste disposal. In this respect, it is quite impossible to extrapolate from the results of disturbance and recolonization experiments conducted on scales of a square metre or two to the likely effects of a manganese nodule mining operation or the deep-sea disposal of some millions of tonnes of sewage sludge. Clearly, to understand the effects of such commercial-scale operations, research must also be conducted at an appropriately sized scale, both in space and time.

But what is the "appropriate scale"? In some cases it may be that conventional-scale research techniques applied to natural or man-made analogues would provide adequate information. In others, this might be unacceptable since no suitable analogues exist. The DISCOL (Disturbance and recolonization experiment in a manganese nodule area of the deep south-eastern Pacific Ocean) project became the first large-scale impact experiment in the deep sea, aiming at an assessment of the effects of extensive environmental disturbance (Schriever *et al.* 1991, 1992, Thiel 1992, Thiel *et al.* 1992, Thiel and Foell 1993, Thiel and Schriever 1990). This project attempts to understand at least some aspects of the environmental impacts of manganese nodule mining by intentionally disturbing an area of more than $11km^2$ of the central south-eastern Pacific at a depth of 4150m. The impact created during this experiment, while very much larger than that of traditional deep-sea research, is nevertheless still relatively small when compared to industrial scales of activities. Consequently, it is unclear whether it would be possible to extrapolate from even this relatively large data set in order to assess the acceptability or non-acceptability of the physical disturbance caused by a commercial-scale operation which is expected to disrupt on the order of 1 square kilometre of sea floor per day, for over 300 days per year, over a minimum twenty year period (see Subchapter 2.1 below).

Developing marine ecological research beyond traditional small scales into such large-scale approaches raises the question of whether these research methods are in themselves environmentally acceptable. It was from this perspective that we considered the call for proposals (93/C203/19) which stated that:

> "The sole objective and topic covered by this call is the assessment of any possible risk likely to affect the marine environment in association with research, monitoring and surveying in marine sciences and technologies."

This call was based on decision 110/94/EC of the European Parliament and the Council on the 4th Research, Technological Development and Demonstration (RTD) programme stating that:

> "..... there should also be technology assessment monitoring the possible risks, advantages and dis-advantages of new technologies developed in this framework programme."

The relevant decision of the Council specific to the Marine Science and Technology (MAST) programme (94/804/EC) stated that:

> "..... an analysis should be made of possible socio-economic consequences and technological risks associated with the programme."

The MAST programme is not currently funding any research in the deep ocean other than that conducted at the traditional scales which, as already pointed out, are so small that their environmental impact is minimal. But we foresee that because of the increasing interest in the deep oceans, particularly from the industrial and commercial sectors, future EU programmes may well be asked to fund such activities. The objectives of this study are therefore to anticipate this development. Its specific aims are to attempt to identify:

- the types and scales of research necessary to assess the ultimate effects of a range of anthropogenic activities contemplated to take place in the deep sea, and in turn to

- assess the likely environmental consequences of this research itself.

There is an inverse relationship between the scale at which environmental research is conducted and the acceptability of the associated environmental effects (Fig. 1.1). Generally, the larger the scale of the research, the less the degree of acceptability. Nevertheless, even the large-scale experiments, monitoring activities and pilot level industrial operations discussed in this report remain, in our opinion, well within acceptable levels when using the criteria established. The authors are aware that this report is based on the views of only a few deep-sea ecologists and should therefore not be considered definitive. Nevertheless, we hope that the report will stimulate discussions on future research activities in the deep sea, on the responsibilities of oceanographers towards the oceanic environment, and on the environmental acceptability of large-scale oceanic research.

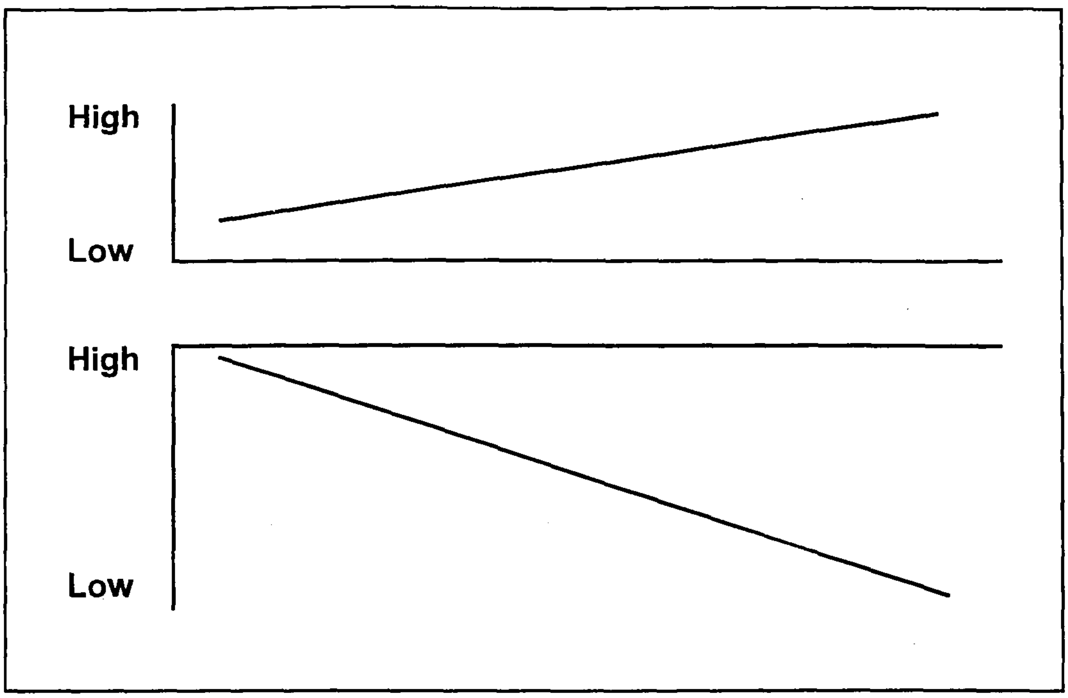

Figure 1.1 Relationship between scale of impact and degree of acceptability.

2. Ores of the Deep Sea

The first metalliferous ores from the deep sea were obtained in 1873 during the circumnavigation of the British research vessel HMS *Challenger*. Strange potato-shaped concretions were brought up in trawl samples from the bottom of the deep Atlantic Ocean and subsequently found to be even more abundant in the Pacific. When these were analysed by the ship's chemist, J.Y. Buchanan, they were found to be rich in manganese and iron and to contain smaller quantities of other metals. Although they aroused considerable interest over the next few decades, particularly as to their origin, these so-called manganese or ferromanganese nodules were not initially regarded as a potentially exploitable resource.

Increasing deep sea research activities in the second half of the twentieth century, and the use of new techniques including acoustic surveying and seabed photography, revealed remarkably high abundances of these nodules in certain areas. In the meantime, the increasing industrial demands for raw materials upgraded the nodules into a potentially valuable resource - if their recovery could be technically and commercially feasible. Restrictions in the availability of terrestrial supplies, and the uncertainties introduced by the vagaries of international politics have stimulated nodule exploration and the development of techniques for their exploitation.

Phosphorites at continental shelf and upper continental slope depths had also already been discovered during the *Challenger* Expedition. However, these and other ocean mineral resources were considered scientific curiosities and received little commercial attention until the publication of *The Mineral Resources of the Sea* (Mero 1965) aroused industrial interest. This first comprehensive publication on the subject stimulated additional research and, during the following 15 years, further deep-sea investigations discovered a variety of new ore types including manganese crusts and polymetallic sulphides in the form of metalliferous muds and consolidated masses.

The question of potentially serious environmental impacts arose early in the discussions of deep ocean mining. Extracting the resources was recognised as a potentially large-scale intrusion into the previously, relatively unimpacted deep sea, and it became evident that environmental safeguards would be a major concern. This realisation resulted in the inclusion of environmental studies in the mining tests of the early seventies, and further investigations have been carried out alongside the later industrial tests and in independent projects to the present day (Thiel *et al.* 1991).

One of the principal results of these environmental investigations has been the realisation that oceanographic research at normal, relatively small scales does not provide an adequate basis for an understanding and evaluation of potential large-scale industrial impacts (Thiel 1991). This insight led to a new approach to deep-sea environmental investigations, monitoring recolonization following an experimental large-scale disturbance in an attempt to mimic certain aspects of commercial mining activities (Thiel and Schriever 1990, Foell *et al.* 1990). These new research approaches, together with more conventional research techniques, are considered in this report.

Chapter 2 is divided into five subchapters. Manganese nodules and crusts, the most important oxide mineralizations found in the oceans, are discussed in Subchapters 2.1 and 2.2, respectively. Similarly, the major sulphide mineral types that may at some point be exploited include metalliferous mud deposits (Subchapter 2.3) and massive consolidated sulphides (Subchapter 2.4). These mineralizations have the potential for being exploited for their rich metal content at some point in the future. Although not a metal ore and generally found in shoaler waters, phosphorite deposits represent another potentially significant target for future ocean mining operations and are reviewed in Subchapter 2.5. Each subchapter is divided into four sections which provide, respectively, a description of the resource, a description of mining techniques contemplated for recovery, a discussion of the potential effects of mining operations, and a view of research appropriate to evaluating the environmental risks associated with conducting mining operations.

Subchapter 2.1 on polymetallic nodules has assumed some sort of a lead function. Industrial explorations and technical developments concentrated on this ore type already at a time, when crusts and massive sulphides were hardly known and certainly not recognized as a potential ore. Environmental studies were conducted also for the mining of metalliferous muds, but the modern approaches, the large scale environmental experiments, remained restricted to nodule deposits. Much of what is said in this subchapter, therefore, applies to the following parts of this report respectively.

Although this section of the report deals with the general exploitation of deep sea resources, it does not cover hydrocarbons. While oil and gas exploitation has already penetrated the deep sea, with drilling already being conducted at depths in excess of 1000m and expected to reach 3000m in the near future, consideration of the risks associated with these activities was specifically excluded in the call for proposals (EU document 93/C203/19). However, the results presented in this report are also generally applicable to environmental research activities related to hydrocarbon exploitation from the deep seabed.

2.1. Manganese Nodules

2.1.1. Resource Description

Of all the potential mineral resources of the sea, manganese nodules have received the greatest amount of study over the years since they were first recovered from the sea floor during the *Challenger* Expedition of 1872-76 (Murray and Renard 1891). Although fossil nodules had already been found in the Alps by C.W. von Gümbel in 1861 (Jenkyns 1977), it is the *Challenger* Expedition which is generally credited for their initial discovery after nodules were dredged from the Atlantic Ocean floor in February, 1873, about 160 miles south-west of the Island of Ferro in the Canary Islands group. At first considered to be not much more than a scientific curiosity, increasing interest in their resource value and economic potential was stimulated by the publications of John Mero during the 1950s and early 1960s. Reflecting the escalation of interest, the nodule literature began to expand exponentially as nodules appeared with increasing frequency in scientific, technical, and popular articles. In a definitive work entitled *The Mineral Resources of the Sea* (Mero 1965), the author pointed out the enormous potential value of nodules as metal ores which resulted in considerable commercial and industrial attention.

Often characterised as brown- to black-coloured, potato-shaped concretions, manganese nodules (perhaps more correctly referred to as ferromanganese nodules or, better yet, polymetallic nodules, terms which more accurately reflect their chemical makeup) range from less than 1cm up to about 25cm in diameter. In addition to size, other nodule characteristics such as shape, texture, abundance, coverage, and chemical composition can also vary widely, often over relatively short distances (Seibold 1978). Understanding the physical, chemical, and biological parameters that control the observed variability, often described as "patchiness", has been central to much of nodule research for the past three decades.

In terms of external morphology, individual nodule concretions are typically spheroidal (rounded), ellipsoidal (elongated and rounded), or discoidal (flattened and rounded) in shape, although asymmetrical forms also occur, especially among smaller nodules before accretional growth has smoothened their contours. In general, young nodules tend to grow through relatively uniform accretion on all sides and thus exhibit spheroidal to ellipsoidal morphologies. As nodules continue to enlarge and press deeper into the sediment, the supply of materials required for growth seems less evenly available and there is a tendency to assume a more discoidal morphology. This may be a result of diminished growth on the lower and upper surfaces which are pressed into the sediment or emerge out of the semi-liquid layer, respectively. The lateral edges or rims remaining within the semi-liquid (sometimes also called peneliquid) layer may continue to grow at higher rates, eventually giving the entire nodule a flattened or compressed appearance (discoidal) with distinctive thickening around the edges, a feature often referred to as an equatorial belt or rim.

The surface textures of polymetallic nodules range from relatively smooth to very granular, gritty, and coarse depending upon the conditions under which accretion has occurred (Sorem and Fewkes 1979). Differences in texture can be correlated with the type of growth processes the nodule is currently undergoing or has undergone during its growth history. Nodules in the north-eastern tropical Pacific's Clipperton-Clarion Fracture Zone (CCFZ) often exhibit a smoothened upper surface resulting from hydrogenetic growth, that is, a growth mode in which colloidal particles of hydrated metal oxides in sea water precipitate directly onto the nodule surface (Halbach *et al.* 1988). Most polynodules and the often abundant mononodules with smooth surfaces also have hydrogenetic origins and are particularly common in low sediment areas such as in proximity to or on the flanks of seamounts. Current patterns in such environments often preclude formation of a high water content sediment layer which appears to be necessary for other modes of nodule growth to occur.

On the other hand, the rougher texture on the lower surface and edges of many CCFZ nodules is attributable to diagenetic growth, a growth mode in which metals or metal compounds present in the interstitial water of the sediment provide the primary source of precipitates. In areas where the semi-liquid sediment layer is particularly thick and encloses much or all of the nodule surface, as for example, in certain localities of the Peru Basin of the eastern tropical South Pacific, the surface may be rough on all sides or may even have a smoother underside. Very large (more than 10cm in diameter), brittle nodules covered by botryoidal mammillae and sometimes described as resembling a blackened head of cauliflower commonly occur there.

All nodules have at their centre a growth nucleus of some foreign material. This acts as a starter substrate upon which new nodule material accretes at typically very slow rates (usually stated to be in the order of a few millimetres per million years although more rapid growth rates have been determined at some localities). Although such items as shark teeth, fish bones, sponge spicules, skeletal elements of plankton, basalt fragments, pieces of pumice, and bits of clay may constitute the growth nucleus, the most commonly found nuclear material is simply a fragment of another nodule (Halbach *et al.* 1988).

Partially as a result of crowding, several individual nodules (mononodules) in close proximity may actually inter grow with each other to form a multi-nucleated polynodule. Such nodules are particularly common in areas near to or upon the flanks of seamounts and other rocky outcroppings where erosional processes are believed to supply copious quantities of rock fragments which then serve as nuclei for hydrogenetic nodule growth. Coverage in such areas may approach 90% as hydrogenetic growth on abundant nuclei gives rise to dense deposits of smooth mono- and polynodules.

In the technical literature distinction is drawn between the commonly used terms "coverage" and "abundance" as employed in describing nodule deposits. Coverage refers to the amount of a given area of sea floor obscured by nodules and is typically expressed as a percentage. It is usually visually estimated from sea floor imagery such as photographs or television. Abundance refers to the mass of nodules per unit of sea floor area and is typically stated quantitatively using such units as $kg \cdot m^{-2}$ or $lb \cdot ft^{-2}$. A value for nodule abundance is usually determined using such sampling devices as box corers or grab samplers, but may also be indirectly estimated from sea floor images. High coverage areas, however, do not necessarily imply good abundance values. The small mono- and polynodules often found near areas of high topographic relief commonly exhibit coverages of 80-90%, but have low abundance values due to their small size. On the other hand, some potentially high quality mine sites with larger discoidal nodules have abundance values in the range of 10-15 $kg \cdot m^{-2}$, but may only exhibit a coverage of 20-30% or so. From an ocean mining perspective, abundance is the more critical factor since economically efficient mining depends upon collecting a certain tonnage per day. Most often, a minimum requirement of about 5000 metric tonnes (dry) per day is given in the technical literature.

Ore grade is another critical element in evaluating the economic potential of a nodule deposit. The higher the average concentration of value metals such as copper (Cu), nickel (Ni), and cobalt (Co) a nodule deposit exhibits, the greater the commercial value of the deposit. Typically, nodules with combined Cu + Ni concentrations in excess of 2% are considered to be at the minimum level to justify commercial interest, although price fluctuations of individual value metals can significantly alter this generalisation. Recently, mining interest has been rekindled for the nodule resources of the Cook Islands area which occur at high abundance levels (up to $60 kg \cdot m^{-2}$) and exhibit Co concentrations of up to 2.5 times the CCFZ averages (Clark *et al.* 1995, Pryor 1995, Bechtel Corp. 1996), while Cu + Ni values range near or below CCFZ average.

The chemical composition of polymetallic nodules also varies considerably as can be demonstrated by analyses of individual nodules obtained within the same sample. Compositional differences have even been demonstrated within different regions of an individual nodule specimen. In addition to such obvious considerations as abundance, coverage, and topography, it is the overall chemical composition of a polymetallic nodule deposit which decides its economic significance and primarily determines its suitability for commercial exploitation.

On a purely economic basis, the metals for which polymetallic nodules will one day prove to be an important source of ore are Ni, Cu, Co, and manganese (Mn). The amounts of Ni, Co, and Mn contained in nodule deposits of the world oceans are believed to greatly exceed known terrestrial reserves. Only land-based Cu reserves remain larger than what is thought to potentially be derivable from oceanic mines. Nodules, however, contain many other elements in addition to these value metals. It is believed that nearly all elements found in sea water are to some extent also contained in polymetallic nodules. Although the assay values in nodules of such metallic elements as gold (Au), silver (Ag), platinum (Pt), titanium (Ti), molybdenum (Mb), and zinc (Zn) are not sufficient to justify mining, recently there has been renewed interest in utilisation of some of these minor constituents. A study is being conducted in the United States which is examining the feasibility of using the high platinum content nodules of the Blake Plateau off the south-eastern coast (Georgia and northern Florida) as a catalytic stack gas cleaner in coal-fired power plants (Woolsey 1992, Buchannon 1995). Reduced nodule material could then be further processed for value metals (J.R. Woolsey, pers. comm.).

There appears to be some correlation between nodule morphology, chemical composition, sea floor topography, and patterns of distribution. For example, nodules that have been accreted through primarily hydrogenetic processes tend to have greater Co and iron (Fe) contents with depressed Mn:Fe ratios, but are usually lower in Cu and Ni than nodules that formed through diagenetic means. Such nodules tend to occur on or near seamounts and similar high relief areas. On the other hand, diagenetic nodules with elevated Mn:Fe ratios (lower Fe and Co concentrations, but higher in Mn) tend to have higher Cu and Ni content and are usually associated with relatively flat areas showing little topographic relief. Nodule chemistry may also be correlated to some extent with water depth (Greenslate *et al.* 1979), but bottom topography (which, of course, influences water depth) may still be the most significant controlling factor with economic deposits generally confined to bathyal and abyssal depths between 4000 and 6000m.

While the vast majority of deposits do not meet the minimal abundance, concentration, and topographic standards required for economic consideration, nodules have been found to occur in all oceans as well as in certain freshwater environments (Fig. 2.1.1), primarily resting at or near the sediment-water interface. Nodule deposits are therefore essentially surficial and two-dimensional (although significant numbers may be found buried in the sediment at some localities and may be targeted for harvesting with more advanced mining systems than presently conceived), but often cover vast areas in contrast to the more geographically restricted, three-dimensional deposits associated with most familiar terrestrial ores. In order to be economically viable and be assured of access to sufficient ore material to harvest over a minimum 20 year mine life expectancy, an ocean mining operation will require mining claims on the order of thousands of square kilometres. The size of individual mining claims and exploration areas currently registered under the United Nations seabed mining regime and under various national legal constructs ranges from about 60,000 to 170,000km^2 and is indicative of the vast spatial requirements needed to conduct ocean mining for manganese nodules.

The fact that most existing claims are located at water depths of 4000 to 6000m within the north-eastern tropical Pacific Ocean, especially between the Clarion and Clipperton Fracture Zones (CCFZ) in the so called "manganese nodule province", further indicates that not all nodule occurrences are of equal commercial interest. Although claims have also been made for exploration and mining rights to nodule deposits in the central Indian Ocean and in the Peru Basin of the tropical South Pacific, it is in the CCFZ that the characteristics of the deposits appear to best meet commercial requirements.

From the point of view of the ocean miner, a nodule deposit must meet certain criteria in order to be considered for mining. As previously stated, the average value metal concentration of the nodules must fall within the range of economic interest. Nodule abundance and coverage must be sufficient to offset mining costs and the topography of the mining locale must be sufficiently benign to allow effective and efficient deployment of costly mining equipment.

Not all areas within a mining claim can meet these criteria. Due to variations in sea floor conditions, current patterns, and topographic variability, nodules may be entirely absent or may occur in economically uninteresting levels of ore grade and abundance. Sediment characteristics and sea floor topography may also not be appropriate for conducting ocean mining due to rock outcroppings, sediment thickness, increased slopes, and similar conditions. If one assumes a deposit with an average abundance of 10 kg $\cdot$ m^{-2} (wet) and an economic requirement for the mining of 1.5 million metric tons (dry) of nodules per year to sustain a mining operation for a minimum 20 year viability, then about 5000 metric tons (dry) must be mined per day (assuming mining operations are conducted for an average of 300 days per year with 65 days of down time for repairs, maintenance, or related reasons). If one further includes an overall nodule recovery efficiency of 70% for the first generation of mining technology, it is possible to calculate that about 1km^2 of sea floor will be mined on a daily basis or about 6000km^2 over the life of a 20 year minesite (Thiel *et al.* 1991). The smallest current claim application encompasses about 65,000km^2 with most others reaching well above 100,000km^2 in area. It would not be unreasonable to conclude that the majority of ocean miners anticipate only about 10-20% of their application areas to ultimately be minable, thereby justifying such large initial claims.

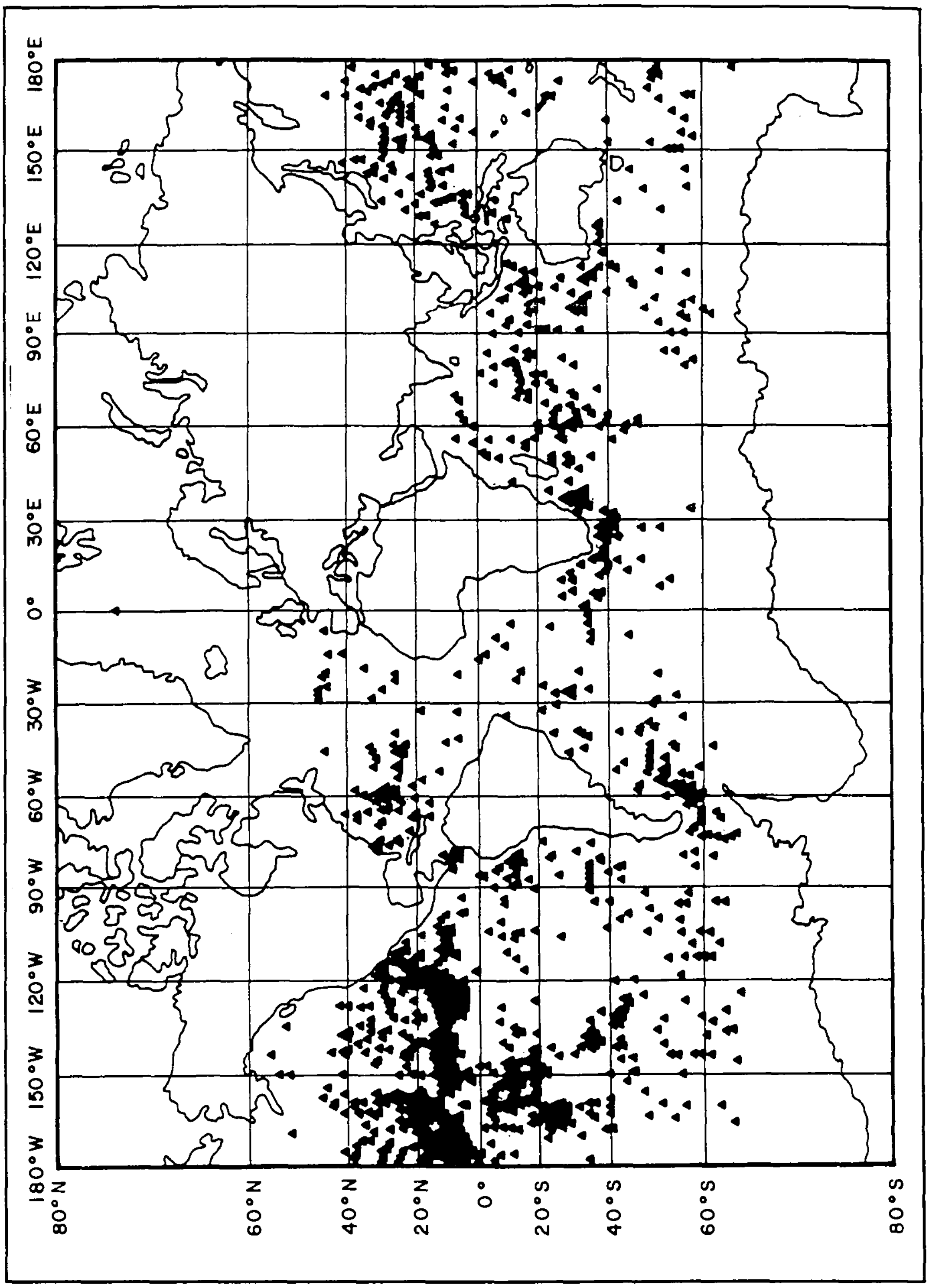

Figure 2.1.1 Distribution of manganese nodules based on stations in the Sediment Data Bank of the Scripps Institution of Oceanography as of March 1980 (from Thiel *et al.* 1991 redrawn from McKelvey *et al.* 1983)

2.1.2. Description of Mining Techniques

Ocean mining presents some unique challenges to those intent upon harvesting nodules for profit or upon achieving assured access to sources of vital raw materials. The most promising mine sites are located far from land and are covered by nearly 5000m of water. Environmental conditions at these depths include total absence of sunlight, low temperature (1-3°C), and immense pressure (500 atmospheres or 50kN · m^{-2}). The complex mining machinery must not only be capable of operating consistently and effectively under these conditions, but must also be remotely directed by the operator aboard the surface mining platform relying upon various direct and indirect sensing technologies to guide the operation. Even though the deposits are essentially surficial in nature, nodules must be efficiently collected over large surface areas in order to amass the minimum daily tonnage required for sustaining a viable operation. The ore must then be lifted to the surface and transferred to transport vessels that shuttle the raw material between the mining locality and the processing plant site.

Possible solutions to these problems have been offered since the earliest days of ocean mining development. They range from the ridiculous to the sublime, from the crude to the highly complex, and from application of near brute force methods to employment of the latest that high technology has to offer. To date, only a few methods have actually been put to test at the proof-of-concept or pre-pilot mining test (PPMT) level, and even then in a very limited sense. New methods and approaches continue to be conceived and existing methods modified as ocean mining development proceeds, albeit at a much slower pace than during the past few decades.

2.1.2.1. Continuous Line Bucket (CLB) System

Among the simplest and cheapest suggested solutions to the problem of collecting nodules from the deep sea floor was to extrapolate technology already in use for mining other raw materials in shallow water or near shore areas. The various dredging techniques being employed in mining sand and gravel, as well as numerous types of placer ore deposits, were thought to be extendible to harvesting nodules from deeper waters. One of the earliest proponents of such an approach was Commander Y. Masuda of Japan who in 1966 conceived a mining system employing a continuous line or able with attached buckets (Masuda *et al.* 1971, Mero 1978). The CLB system is deployed from a slowly moving ship (Figure 2.1.2) which relies on side thrusters in order to move at right angles to the bucket line and thereby effect harvesting of nodules at depth. The lower end of the loop of buckets engages the seabed and nodules are mechanically scraped into the buckets. The entire loop is rotated by means of friction drives at the bow and stern of the mining ship.

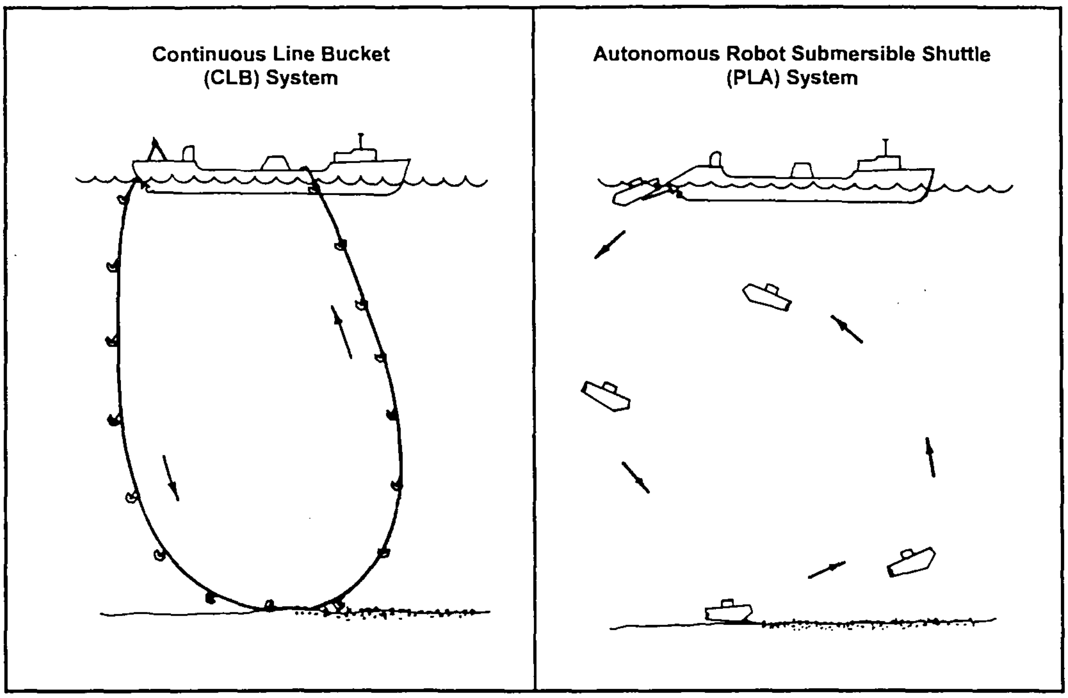

Figure 2.1.2 Continuous Line Bucket (CLB) mining system and the Preleveur Libre Autonome (PLA) or autonomous robot submersible shuttle system. (not to scale) (from Thiel *et al.* 1991)

Bucket designs varied, but generally required that they consist of wire mesh, perforated steel boxes, or similar porous structure so as to allow sediment and other materials collected with the nodules to be largely rinsed off during the hoisting process. Some type of bucket closure mechanism was also thought to be desirable to prevent excessive loss of harvested ore during retrieval. The depth of bucket penetration into the sediment was believed to be controllable through a combination of adjusting mining ship speed and manoeuvrability, and use of angled teeth or sled-like runners on the buckets themselves.

The CLB system was subjected to a series of laboratory and shallow water at-sea tests in the late 1960s. Results were sufficiently encouraging to conduct deep water tests in 1970 and 1972, but the system performed to expectations only for short time periods (Masuda *et al.* 1971, Mero 1978). Mechanical problems with the traction drives aboard the mining vessel, with the connectors used to attach buckets to the cable, and with tangling of the outgoing and the incoming bucket line legs resulted in the system being inoperative much of the time. A French concept for utilising two surface ships to handle the incoming and outgoing CLB loops, respectively, and thus allow for further separation with less chance of entanglement, was never tested due to lack of sufficient funding.

A modified CLB concept in which buckets are transported in an enclosed tube-like channel has been suggested, but not yet tested at sea. The CLB system is currently considered to be insufficiently controllable, excessively subject to entanglement, and too inefficient in ore recovery for practical use in nodule exploitation, although it remains under consideration for use in other ocean mining activities (see Subchapter 2.2 below). A simpler wire-line trawling operation is under consideration for harvesting the cobalt-rich nodules of the Cook Islands region (Bechtel Corp. 1996).

2.1.2.2. Autonomous Robot Submersible Shuttles

In contrast to the relatively simple CLB system, a very sophisticated nodule mining system involving a series of robot submersible shuttles was under study and initial development in France during the early 1980s (Moreau 1984, Lenoble 1989, 1990). This system, called Preleveur Libre Autonome (PLA), envisioned a series of remotely controlled or programmed autonomous submersible vehicles (Fig. 2.1.2) designed to descend to the seabed like a glider, utilise two Archimedean screws for sea floor propulsion while gathering nodules, and return to the mining vessel relying on syntactic foam incorporated into the vehicle's structure for buoyancy (Marchal 1984).

Further work on the development of the PLA system has ceased for the moment since costs increased substantially with the size of the vehicle required to collect nodules at economically reasonable rates and capacities. Such advanced concepts may be revived in the development of 2nd and 3rd generation mining technology, but current costs and technical constraints, particularly with respect to considerations of buoyancy versus capacity and the substantial energy requirements of a robot submersible, eliminate this system from further immediate consideration.

2.1.2.3. Collector Systems

Representing a combination of the hydraulic dredging technique widely used in shallow waters combined with those borrowed from the offshore oil industry, collector systems for use in ocean mining are widely believed to be the equipment of choice for first generation nodule mining at abyssal depths. A number of variations of collector systems have already been designed and tested while others are still under development or are awaiting sea trials. Collector systems have several basic components in common.

All systems require a nodule collector or pick-up device which may be mounted on a collector vehicle capable of self-propulsion and steerage (active mobility) or it may simply be towed by the mining ship using a pipe string as a tow cable (passive mobility), although some level of steering capability may also be incorporated into such passive designs. Nodules would be picked up off the sea floor using hydraulic, or mechanical methods, or some combination thereof (Fig. 2.1.3). In some systems, nodules would then be fed into a hopper or storage device before entering the suction point at the bottom of the pipe string for upward transport to the mining vessel. Other systems would eliminate the hopper and feed nodules directly to the pipe string. In some cases, preliminary crushing or size sorting of nodules may be required to prevent clogging.

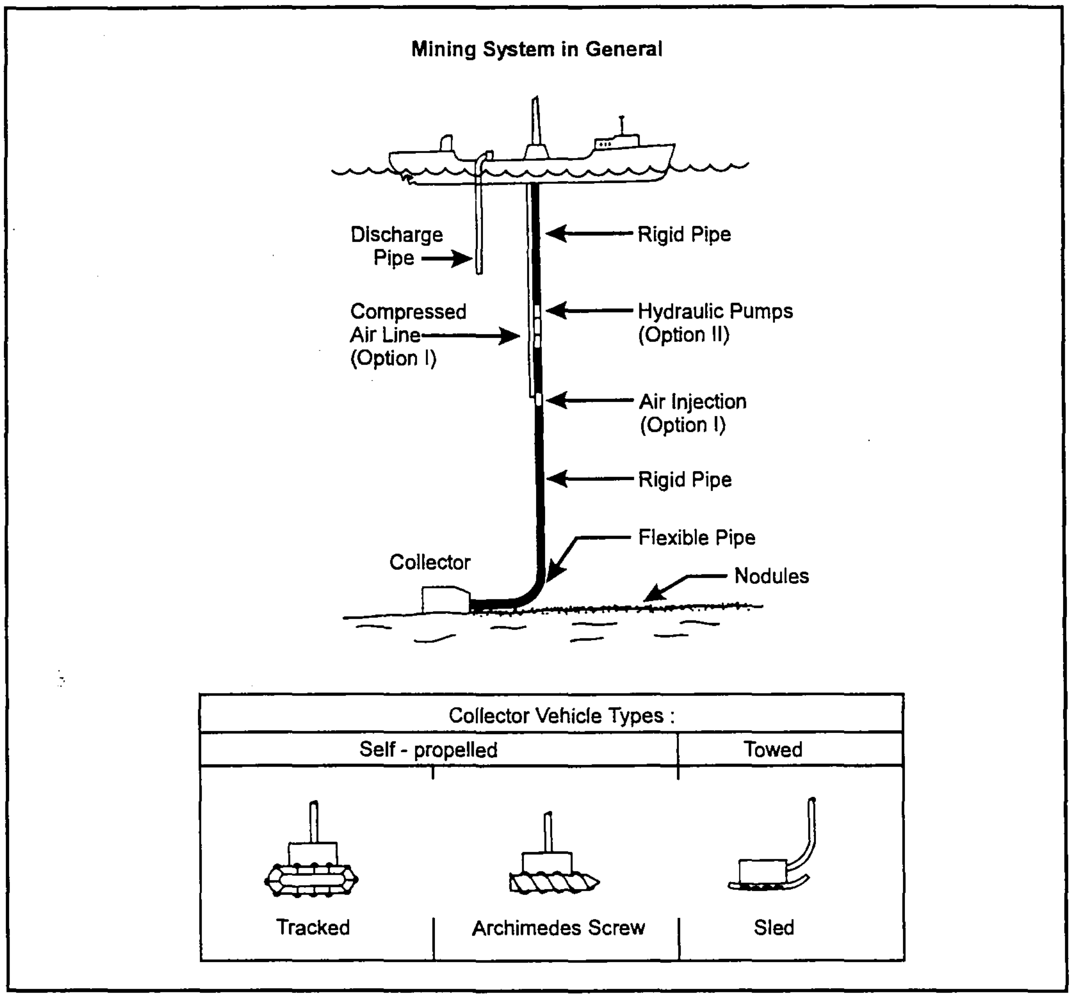

Figure 2.1.3. Generalized mining system using either the air injection (Option 1) or submerged hydraulic pump (Option 2) approach. Collector vehicles can either be self-propelled, for example, tracks or the Archimedes screw principle, or towed in a sled-like configuration. (not to scale) (from Thiel *et al.* 1991)

At the surface the ore solids are separated from the water used as transport medium. The rigid pipe string is usually connected to the collector vehicle by a flexible conduit section. The upper end of the pipe string is typically mounted to a gimballed platform to alleviate excessive forces upon the pipe string generated by wave and ship motions. Upward transport through the pipe string is powered either by air injection (air-lift method) or submerged in-line pumps (hydraulic lift method) which would serve to create the flow of water required to lift the nodules to the surface. Transfer to the ore carriers is generally conceived to occur in the form of a slurry with additional separation of ore and water aboard the transport ship. The ore carriers would then transport the nodule ore to the processing site which is expected to be a terrestrial location, although at-sea processing has also been discussed presenting its own suite of potential environmental problems, not further discussed in this report.

2.1.2.3.1. Mechanical Collectors

Mechanical collectors generally utilise the potato harvester or similar type of conveyor belt system and incorporate a wide assortment of mechanical lift devices (Fig. 2.1.4). In terms of deep sea mining, a nodule harvesting machine would consist of some type of cutting edge at the base of the conveyor belt, the conveyor belt itself rigged with rows of flexible or rigid pins or scoops to conduct nodules along the conveyor belt, and a hopper for intermediate storage of nodules before their introduction into the pipe string.

An obvious drawback of such a system is that considerably more sediment than nodules may be collected and conducted to the hopper. The cutting depth can to some extent be regulated or adjusted in order to minimise collection of sediment and maximise collection of nodules. However, the high variability of nodule deposits makes such optimisation difficult at best.

A successful test of what was basically a prototype mechanical collector mounted to a sled and connected to a pipe string that utilised the air-lift to achieve vertical transport was conducted in the summer of 1970 (Geminder and Lecourt 1972). In this first PPMT, Deepsea Ventures, Inc., later to become the research and development contractor to the Ocean Mining Associates (OMA) consortium, tested such a collector system at 762m depths on the Atlantic Ocean's Blake Plateau (Kaufman and Latimer 1971, Lecourt and Williams 1971). The towed collector sled was fitted with a series of tines to exclude material above a certain cut-off size and thereby reduce the chance of pipe string clogging. Material passing the first set of tines was channelled to the suction point by a second set of more closely spaced tines. The first nodules arrived aboard the mining vessel on July 30, 1970. Pumping rates between 10 and 60 tons/hour were successfully achieved during this PPMT which was the first to have successfully demonstrated the feasibility of nodule mining using air-lift and a passively towed collector, at least from the relatively shoal waters of the Blake Plateau.

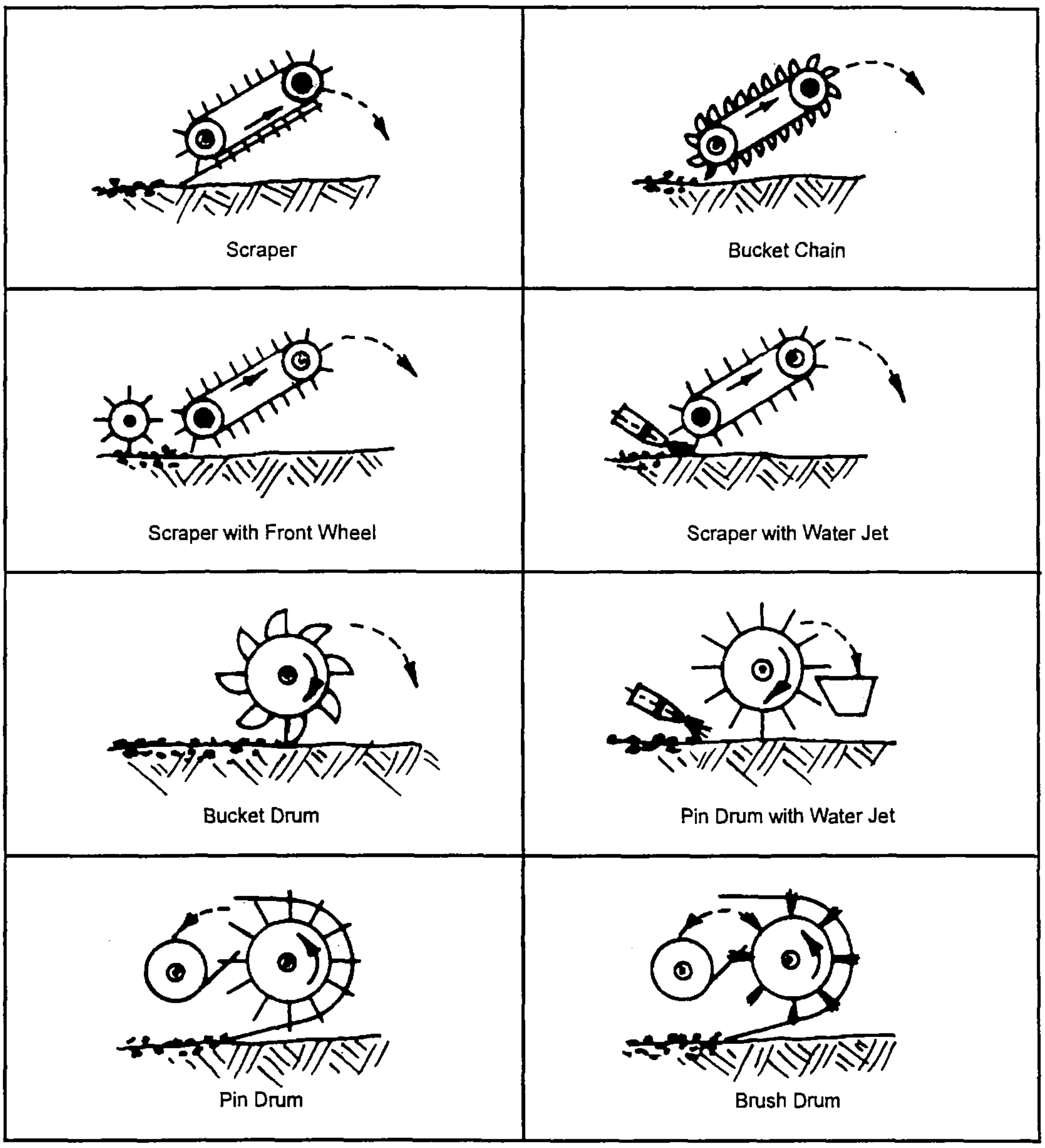

Figure 2.1.4 Various nodule pick-up principles. (from Thiel *et al.* 1991 redrawn from Bath 1989)

2.1.2.3.2. Hydraulic and Hybrid Collectors

In order to address the problems of efficiently collecting nodules over a wide area and providing a steady supply of ore to the end of the pipe string while allowing for seabed irregularities and patchiness of nodule distribution, the collector would have to be capable of harvesting nodules over a wide swath, perhaps up to 20m across. At the same time, the amount of sediment collected with the nodules should be minimised, both from an economic and environmental perspective. Nodules are also often partly immersed in the sticky sediment and require considerable force to be extracted. A possible solution to these problems is offered by a collector design that utilises hydraulic principles. In such designs, ambient sea water is pumped through jets directed so as to loosen nodules from the sediment, force them up into the collector, and rinse off much of the extraneous sediment still adhering to the nodules before they enter the pipe string. Generally, hydraulic approaches combined with various mechanical methods are described as hybrid systems (Amann 1990).

Several versions of hydraulic and hybrid collectors have been utilised during at-sea proof-of-concept test campaigns starting as early as 1977. The Ocean Management Inc. (OMI) consortium conducted the first successful PPMT at abyssal depths with a recovery of nodules from over 5000m south-east of Hawaii on March 28, 1978 (Fellerer 1980). A variety of passive collectors and both the air-lift and hydraulic pumping methods were successfully tested. In November of 1978, the OMA consortium also succeeded in recovering nodules from 4500m depths using a passive collector and an air-lift system. Less successful PPMT campaigns were conducted by the Kennecott (KCon) and Ocean Minerals Company (OMCO) consortia, with the later firm testing a massive self-propelled collector system utilising Archimedes screws (Welling 1981).

2.1.3. Potential Effects of Mining

All currently envisioned methods of collecting polymetallic nodules from the abyssal sea floor (see Section 2.1.2 above) will be accompanied by certain levels of environmental impact. The scope and severity of these impacts may vary according to the mining location and the design and operational mode of the mining system eventually utilised. At present, however, the full extent of such potential impacts remains unknown, poorly understood, and not accurately predictable.

A variety of potential environmental effects of polymetallic nodule mining have been identified (Thiel *et al.* 1991). Some of these have been shown not to have any deleterious effects or have been demonstrated to have very minor or otherwise acceptable impacts without the danger of producing serious long-term, wide-ranging, and undesirable consequences. Nevertheless, certain potential problem areas remain, particularly with respect to plume generation resulting directly from the mining process at the sea floor and from discharge of water containing sediment and nodule fines from the mining platform (Amos *et al.* 1972). Both processes have been insufficiently studied to be able to make adequate assessments of environmental consequences.

Direct impacts within the mining collector track can not be avoided and will have to be considered as a trade off if ocean mining is ever to be conducted. The abyssal benthic fauna residing upon and within (Thiel *et al.* 1993) the nodules will most certainly be lost in the mining areas from which nodules are harvested. The essentially permanent removal of nodules, which represent the hard substrate upon which these nodule epi- and infauna depend, will alter the environment considerably and will prevent recolonisation by larval immigrants. The community to be expected in mined out regions after recolonisation has occurred will be less divers and closely resemble that of nearby nodule-free areas.

Sediment dwelling organisms inhabiting the upper few centimetres of the sea floor between and beneath the nodules are also vulnerable, although a small percentage may survive the encounter with the mining machinery depending upon its mode of operation. It is rather obvious that active collector systems depending upon self-propulsion are more likely to penetrate deeper into the sediment and consequently do greater damage than passively towed systems that glide, roll, or may even be suspended over the sea floor during their operation. Thus, a self-propelled collector using the Archimedean screw principle and, to a lesser extent, one using a caterpillar track can be expected to have proportionately greater impacts than a towed collector gliding over the bottom on sled-like runners. The compression and depth of penetration into the sediment and accompanying degree of disturbance and sediment suspension would undoubtedly differ considerably among such systems.

Another variable which remains unknown is the precision with which a collector is capable of removing nodules without disturbing any more sediment than absolutely necessary. Mining industry scientists and engineers are well aware that there is no profit in introducing massive quantities of sediment into the pipe string along with the nodule ore. Most mining systems under consideration attempt to eliminate unwanted sediment before vertical transport by rinsing the nodules prior to introduction into the pipe string. However, anyone who has experienced collection of nodules at sea with such devices as dredges, grab samplers, and free-fall grabs knows that sediment adheres securely to the rough nodule surfaces and is not fully washed off even after being subjected to several kilometres of rinsing in the water column during gear recovery. When relatively undisturbed samples of a nodule-covered sea floor are retrieved with devices such as box corers, it is always instructive to note the considerable force needed to withdraw the nodules from their sediment matrix and the cohesive nature of the sediment itself. Most collector systems anticipate a scraping action during nodule pick-up, at least down as far as the lower portion of the nodules lie within the sediment which is usually a few centimetres. The degree of precision with which this scraping penetration can be controlled, thereby reducing unnecessary disturbance of sediment below the nodule bases, will be a key variable upon which calculations of volume of sediment disturbed will be based. Unfortunately, the major portion of the abyssal benthic species assemblage resides within the same upper few centimetres of sediment (Thiel 1983, Gage and Tyler 1991) and will therefore be negatively impacted by even the most precisely controlled mining collector. Nevertheless, from the perspective of limiting the development of a benthic plume, the objective should be to minimise as far as possible the depth of collector penetration and consequent sediment disturbance and mobilisation.

Despite the extensive damage to be expected within the collector tracks, no concurrent loss of species is anticipated to result from mining the sea floor. The destruction of a certain percentage of a given population can be assumed, but most abyssal macro- and megabenthic organisms are expected to be widely distributed and are unlikely to be totally extirpated as a result of deep-sea mining test activities (Foell *et al.* 1991, Thiel 1992). The situation is less clear with some of the smaller and less conspicuous biota whose range of distribution is poorly known. Many of these organisms are new to science and species are continuing to be discovered and described based on material collected during almost any expedition to previously unexplored or relatively infrequently visited regions of the oceans. The situation could eventually parallel what is happening to the diversity of tropical rain forests where species are being lost on a daily basis before science has even had an opportunity to give them a name. One aspect of nodule mining which will greatly assist in maintaining species diversity is that mining will be restricted to areas which meet all the necessary requirements from economic, logistical, topographic, and geological perspectives. Areas suitable for mining are relatively restricted in size. Mining localities seem to fall into the range of 20 to several hundreds of square kilometres and are separated from each other by regions where various prevailing conditions (nodule abundance, size, ore grade, topography, obstructions, etc.) do not favour or prohibit exploitation (Usher *et al.* 1987, Thiel *et al.* 1991). Such areas will therefore not be directly impacted and should be able to serve as reservoirs from which nearby mined-out areas may ultimately be re-colonised, at least by those organisms not requiring hard substrata.

Although such reasoning may result in a conception of ocean mining and its environmental consequences being confined to relatively small and widely separated areas, it is important to remember that to this point, only direct and near-field impacts, that is, impacts resulting from the passage of a collector and confined either to the collector track itself or to nearby (within several metres) regions, have been discussed. The by far less understood and potentially more dangerous consequences of nodule mining have to do with far-field impacts resulting from current driven redeposition of particles from the plumes. Although the massive size of existing ocean mining license areas (most are larger than many nations of the world; a typical 150,000km^2 ocean mining claim is about half the size of Italy) may still appear small when compared to the apparent vastness of the oceans, it should be recalled that potentially deleterious impacts may not be limited to the mining areas themselves. The World Ocean represents a vast continuum and, unlike the neatly executed lines drawn by man to separate territories and create borders based on artificial legal constructs, no physical boundaries separate ocean mining license areas, either from each other or from adjoining ocean space. It is possible to state with some confidence that the portion of sea floor that will be directly affected corresponds to the areas actually mined, but the portion that will be indirectly affected is considerably more difficult to predict and currently remains unknown (Thiel *et al.* 1991).

Indirect effects of mining primarily derive from the various plumes that will be generated in conjunction with mining operations. The discharge of unwanted sediment at or in proximity to the sea floor is anticipated to produce near-bottom plumes which may vary considerably in their three-dimensional extent in accordance with the nodule-sediment separation and sediment discharge methods designed and employed. Additional variability is introduced by sediment characteristics, physical properties of the sea floor, and bottom current patterns in the specific mining locale which can generate widely differing environmental impacts even when the same collector system is utilised. Efforts to model mining plumes are hampered by lack of sufficient ground truth data and other relevant information.

In order to achieve a better conceptual framework for the massive material transport associated with a nodule mining operation, an example (Table 2.1.1) based on rather conservative figures commonly reported by industry may be enlightening (Thiel *et al.* 1991). A typical 1.5 million metric ton (dry) per annum nodule mining operation requires recovery of 5000 metric tons of nodules on a daily basis. This is accomplished by removal of nodules and sediment matrix to a depth of 2.5cm from approximately 1km^2 per day which corresponds to about 25,000m^3 of material removed (the variable portion of nodule volume emerging above the sediment surface is ignored for purposes of this computation). A typical mine site may exhibit a nodule coverage of about 20% and if all the sediment could be successfully rinsed off before the ore is introduced into the pipe string, then about 20,000m^3 of sediment might be mobilised and suspended in a near-bottom plume which would drift horizontally with a speed and direction dictated by ambient currents which in most cases are unknown or poorly understood. Suspended particulate matter would resettle at various, as yet unknown, distances from the point of plume generation in accordance with the size, shape, flocculation properties, and sinking rates of individual particles. Such a volume of sediment corresponds to what could be transported with boxcars of a 20km-long freight train. This same volume would be moved on a daily basis for 300 or more days per year.

Table 2.1.1 Conservative sample calculation of sediment volume mobilized during the massive material transport associated with nodule mining operations (modified from Thiel *et al.* 1991).

Assumptions:

1. average nodule abundance suitable for commercial recovery of 10kg $\cdot$ m^{-2} wet mass (corresponding to 7kg $\cdot$ m^{-2} dry mass)

2. annual tonnage requirement per mining operation of $2.15 \cdot 10^6$ wet metric tons (corresponding to $1.5 \cdot 10^6$ dry mt)

3. active mining for 300 days per annum

4. a daily requirement of 5,000 wet metric tons per mining operation

5. an overall nodule recovery efficiency (from the seafloor to the shore-side stock pile and including losses attributable to sweep efficiency (80%), collector pick-up efficiency (90%), abrasion in pipestring, transfer operations, etc.) of 70%

6. sediment specific gravity of about 2.65

7. average depth of nodules within sediment (half of the diameter) of only 2.5cm

8. depth of penetration into sediments by nodule collector during harvesting of 2.5cm

9. average volume ration of sediments to nodules in the upper 2.5cm of sediments of 4:1 (i.e., about 20% of the upper 2.5cm of seafloor constitutes nodules)

10. Sediment volumes calculated to be mobilized per mining operation:

on a daily basis	$2 \cdot 10^4\ m^3$
on an annual basis	$6 \cdot 10^6\ m^3$

In addition to not understanding or being able to model adequately the formation, configuration, movement, and redeposition of such massive volumes of suspended particulates in the form of a plume, the effects upon benthic organisms within the redeposition area of the plume are also unknown. It has been suggested that there is a possibility for wide-ranging deleterious effects, both from direct burial and asphyxiation of the benthos, and from release and mobilisation of various chemical ionic species which had previously been held immobile within the sediment. Areas affected may be many times larger than the area originally mined. However, it is expected from observations on particle aggregation and laboratory plume experimentation employing natural sediments from the Peru Basin in the Southeastern Pacific that much of the particulate matter will resettle soon after its release from the collecor system and blanket the newly generated mining tracts. Even if some individuals were to survive the severe impact of the collector vehicle within the tract, smothering by resedimentation and the devastation of the sediment surface, the general food source of sediment-living species, should barely allow further existence. The entire problem complex is ill understood and requires considerable further study on time and spatial scales sufficiently large to allow extrapolation to full-scale mining.

The situation is similar for the sediment plumes generated by discharging excess sediment, nodule fines, and bottom waters from the mining platform. The daily discharge requirements of a 1.5 million metric ton (dry) per annum nodule mining venture have been provisionally estimated to consist of 25,000m^3 of bottom waters containing 1600 metric tons of sediment and 250 metric tons of nodule fines (Ozturgut 1981). Such calculations may well be very conservative and actual discharge levels could again vary considerably in accordance with differing nodule type, mining conditions and sea floor characteristics. Depending upon the discharge depth selected, such plumes could negatively influence the plankton community, interfere with primary production, affect the chemistry of waters at and below the depth of discharge, and have numerous other potentially deleterious consequences. Test mining of the Red Sea metalliferous mud has demonstrated the desirability and feasibility of utilising sub-surface discharge (see Subchapter 2.3), an approach that should also receive greater consideration during future PPMTs and pilot mining operations (PMOs) focusing on nodule recovery.

2.1.4. Research Required to Evaluate the Effects of Mining

Perhaps because its initial developmental stages occurred during a period of increasing environmental awareness or because of the notoriety associated with the potential for a major new penetration of an area previously considered to be too remote and inaccessible for exploitation, the deep ocean mining industry has from the very outset demonstrated environmental consciousness to a greater extent than most other industrial endeavours during their infancy. Considerable funds from governmental agencies and industry have already been expended in making baseline assessments of areas likely to be impacted by nodule mining and in monitoring industrial PPMTs during the late 1970s. Support has been provided to various investigations addressing specific topics with relevance to potential ocean impacts such as benthic recolonisation, effects of discharge plume particulates upon near surface macrozooplankton including fish larvae, and studies of discharge characteristics of commercial mining operations. Additional funds have supported research addressing near bottom plume effects including small-scale laboratory and field experiments. Several large-scale field experiments have also been initiated and are continuing to provide valuable information.

Concerns had been voiced as early as in the 1960s about possible environmental impacts associated with harvesting nodules from the ocean floor. Even the earliest *in situ* tests of mining technology, including the 1970 suction dredge mining test on the Blake Plateau of the Atlantic Ocean and the 1972 CLB dredge test in the Pacific Ocean were monitored by scientists attempting to gain a better understanding of what the potential consequences of eventual large-scale commercial mining activities might be (Amos *et al.* 1972, 1973, Amos and Roels 1977, Roels *et al.* 1972, 1973, 1974).

As industrial interest in ocean mining continued to expand during the 1970s, efforts to identify and understand the potential environmental effects associated with exploiting the deep seabed increased as well. Environmental baseline conditions were assessed in the Sargasso Sea during the Bermuda Rise Study in 1972. The central North Atlantic Ocean gyre of the Sargasso Sea is in many ways similar to the oligotrophic North Pacific Ocean area where most commercial nodule mining interest had by then already been focused. Of particular interest was determining if surface discharge of bottom waters laden with sediment would stimulate phytoplankton productivity. A preliminary assessment concluded that impacts would probably be minimal, although this remained to be verified. Additional work to accumulate data on baseline conditions was also initiated 1972 in the Pacific Ocean aboard the German FS *VALDIVIA* which was at the time engaged in mineral exploration cruises within the CCFZ area.

In order to meet requirements of national environmental legislation, the United States in 1975 initiated a comprehensive research program designed to produce an environmental characterisation of the CCFZ region which had been targeted by the nascent ocean mining industry and to monitor the PPMTs planned by industry toward the end of the decade (Thiel *et al.* 1991). The programme was named the Deep Ocean Mining Environmental Study (DOMES) and, together with the Metalliferous Sediments of the Atlantis II Deep (MESEDA) programme which accompanied metalliferous mud mining development in the Red Sea (see Subchapter 2.3), was one of the first extensive environmental research programmes to be conducted in anticipation of the dawn of a major new industrial activity.

Despite all these efforts, several problem complexes have not been fully addressed and require further study, primarily because they need long-term and large-scale *in situ* monitoring of actual ocean mining operations. Opportunities to conduct such monitoring have not yet been available, but are expected to become so during the PMOs planned by industry prior to making a commercial level commitment. Additional work is also required, particularly accumulating detailed knowledge of physical, chemical, geological, and biological baseline conditions. A greater understanding of the communities that presently exist in the various ocean mining license areas and their interaction with the environment is equally important. Before environmental effects of full-scale ocean mining can be adequately assessed, the *status quo* of the water column and sea floor must be known in order to allow detection and measurement of any changes that may be attributable to exploitation operations. The deep seabed is to a large extent a vast *terra incognito* which is still just beginning to be explored and better understood. Since the anticipated industrial PMO activities continue to be delayed, environmental efforts to monitor such activities can presently be directed only at planning, organisation, technique development, and theoretical considerations associated with future monitoring efforts. There are, however, several other areas of research with relevance to ocean mining which should be immediately and fruitfully pursued.

One such area involves accurate and long-term measurements of ambient current conditions in the various mining license areas already identified. Better measurement and a fuller understanding of the cyclical nature of surface, mid-water and bottom currents are needed since these would be of considerable value to attempts to model the formation and fate of mining plumes and can significantly affect mining operations. Only a rough understanding of daily, seasonal, or annual cyclical changes in current direction and velocity as now available for a few sites in some of the license areas will not allow adequate modelling and prediction of the fate of sediment plumes associated with mining activities.

Another subject of interest is obtaining a better understanding of the sediment and sea floor topography in the already identified mining localities. Sediment properties and topographic characteristics of the seabed can certainly influence plume formation and may control the types of benthic communities present in a given region. From a mining engineering perspective, such information is vital to the design and operation of a collector which may encounter highly variable conditions in a specific mining site.

A subject that is already being pursued, but which should perhaps be more profitably addressed by focusing greater attention to the areas identified as future mining locales, is basic research on oceanic ecosystems and their reaction to environmental disturbances. In the present world-wide climate of tight research budgets, it would be advantageous to combine the needs of applied research with the interest that many ocean scientists have in conducting basic research. The abyssal regions of most of the oceans remain unexplored and relatively poorly understood. If basic research questions can be addressed equally well through studies performed in or near potential mining areas as through efforts directed at areas that are not expected to ever be mined, it would seem desirable to conduct such programmes in or near the mine sites so that the information and knowledge obtained is more directly applicable to applied research questions as well as to pursuit of basic science.

In this regard, the United States has developed the concept of stable reference areas (SRA) of two distinct types, namely, impact reference areas (IRA) and preservational reference areas (PRA). These two categories of reference areas are to be established in such a way that IRAs are situated in mined out areas to minimise inherent environmental differences and thereby permit statistically valid assessments of mining impacts, while PRAs are located sufficiently distant from mining sites so as to ensure that the biota are not affected by mining activities (National Research Council 1984). PRAs are also to be protected from other anthropogenic influences such as dumping of wastes in order to permit unbiased studies and allow comparisons to be made with results obtained from IRA sites. At least one PRA and IRA have been designated to date in the CCFZ (Greenwald and Hennigar 1988), with the PRA serving as the site of a Benthic Impact Experiment (BIE) designed to elucidate certain aspects of benthic plume burial effects upon the bottom fauna (Trueblood and Ozturgut 1992). In order to achieve full coverage of all potential world-wide nodule mining areas, it is anticipated that additional PRA and IRA sites will be designated, possibly at locations where experimentation has been or is being conducted or where prior research has already generated a significant body of information (Thiel *et al.* 1991). PRAs should be designated so as to cover more than one national claim in order to combine environmental study efforts internationally.

Small- and large-scale experiments on burial effects and recolonisation have been and are continuing to be conducted by several countries, usually in areas near or within their mining license application areas. A prime example of this type of study is the German DISCOL (for *dis*turbance and re*col*onization) project which was initiated in the South Pacific Peru Basin near the north-east corner of a mining claim in 1989 (Thiel 1991, Schriever 1995, Thiel *et al.* 1995, Schriever *et al.* 1997). Developed upon the results of prior German investigations during the MESEDA study in the Red Sea (see Subchapter 2.3) and the American DOMES project in the

North Pacific, DISCOL represents the first long-term, large-scale attempt to simulate somewhat an ocean mining environmental disturbance and examine its effects upon the abyssal benthic community. The first DISCOL cruise, during which a study site was selected, baseline samples were obtained, and a disturbance was created within a 11km^2 circular area of seabed at 4150m depths followed by an initial post-impact sampling series, took place in February-March 1989. Three subsequent cruises (September 1989, February 1992, and January-March 1996) obtained post-impact samples 6 months, 3 years, and 7 years after disturbance, respectively. Although DISCOL is a broad study that does not claim to mimic the effects of future ocean mining activities, it continues to provide basic insights on a number of questions relevant to ocean mining risk assessment. DISCOL is accompanyed by geological (Stackelberg 1996, Wiedecke and Weber 1996), geochemical (Koschinsky and Halbach 1995, Koschinsky *et al.* 1997, Drodt *et al.* 1997, König *et al.* 1997) and plume modelling (Zielke *et al.* 1995, Jankowski *et al.* 1996, Jankowski and Zielke 1997, Segschneider and Sündermann 1997) studies.

Other large-scale investigations have been conducted since 1991 as joint efforts by groups of scientists from the United States and the Russian Federation (Trueblood 1993, Trueblood and Ozturgut 1997), later joined by Japan (Kaneko *et al.* 1995, 1997, Fukushima 1995) and the InterOceanMetal (IOM) Organisation (an international consortium whose membership includes Bulgaria, Cuba, Czech and Slovak Republics, Poland and Russian Federation). This series of BIE studies attempted to quantify more precisely the effects of sediment redeposition generated by simulated ocean mining operations upon the abyssal benthos using a disturber device fitted with pumps to suck up and redistribute sediment over an area down current from the direct impact region. Baseline samples followed by post-disturbance samples were obtained, while sediment plume redeposition was to be monitored by use of sediment traps. To date, however, these investigations have been less than effective due to equipment failures, losses, and related problems. Efforts such as DISCOL and the BIE-type experiments are not dependent upon, and therefore do not have to wait for, the onset of industrial activity. These types of experiments continue to provide invaluable information, techniques, and experience that will be extremely useful once commercial PMO activities commence.

Based on the results of the various large-scale projects, their continuation should be discussed to develop more effective disturbance experiments. Although no final proposal can presently be offered, the authors have suggested that a visually controlled sediment blanketing study be conducted employing a manned submersible for enhanced control of all experimental changes and activities. It seems advisible to discuss such an experiment internationally together with the pioneer investors and to carry out this certainly costly undertaking cooperatively.

Development of techniques suitable to monitoring efforts, including plume traceability, both during the PMOs and on-going during commercial mining, are another requirement that should be immediately and more actively pursued during this period of relative quiescence. Any approaches that would result in lowering costs and reducing the need for field efforts such as utilisation of robot monitoring systems, automated sampling and sorting, among others, are highly desirable and may also have application in other fields or for other purposes.

Identification and baseline status assessment of suitable indicator organisms which would facilitate monitoring and allow early detection of environmental impacts that may not be otherwise obvious is yet another objective that should be immediately pursued. With relatively high species diversities, but low populations characteristic for much of the deep-sea ecosystem, it may be that indicator groups or higher taxa may have to be identified and substituted for individual indicator species. Since so little is known of abyssal ecology and the communities of organisms found in potential mining locales in the North and South Pacific and Indian Oceans, the current slowdown in industrial ocean mining development should be viewed as an opportunity to accumulate knowledge in these areas.

Since mining of polymetallic nodules was first to be developed, environmental studies on deep-sea nodule mining have assumed a lead position over the mining of other ores (see Subchapters 2.2 - 2.5). Much of what is hypothesised later in this report is based on the experience gained in many years of environmental research in the deep sea. Any further development above the status reached so far should be discussed and probably conducted with international cooperation.

2.2. Manganese Crusts

2.2.1. Resource Description

As was the case with nodules, the discovery of manganese crusts (also interchangeably referred to as iron-manganese crusts, encrustations, ferromanganese crusts or cobalt-rich crusts) may again be traced to the HMS *Challenger* Expedition of 1872-76 (Hein *et al.* 1986). Mention was already made in the Challenger Reports of dredged fragments broken from "huge concretionary masses especially near the top of volcanic seamounts" (Murray and Renard 1891), an obvious reference to encrustations of ferromanganese oxides that commonly occur at such locales. Scientific study of crust deposits, however, did not advance as rapidly as that of nodules. This was probably because commercial interest in crust mining did not develop to any significant extent until legal uncertainties about access to the highest grade nodule deposits on the abyssal plains being considered for international jurisdiction stimulated a renewed look at crusts as a potential resource. Cobalt-rich crust deposits were known to occur at very favourable ore grades, at great abundance levels, in shallower depths, and within the Exclusive Economic Zones (EEZs) of a number of nations, particularly in the Pacific Ocean (Claque *et al.* 1984, Commeau *et al.* 1984, Clark *et al.* 1984, Clark and Johnson 1986, Hein *et al.* 1986, Halbach and Manheim 1984, Halbach *et al.* 1987, 1989). Although manganese oxide encrustations may be found in all major oceanic regions, it is the Pacific Ocean with its numerous seamounts and extensive volcanism ("Ring of Fire") which appears to offer the greatest potential for eventual crust mining acitivity by virtue of the higher abundance and quality of crust deposits it holds.

Manganese crusts may be categorised as another member of the same class of ore-forming minerals to which the abyssal polymetallic nodules (Subchapter 2.1) also belong. Indeed, crusts and certain types of nodules (herein called seamount nodules to distinguish them from abyssal nodules) often occur together at some localities, sharing many geochemical characteristics and modes of growth. Just as with abyssal nodules, the primary occurrence of metals in crusts and seamount nodules is in the form of metallic oxides which, together with the metallic sulphides that constitute the subject of later chapters, represent the two largest classes of ore-forming minerals on the planet.

Unlike abyssal polymetallic nodules, however, these pavement-like encrustations tend to accumulate on submarine rock outcroppings such as those on the summits, plateaus, terraces and flanks of seamounts, ridges and guyots, as well as on submarine slopes of islands, sunken atolls, underwater volcanoes, and other abyssal elevations, particularly where the slope of the seafloor and the current regime precludes accumulation of sediment (Manheim *et al.* 1983, Pratt and McFarlin 1966). The water currents are also the primary supplier of metal ions for crust accretion which occurs predominantly through the hydrogenetic processes previously described for nodules. Seamount nodules and crusts may occur together in areas where rock outcroppings are interspersed with small, sediment filled basins, and crusts may even form over indurated sediment or talus substrate layers (Moorby *et al.* 1984). The metallic composition of such sediment-based encrustations, however, reflects their hydrogenetic origin since they are characterised by lower nickel (Ni), copper (Cu), and zinc (Zn) values and elevated iron (Fe) and cobalt (Co) similar to the more typical crust occurrences on basalt and other igneous rock substrates. These surfaces provide the necessary nuclei upon which accretion will most commonly commence.

Some exceptions to this generalised view occur on sedimented terraces and at the base of slopes where crust material may have been aggregated after fragmenting from its substrate host and sliding downslope. Such processes of fragmentation may also result in the formation of nuclei suitable for growth of hydrogenetic nodules. Mixed deposits of crust slabs and nodules are commonly found at the base of high relief areas.

Ferromanganese crusts are often referred to as cobalt-rich crusts since Co is enriched to a level of two to five or more times its typical concentration in abyssal nodules. At current prices, Co probably represents the dominant value metal contained in the crust deposits, although other metals such as vanadium (V), molybdenum (Mo) and platinum (Pt) may also make significant contributions to ore value. As was the case with abyssal nodules, crusts are, however, also subject to considerable local variability in ore grade. Some of the known factors that influence this variability include water depth, substrate type, geographic location (latitude), current regime, and the geological history in terms of tectonic movement of the substrate material.

Assays of the Co contents of encrustations occurring in the Central Pacific Ocean region have shown levels of this metal to be in the range of 0.5 to 2% (although the averages lie closer to 0.6-0.7%), well above the levels commonly found in nodules. Cobalt concentrations appear to be inversely related to water depth (Clark and Johnson 1986) with higher concentrations occurring above 2000m. In contrast, Ni and Cu levels appear to vary directly with depth, with higher concentrations found below 4000m.

Platinum concentrations in crust occurrences also exhibit high levels ranging from 0.2 to 1.2 ppm (average about 0.5 ppm) (Halbach *et al.* 1988). Even higher Co and Pt concentrations have been reported in the literature, particularly in the outermost oxide zone of crust samples (Le Suave *et al.* 1989). In contrast, results of studies on the encrustations found in the Central North Pacific area (Halbach *et al.* 1987) have shown Pt enrichment to occur in the lower or older crust layers. Other value metals associated with abyssal nodules, particularly Ni and Cu, display concentrations in crusts and seamount nodules to be at levels somewhat below what would be considered desirable for commercial deposits.

Crust thickness on suitable substrate material varies from mere stains of less than 1mm to thick encrustations of 15cm or more. Exceptionally thick crusts (20-40cm) have been reported in the literature (Friedrich and Schmitz-Wiechowski 1980), but are not the norm. An average thickness of about 3cm (most commonly crusts range between 2 to 5cm) is characteristic of the Central Pacific area deposits. The surface texture of crusts ranges from being quite smooth in areas with strong currents to knobbed with botryoidal proturbances in more protected or low current regimes. The internal structure is characterised by laminations of layers of varying chemical composition which undoubtedly reflect environmental conditions over the growth period of the encrustation. Growth rates for crusts may be similar to or somewhat faster than those of nodules with figures of a few millimetres per million years commonly reported (Clark *et al.* 1986).

Oceanographic cruises specifically focused on surveying crust deposits are relatively recent events. A series of cruises of the German research vessel *SONNE* to the mid-Pacific seamounts and Line Islands areas in 1981, 1983 and 1984 were the first expeditions dedicated to studying encrustations (Hein *et al.* 1986). Several cruises of the United States Geological Survey's RV *S.P. LEE* and by the University of Hawaii in the mid-1980s also focused on crust studies, particularly in the mid-Pacific mountains, Line Islands, Hawaiian EEZ, and Marshall Islands areas. Crust samples obtained prior to these pioneering efforts were usually acquired incidental.

Extensive crust deposits occur at lesser depths and in great abundance within various national EEZ areas, thereby remaining independent from the economic and legal constrictions currently expected to be associated with conducting mining for polymetallic nodules in the so-called "international deep seabed area" under the "common heritage of mankind" principle. It is expected that development of crust mining will progress as some of the same economic constraints that currently inhibit further development of nodule mining, particularly metal market conditions based on supply and demand factors, are alleviated.

2.2.2. Description of Mining Techniques

Mining the metallic oxide deposits that occur in the form of encrustations presents an entirely different technological challenge from that of nodule mining. Whereas commercially interesting nodule deposits are found in sedimented basins with relatively flat and benign topography at depths ranging from 4000 to 6000m, crust deposits with commercial potential are typically associated with more rugged topography and greater slopes, although at considerably lesser water depths (800 to 2400m). While nodules are found lying on top of the consolidated sediment or loosely embedded in the semi-liquid sediment layer, crusts are usually firmly attached to the rock substrate upon which they have precipitated and require *in situ* separation before the ore can be brought to the surface.

Technical developments of commercial mining systems for crust mining operations and processing methods have not been pursued as actively as that for nodules. In both cases, the processing flow is of interest from an ocean environment perspective only if it is partially or entirely conducted "at sea" and results in discharge generation. Some limited work has been done on crust processing including innovative suggestions that may utilise *in situ* beneficiation steps (Narita *et al.* 1990, Hirt 1992, Hall 1993). Nevertheless, the U.S. Minerals Management Service (MMS) of the Department of the Interior funded several programmes in the 1980s to assess the mineral potential of the U.S. EEZ (Smith *et al.* 1985). Several engineering studies were undertaken that addressed the preliminary design for cobalt crust mining and that pointed out gaps in the knowledge and understanding of crust deposits which needed to be filled before designs could be advanced sufficiently for proof-of-concept tests to be performed (Halkyard 1985, Latimer and Kaufman 1985). In addition to suggestions on possible mining techniques, development of improvements on survey technology were also supported under the MMS programme (Toth and Amerigian 1986).

Efficient and economic crust mining depends upon minimising the dilution effect caused by the nearly unavoidable harvesting of some substrate material along with the crust. Since the thickness of the encrustation may vary from less than 1cm to well over 15cm, the design of the mining device must take this into account in order to harvest efficiently and reduce as much as possible the fragmentation and uptake of diluting substrate. Additional data on small-scale crustal microtopography (over scales of a few centimetres to tens of meters) and variability in the physical nature of the deposit (particularly, the properties of the crust-substrate interface) are therefore essential for mining system design.

Another relative unknown is the seafloor topographic variability in areas of crust deposits. Encrustations in the more rugged, high relief areas may well be considered unminable, at least with first generation techniques. Considerable mapping and related assessment needs to be done in order to better define potential mine site locations and ensure that the mining devices are capable of handling terrain variability.

2.2.2.1. Continuous Line Bucket (CLB) Crust Mining System

Although the CLB system was originally designed and tested as a manganese nodule mining system, it could be applied to crust mining with relatively minor modification (Masuda 1985, 1987, and 1991). The primary advantage offered by a CLB system is its relative mechanical simplicity and lower cost of operation. Additional advantages include the essentially water- and sediment-free nature of the material harvested which may considerably alleviate potential environmental problems associated with surface or midwater discharge of bottom waters, sediment, fines, and other rejected materials (Masuda and Cruickshank 1995). The nature of the CLB system also permits greater flexibility in that its application is not as constrained by economics of scale considerations.

The main disadvantages associated with the use of a CLB system are its relative inefficiency and still unproven ability to discriminate between crust deposits and host rock. Despite recent improvements in the CLB technique, its application may be limited to regions where fragmented crust deposits are found (Masuda and Cruickshank 1995). Furthermore, the potential for loss, entanglement and other damage would appear to be increased in areas with relatively rough topography which characteristically harbour the most promising crust deposits.

2.2.2.2. Combined Mechanical Fragmentation/Hydraulic Lift Systems

Crust mining systems falling into this category again borrow heavily from engineering designs originally conceived for conducting nodule mining operations. After the United States failed to adopt the Law of the Sea Treaty in 1982 and proclaimed establishment of a U.S. EEZ in 1983, commercial interest in the mineral occurrences of the EEZ, especially within the Pacific Ocean, was renewed. In preparation for a proposed outer continental shelf lease sale of cobalt crust resources in the Hawaiian Archipelago, the Department of the Interior's MMS and the State of Hawaii jointly funded a number of studies addressing likely scenarios for future exploitation of the manganese crust resource. These included at least two studies that attempted to outline possible mining methods.

Halkyard (1985) proposed a likely crust mining scenario involving a bottom crawler (tracked) mining vehicle connected with a surface mining ship by a hydraulic lift pipe string and electrical umbilical. The mining device (miner) would be self-propelled and capable of travelling at about $20cm \cdot s^{-1}$. It would be about 8m wide, 13m long and 6m high and would weigh about 100 tons in air.

The miner would be equipped with articulating cutting devices (or alternatively with drag rippers, rotary cutting drums, impact chisels, or scrapers) which would permit fragmentation of the crust while minimising the harvest of the diluting substrate material. Immediately behind the cutter heads would be a group of pick-up mechanisms such as articulated hydraulic suction heads or mechanical scraper/rake type devices which would gather up to 95% of the fragmented crust material and transfer it to a gravity separator (to further reduce the quantity of diluting substrate) prior to introduction into the pipe string for transport to the surface vessel.

Such a miner could be actively steered (based on real-time data supplied by optical and sonar systems) around obstacles. It would generally follow a mining plan involving non-overlapping parallel tracks adjoining previous mining swaths laid out along isobaths. Changes in crawling speed could be used to accommodate variable deposit densities so that a fairly consistent rate of crust mining may be achieved.

The pipe string would carry a 10-20% slurry of fragmented crusts, substrate and bottom water. Compressed air injection would occur at about 1/3 of the water depth to supply the necessary lift force. Separation of ore from the slurry mixture would occur aboard the surface vessel from which fines and waste water would be returned to the ocean. A hypothetical 1 million wet tons per year operation would require the miner to be active for about 225 days. This allows sufficient days for down time due to weather, maintenance and repairs.

A somewhat more sophisticated crust mining system capable of sensing the crust thickness and adjusting the cutting depth accordingly was suggested by Latimer and Kaufman (1985). The sensing system would take advantage of the wide differences in sound propagation velocities characteristic of sea water, cobalt crusts, and typical crust substrates such as basalt and hyaloclastite (a highly fractured volcanic rock formed during undersea eruptions). These differences would allow a microprocessor to determine the depth to the crust-substrate interface (or crust thickness). The depth of penetration of the cutting mechanism would be adjusted in response to the apparent crust thickness thereby greatly reducing wear on the cutters and minimising dilution with substrate material.

2.2.2.3. Submarine Solution Mining System

A novel *in situ* crust mining system which attempts to adapt a terrestrial leach mining approach to removal and recovery of value metals from cobalt-rich crusts on the deep seafloor is under development at the University of Hawaii (Zaiger 1994). This method involves deployment at up to 2000m depths of a large (up to 40,000m^2) "containment and regulation cover" (CRC) consisting of a flexible membrane laminated over a wire rope grid. The membrane is bordered by a system of perimeter dike tubes that would be filled with a heavy fluid such as barite drilling mud to seal the CRC to the substrate, allow maintenance of negative pressure within the edge seal perimeter, minimise the dilution of the extraction fluid by ambient sea water, and minimise escape of leach liquids into the surrounding environment.

With the assistance of ocean-going tugs equipped with deep-sea winches, a CRC would be towed to the mine site on detachable pontoons in a double roll configuration, then detached, lowered to the seabed and unrolled. Spreading of the membrane across the seafloor is facilitated by inflating a spreading tube network within the CRC. This is also fitted with a gridwork of perforated discharge and suction tubes that connect with a submersed pump station via header tubes. The pump station is in turn connected to a mooring buoy floating on the ocean surface via neutrally buoyant hoses. Extraction solution would be pumped from a mining/processing vessel moored to the buoy into the perforated discharge tubes. After sufficient time to allow the leach process to proceed, the enriched solution would be sucked from beneath the membrane and pumped back aboard the mining platform for further processing and removal of value metals.

The mining system design foresees the simultaneous use of two or more CRCs in order to reduce down time for the surface vessels (Zaiger 1994). After leaching is completed and the enriched fluid has been removed from beneath the CRC, the mining vessel moves to another site previously covered by a CRC to initiate the extraction process. An auxiliary support vessel would be responsible for guiding the deployment and recovery of the membranes by the tugs, as well as for introduction and evacuation of sealing fluid, pumping up of the spreading tube network, towing and placement of the mooring buoy and submersible pump, and neutralisation of residual leach solution beneath the CRC once most of it has been withdrawn. Neutralisation of the seabed beneath the membrane prior to its removal would be accomplished through washing with a weak lime solution followed by a sea water rinse.

The concept of this *in situ* leach mining system offers several economic advantages over the other crust mining systems under discussion. The technique involved is relatively simple and based on terrestrial mining analogues. It does not require movement over the often highly irregular topography associated with seamount mine sites. Consequently, there is no need for obstacle avoidance and highly precise navigation systems. The value metals are removed in solution and there is no need for mechanical separation from the underlying substrate. Transport to the surface is more energy efficient since only a single phase (liquid) is involved. At-sea processing reduces massive material and distance transportation requirements. Potential environmental effects of sediment plumes associated with seabed mining and dewatering discharges are essentially eliminated.

On the other hand, solution mining also has its drawbacks. Potential mining areas are limited to relatively flat sites with benign topography and low permeability of the underlying substrate. Furthermore, the substrate material must be relatively non-reactive to prevent excessive leach solution consumption. Escape of this fluid into surrounding waters or dilution by inflow of sea water below the CRC may be an ongoing problem. Zaiger (1994) suggests that a remotely operated vehicle (ROV) may be used to inspect the integrity of the edge seal and repair small tears of the CRC. Larger holes must be repaired when the membrane has been recovered. Additionally, the number of ships to be employed in this mining method would result in tremendous costs: 1 mining ship, 2 ocean tugs, 1 auxiliary support vessel, 1 ROV, probably together with its mother ship.

Preliminary results have also indicated that a build-up of slime-like surface layer products on the crust resulting from the leaching reactions are a major rate-limiting factor and must be addressed. The entire solution mining approach remains an interesting, but unproven concept which requires considerable further development through laboratory and field testing.

2.2.3. Potential Effects of Mining

The most obvious and direct influence of crust mining would occur within the mining area itself. In contrast to nodule mining, sediment mobilisation may present somewhat lesser potential for widespread deleterious influence since most crusts are attached to other hard substrate materials and located in less heavily sedimented areas. There would appear to be little chance for generation of massive sediment plumes, even in the case of removal of crusts from indurated sediment. Due to their average greater density and particle size, those substrate materials that are disturbed and rejected by the mining system would most likely be returned to an area proximal to the mining site and would be expected to have minimal effect upon areas further removed.

The loss of fauna, especially the sessile forms, within the mining area would have to be accepted as a trade off. However, since the desired hard substrate material (crust) is in most cases attached to another, even harder substrate (often basalt or related igneous rocks) which would not be removed, there should be sufficient suitable localities remaining for recolonization by sessile biota requiring hard substrate for attachment.

Depending upon the mining method selected, transport of ore through the water column to a surface mining vessel would also have a potential for possibly undesirable effects. While it is likely that an open vertical transport system such as that associated with the CLB approach may provide relatively clean ore when a bucket arrives at the surface, the "rinsing" process or separation of fines, adhering sediment, and other debris will have already taken place at various intermediate depths during the long hauling and recovery procedure. Although such "rinsing" cannot be avoided, it is hoped to occur primarily at near-bottom depths which is ideally where discharges generated by hydraulic mining systems would also be found. However, this general expectation may not be correct. Following the retrieval of a multiple corer, a rather simple structure in its sediment penetrating parts, video imaging demonstrated that adhering sediment was still being washed off in surface waters after recovery from a depth range of more than 2000m.

Hydraulic mining systems would convey crushed crust in a bottom water matrix to the mining vessel through a closed vertical pipe string much as envisioned for nodule mining. There would appear to be little danger to the environment from such passage unless there is a catastrophic failure which itself would trigger a shut-down of the operation and limit any damage. However, the need to separate the slurry aboard the mining vessel would result in the generation of a discharge containing sediment, crust fines, substrate material, cold bottom water, and biogenic debris which would be returned to the ocean at some lower depth yet to be determined. Both scenarios would introduce rejected material somewhere in the water column.

Solution mining should produce no dewatering discharges, although a need for some effluent discharge may be associated with the at-sea processing method. Escape of some leach or enriched solution from beneath the CRC into the surrounding waters could be anticipated. A possibly more serious consequence may accompany a catastrophic containment membrane failure in which the pregnant leach solution is introduced into ambient bottom waters. Although the bulk of the metals (Fe, Mn, Cu, Ni, Co) released from crust may not bioaccumulate (Zaiger 1994), there is a potential for trace elements contained in crusts to be released with possibly deleterious effects.

A somewhat significant difference between nodule and crust mining discharge elimination is the depth at which it may occur. Based on work conducted in the Red Sea (see Subchapter 2.3) on metalliferous mud mining, disposal depths for discharge were recommended to be below the region of active vertical migration, ideally below 1000-1100m. Similar disposal depths may be recommended for nodule mining, but may not be possible for some instances of crust mining since the mining depth itself may already be less than 1000m, lying within the zone of active vertical migration. Under these circumstances the discharge must occur close to the seafloor.

Finally, as with nodule mining, the areal extents of sites likely to be economically suitable candidates for crust mining will be limited and should not give rise to excessive concerns over a potential for species extirpation. To date, there is no known crust-dwelling or dependent taxon comparable to the nodule infauna described from sediment-filled crevices and fissures of nodules occurring in the Peru Basin (Thiel *et al.* 1993). Moreover, topography plays an even greater role in crust mining than in its nodule mining counterpart since most crusts tend to occur in or near regions with relatively high bathymetric relief. Crust mining sites would probably be of smaller areal extent and more patchy in distribution than nodule mine sites. Extensive crust covered areas occur for which appropriate technology suitable for economic and selective removal of crustal material does not exist and would be uneconomic to develop. Such areas might be viewed as reservoirs of species from which recolonization of barren hard substrate created by removal of crusts and the associated epifauna could take place.

2.2.4. Research Required to Evaluate the Effects of Mining

If and when current concepts and models for conducting Co-rich crust mining operations move closer to becoming reality, the technical approaches being considered will also become better defined and will allow more realistic assessments of potential environmental consequences. At present, we can only address some general concerns that might arise with mining systems and procedures which have been suggested, but are far from being implemented.

Additional work is needed on describing and assessing physical, chemical, geological, and biological baseline conditions in potential crust mining areas, both on the seafloor and within the water column. Early detection of potentially serious environmental effects attributable to crust mining activities is dependent upon a thorough knowledge and understanding of existing or *status quo* conditions. Abnormal developments or changes may only be recognisable when comparisons to normal conditions are feasible. Since crust mining will most likely occur in proximity to oceanic islands and in shallower waters, interference with shallow water communities, commercially important species, and recreational activities may become severe. Baseline studies must be conducted while keeping other uses of the sea in mind.

The fauna associated with crusts and its ecological relationships to other oceanic systems remains poorly known and understood. There may be little danger of initiating species extinctions as a result of crust mining activities, since areas suitable for mining, independent from the method applied, are probably rare, and species in general have a wider distribution. As no mining schemes are known today, we recommend thorough consideration when mining claims are delineated. If we assume a seamount to exhibit favorable mining conditions on its top and down the flanks, the crust blanket might be totally mined out and its fauna extirpated. Hard substrate fauna and bacteria in the deep sea are poorly known, but there should be little danger of destroying the complete inventory of such an area. Certainly, investigations and evaluations must be conducted when the mining area becomes known. The general, larger area must be taken into account, i.e. other seamounts proximal and distal to the model seamount, to evaluate the possibilities for recolonisation.

Certainly one complex of impacts that must be further examined is the physical and chemical character of the discharge that can be expected with most crust mining systems, as well as its likelihood of causing environmental damage. The potential for generation or mobilisation of fines and other materials with possible toxic effects should be probed in both qualitative and quantitative terms. Unlike most nodule occurrences, many potential crust mining localities are in proximity to inhabited coasts and islands, and occur at considerably shoaler depths. De-watering discharges and tailings disposal under such conditions becomes more critical since the likelihood of interference with other human uses of the sea is considerably enhanced.

Since no pre-pilot mining tests (PPMT) focused on manganese crust mining have taken place to date, meeting the long-term and large-scale *in situ* monitoring requirements has also not been possible. It would seem necessary and prudent to accompany any planned at-sea test of crust mining technology with an environmental monitoring programme so that the required data can be obtained. Future PPMT and PMO activities must be viewed as opportunities to observe and study impacts on the seafloor and within the water column. PMO impacts particularly must be intensely investigated so as to allow extrapolation to full scale mining. However, it would also be useful to conduct experimental studies as precautionary activities to allow early identification and conceptualisation of potential impacts. Submersible diving and observations on crusts, its underlying substrates, crust fragmentation, and sediment disturbances may provide some initial limits to potential problems associated with crust mining.

There are other environmental concerns if currently contemplated *in situ* benefication or preliminary processing methods are employed (Hirt 1992, Hall 1993, Zaigler 1994). It is difficult to believe that any leaching in the sea would not contaminate the sourrounding areas. Those who have seen photographs of rocky deep seafloors would regard it as nearly impossible to completely seal off the CRC at the perimeter of the covered area. Leaching and enriched solutions will most certainly leak out to different extents and contaminate the area, while heavier fluid may enter fissures and crevices. These effects must be investigated beforehand and could partly be addressed in laboratory studies. The same is true for the neutralisation fluids. The literature to date does not answer the question of where tailing fluids from the leaching or the neutralisation and on board processing will remain. It seems impossible to transport them back to the coast and therefore they may be disposed of at sea. This certainly needs thorough studies on their reaction with sea water and their effects on organisms. *In situ* leaching may create another problem. The process will not eliminate the total polymetallic crust layer, but is thought to leach out specific metal components and will dissolve certain other crust constituents. However, the remaining material will further cover the rocks below the crust. Described as slime-like surface layer products which remain after leaching, these need to be investigated on their stability against dissolution and the components which may be dissolved and transported by the water. Studies of this type should certainly be commenced in the laboratory, but *in situ* investigations employing a submersible should also be conceived.

2.3. Polymetallic Sulphides: Metalliferous Muds

2.3.1. Resource Description

Metalliferous muds (which may also be referred to as metalliferous sediments or hydrothermal-sedimentary deposits in various publications) are a type of unconsolidated hydrothermal mineralisation associated with heated, convectively circulating seawater near spreading centres and other volcanically active sites. As is the case with the massive consolidated polymetallic sulphides discussed in the next chapter, cracks and fissures in the basement rock (usually basalt) allow ambient seawater to penetrate several kilometres downward into proximity to a heat source such as a magma chamber or similar igneous intrusion. The superheated water then rises again and undergoes chemical reactions with the material through which it passes (Fig. 2.3.1). By such processes, the seawater becomes enriched with base and precious metals leached from the evaporite layers, sediment and rock beneath the sea floor. These metals may then be deposited out as metal oxides, sulphides, silicates, and carbonates in the covering layers of sediment forming metalliferous mud (Bignell 1978). Such metal-enriched deposits are generally found in heavily sedimented, basin-like regions of mid-oceanic ridges where tectonic activity has provided the prerequisites favouring their formation.

Most metalliferous sediment deposits occur at past and present centres of sea-floor spreading which suggests that their formation is tied to the generation of new ocean floor (Cronan 1976, 1977). Results from analyses of cores obtained during the Deep Sea Drilling Project (DSDP) have shown that basal metalliferous sediments on the East Pacific Rise, for example, are compositionally distinctive in that they have higher concentrations of iron (Fe), manganese (Mn), nickel (Ni), copper (Cu), lead (Pb), zinc (Zn), and several trace elements than do normal pelagic clays or crustal rocks. On average, their composition is more similar to recent East Pacific Rise-crest sediments which suggests that they may have been transported to their present position by sea floor spreading processes. The chemical composition, relatively thin layering, diffuse nature, and location under considerable layers of sediment overburden make the vast majority of metalliferous sediment deposits uninteresting from a commercial mining standpoint. Nevertheless, a better understanding of their formation and occurrence could greatly contribute to finding land-based deposit analogues of greater economic significance. To date, only the metalliferous mud deposit of the Atlantis II Deep in the Red Sea has demonstrated sufficient economic potential to be considered an ore reserve.

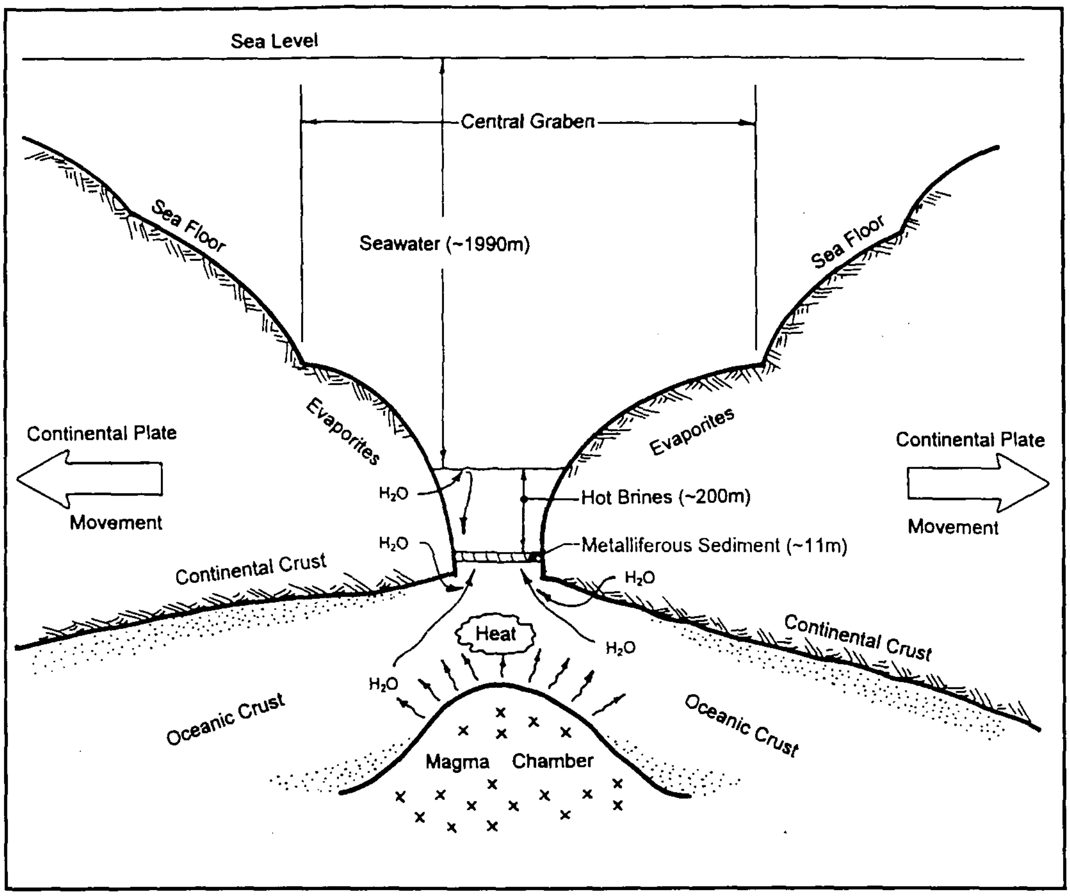

Figure 2.3.1: Generalised section through a Red Sea metalliferous brine deposit indicating its mode of formation (not to scale).

Although fossil metal-rich sediment deposits are well known from terrestrial occurrences and have been detected in cores from many areas of the World Ocean, the initial discovery of actively-forming metalliferous sediments occurred in the mid-1960s in the axial rift of the Red Sea. Research vessels of the United States, Great Britain, and Germany traversed the region between 1959 - 1965 during the International Indian Ocean Expedition and occupied several stations along its central axis. Scientists aboard these vessels detected near bottom temperature and salinity anomalies, and collected heavy metal-enriched sediments and hot brines from about 2000m depths of several deeps in the median valley (Fig. 2.3.2) (Swallow and Crease 1965, Miller *et al.* 1966, Degens and Ross 1969, Amann *et al.* 1973, Schoell and Hartmann 1978). Although it was known as early as in the 1880s that warm and unusually salty water characterised the Red Sea main water body at depths in excess of a few hundred meters, this finding had previously been explained in terms of the high predominance of evaporation over precipitation which theoretically resulted in a downward movement of warmer and saltier surface water layers (Degens and Ross 1969). The lowest temerpature of 21.5°C exists in the northern Red Sea during winter, and it is this water that sinks to the seafloor and constitutes the main water mass. It is currently known that the high salinity found in the hot brine layers in restricted basins of the central graben (of up to 7 times that of normal seawater) is due to salts being dissolved from evaporite layers by hydrothermal circulation. The presence of both a thick sediment cap and of highly saline brines may influence the process of metalliferous sediment formation through a combination of physical and chemical interactions with the rising hydrothermal fluids. As is characteristic of active terrestrial geothermal areas, the hydrothermal activity of the Atlantis II Deep is known to change over time resulting in corresponding alterations in the chemical and physical parameters of the brine and consequent fluctuations in the quantity and composition of precipitating metal compounds (Schoell and Hartmann 1978). Mean temperatures of the lower convection layer had increased by nearly 10°C since initial measurements were made in the mid-1960s.

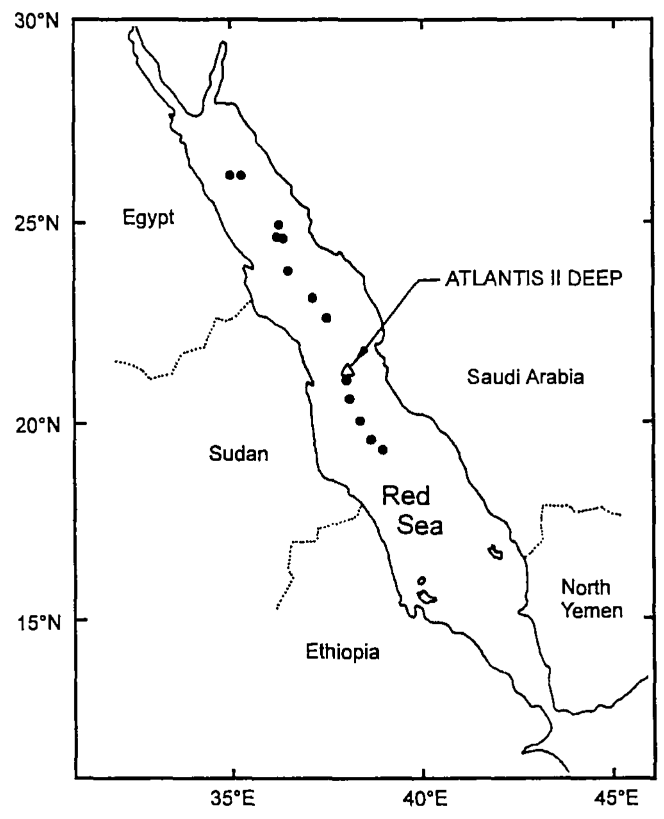

Figure 2.3.2 Red Sea metalliferous mud deposits.

Additional metalliferous sediment deposits, both ancient and still in the process of being formed, have since been discovered in other areas of the Red Sea and in many parts of the World Ocean. None, however, are as large as that of the Atlantis II Deep which rivals many terrestrial sulphide ore bodies in size. This deposit remains the most significant metalliferous sediment occurrence discovered to date and has been the subject of extensive studies of its commercial potential conducted by the German firm Preussag AG since 1969. In 1976, Preussag received a contract from the Saudi Sudanese Commission for the Exploitation of the Red Sea Resources (or simply Red Sea Commission) to conduct further exploration, investigate the feasibility of commercial mining, and carry out environmental studies (Thiel 1991). These studies were undertaken during the years 1977 - 1981; then they were cancelled by the Commission due to reduced funding availability.

In terms of value metals contained in the Atlantis II deposit, it has been estimated that more than 32 million tons of metals (including Zn, Cu, silver (Ag), gold (Au), cobalt (Co), cadmium (Cd), Pb, Ni, Mn, Fe and numerous trace elements) are located in the approximately 60km² area of the Deep, in large part confined to depths in excess of the 1990m isobath (Nawab 1984). At approximately this depth, a 200m thick, hot, highly saline and metal-rich water layer (with up to 25% salt and at temperatures exceeding 64°C) fills the steep-walled depression down to the sediment which at its surface is rather liquid.

The deeper, variegated, and partially laminated levels (average thickness of about 11m) of the metalliferous sediment are horizontally stratified and can be divided into distinct lithostratigraphic units based on the predominant mineral facies occurring therein (Bäcker 1980, Blissenbach and Nawab 1982, Nawab 1984, Mustafa *et al.* 1984). An upper, and economically uninteresting, amorphous-silicate (or AM) zone with a thickness of 2 - 4m overlies an approximately 2 - 4m thick upper sulphidic (SU_2) zone containing value metals in the form of sulphides. This layer is separated from a second or lower sulphidic (SU_1) zone (from 1 - 4m in thickness) which also contains economic metal sulphide concentrations by another uneconomic or barren layer, the 0.5 - 11m thick central oxide (or CO) zone. Beneath the lower sulphidic zone lies a 1.3 - 6m thick detrital-oxidic-pyritic (or DOP) zone which has some commercial promise since it too contains scattered sulphide facies that are of economic potential, but which was insufficiently sampled throughout its depth range to permit full reserve calculations. Only the uppermost meter of the DOP zone is included with the AM, SU_2, CO, and SU_1 zones (together referred to as Unit A) when statistical estimations of proven reserves were made. The still unproven portion of the DOP is referred to as Unit B and may yet considerably enhance the proven reserve estimate. Below the DOP zone lies the basement rock, generally consisting of geologically young basalt. Cores taken from the stratified mud assemblages that constitute the lowermost region of the hot brine deposit (Fig. 2.3.2) show distinctive colour banding ranging from bright white, yellow, orange, and red through grey-black colours reflecting the oxide and sulphide mineral facies and metal compounds occurring therein (Amann *et al.* 1973, Bäcker 1980).

The solid portion of the mud on average represents from 8 - 13% by unit volume with the remainder being brine. On a dry and salt-free basis, the mud contains 2 - 5% Zn, 0.3 - 0.9% Cu, and 0.00006 - 0.0001% (60 - 100ppm) Ag. It has been estimated that the total solid portion contained in Unit A of the Atlantis II Deep mud layers consists of about $100 \cdot 10^6$ tonnes. Of this amount, about $2 \cdot 10^6$ tonnes are Zn, $0.4 \cdot 10^6$ tonnes are Cu, $5.7 \cdot 10^3$ tonnes are Co, $3.6 \cdot 10^3$ tonnes are Ag, and 47 tonnes are Au (Guney *et al.* 1984). Given the currently estimated recovery efficiencies of mining and processing the ores, it is anticipated that a mining operation scaled to recovery of $6 \cdot 10^4$ tonnes of Zn on an annual basis (with proportionate recoveries of Cu, Ag, Au, and by-products such as Co, Cd, and gypsum) would be sustainable for at least 16 years (Mustafa *et al.* 1984). These estimates could be raised significantly if ore resources contained in Unit B are brought to proven reserve status and included in the recovery calculations. A mining operation extending over 20 or more years might then be viewed as being economically feasible.

2.3.2. Description of Mining Techniques

To date, only a single case of metalliferous mud mining has been pursued and developed through the proof-of-concept or technical feasibility stage which includes development of suitable mining technology and preliminary field testing of a scaled down version in the deep sea concluded in March - June 1979. The ore resource in question is that contained in the Atlantis II Deep of the central Red Sea graben and the mining system designed especially for harvesting the metal-rich mud was developed and tested by Preussag AG (Amann 1985, 1989, Nawab 1984, Mustafa and Amann 1980, Mustafa *et al.* 1984).

The Red Sea mud mining PPMT had three primary objectives:

- to establish the technical feasibility of mining the metalliferous sediment by diluting the mud with sea water and pumping the resultant mixture using a multi-stage, centrifugal deep water pump up a 12.5cm (5 inch) diameter pipestring to the mining vessel.

- to confirm shipboard concentration and enrichment of the complex mineral mixture using ultra-fine grain flotation in seawater as a viable beneficiation technique.

- to examine the effects of discharging the flotation tailings upon the marine environment and to minimise their environmental impact.

The first flow of metalliferous mud was achieved on May 1, 1979, and a total of nearly 16,000 tons of mud/brine/seawater mixture were eventually mined from four localities within the Atlantis II Deep during a cumulative 195 hours of active pumping spread over a five week period.

The technical approach (Fig. 2.3.3) used for the metalliferous sediment test mining was in many ways similar to that developed and tested for nodule mining, although the pick-up method at the lower end of the pipestring differed significantly to match the requirements of mining a fluidised ore material. A surface mining platform (in this case the *SEDCO 445*, the first dynamically positionable drillship to have been constructed world-wide) lowered the pipestring into the 2200m depths of the Atlantis II deposit. The pipestring terminated in a specially designed suction pickup head with a vertically vibrating screen. A pump module containing an electric motor and pumps was mounted to the pipestring more than 200m above the suction head so that intake of ambient sea water would occur above the level of the hot brine layers (Mustafa and Amann 1980).

The relatively slow vibrations of the suction head screen or sieve (frequency of 14-16 cycles/sec with an amplitude of 5mm) were intended to provide mechanical stimulation to the metal-rich consolidated sediments, but were in themselves considered to be insufficient to allow mobilisation and flow for transport. Sea water was directed downward through a pressure hose, and ejected around the suction head to facilitate dilution of the sediments to the optimum flow conditions of 40-70g $\cdot$ l^{-1} solid content which met the design parameters of the sediment pump.

The dilute nature of the ore material recovered aboard ship demanded some form of concentration and preliminary beneficiation prior to transfer to a shore-based processing facility in order to be economically viable. The already diluted mud was mixed with surface sea water so that the resulting slurry contained 1 part of mud and 3 parts of water. After conditioning and homogenising the mud mixture, a flotation technique was employed involving injection of air into a cell containing the diluted metalliferous sediment and special reagents which cause the metallic compounds to aggregate on the surface of rising air bubbles (Blissenbach and Nawab 1982). The resulting concentrated foam can then be skimmed off at the surface of the cell. Despite the extremely small average size of sediment particles (80% were less than 0.002mm in diameter) and possible interference induced by shipboard motion, the flotation method worked well and rivalled results of similar land-based operations. Recovery rates of 60 - 70% were

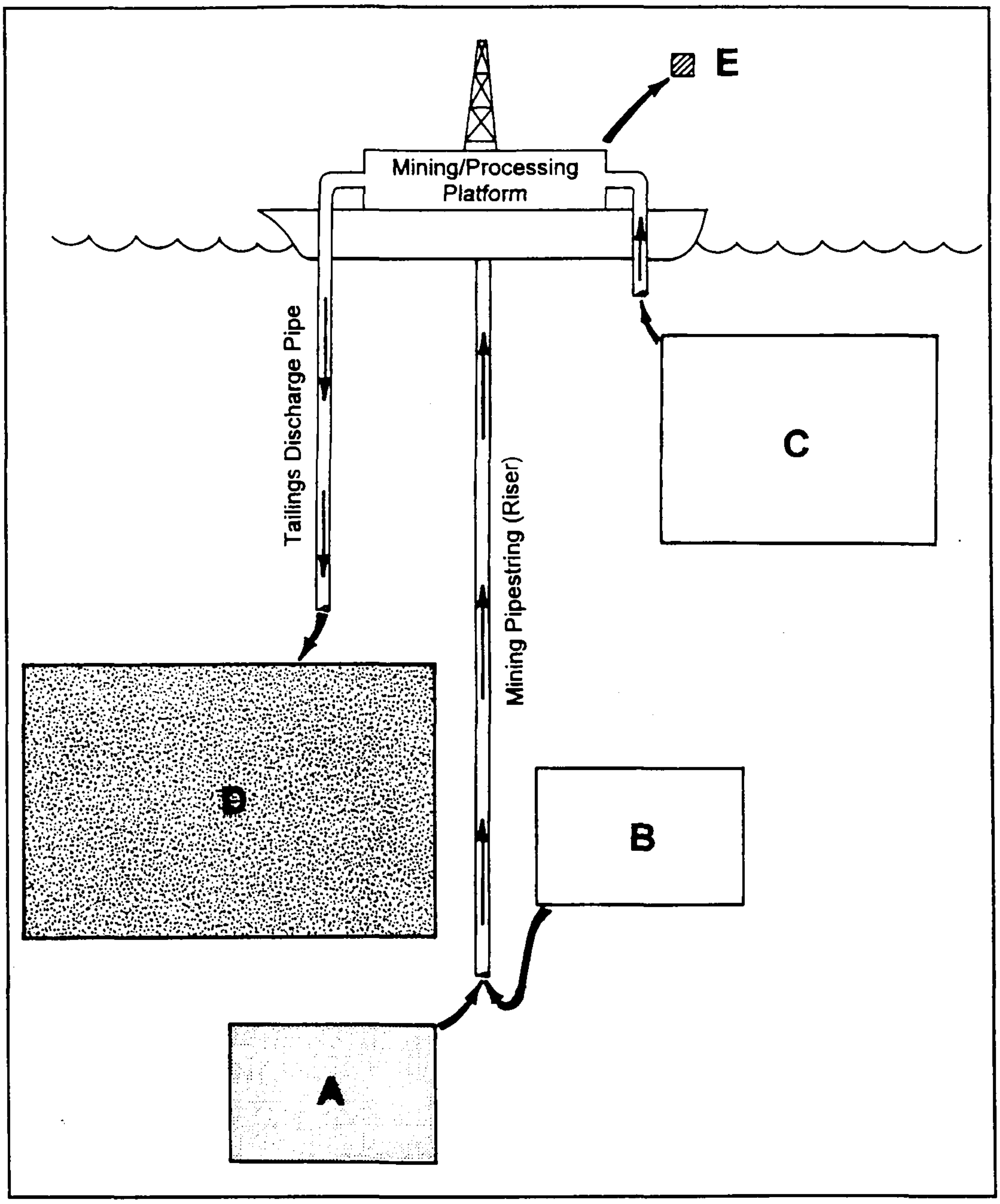

Figure 2.3.3 Daily intake and discharge requirements of a hypothetical metalliferous mud mining operation in the Red Sea. A = intake of *in situ* metalliferous sediment (brine/mud mixture containing 8-13% solids) at rate of $100 \cdot 10^3$ t/day. B = intake of seawater/brine dilutant mixture at rate of $100 \cdot 10^3$ t/day. C = intake of near surface seawater for dilution and processing at rate of $200 \cdot 10^3$ t/day. D = discharge of tailings (seawater, brine and about $10 \cdot 10^3$ t/day of fine particulates) at rate of $398 \cdot 10^3$ t/day. E = ore concentrate product for transfer to shore-based metallurgy plant at rate of $2 \cdot 10^3$ t/day. The size of the boxes in the drawing approximate the volume of materials involved.

achieved at concentration factors ranging on average from 8 to 10 (up to 15 was obtained after tentative optimisation of the process). Concentration of Zn exceeded 40% with correspondingly proportionate values for Cu and Ag.

The tailings generated by the concentration processes were returned to the sea via a 6-inch (15cm) diameter disposal pipe assembled from 6m long steel sewage pipe sections. This pipe was attached to the *SEDCO 445* bow anchor line with the anchor serving as a weight to maintain a vertical aspect. The over-all length of the disposal tailings pipe was 400m, a compromise between the minimal 800m length (although 1100m was considered even more preferable) suggested by ecological considerations and the need for precise mining vessel manoeuvrability and efficient use of expensive shiptime during a limited duration feasibility test. A 400m discharge depth - based on a compromise agreement between the oceanograpers conducting the environmental studies and the economists/mining engineers responsible for commercial aspects of the test - also avoided potential impacts in surface and near-surface waters and greatly increased the prospects for plume location and monitoring (Abu Gideiri 1984a, b, Thiel 1991).

Results of this limited scope and scaled down PPMT indicated that mud mining was technically feasible, although several areas were identified in which further technological improvement or refinement of knowledge and understanding were required. Since the eventual rate of return on investment has been estimated to be in the somewhat marginal 15% range (Nawab 1984), additional development of metalliferous mud mining in the Red Sea is dependent upon typically unpredictable metal market price fluctuations. Project maturation and commercial realisation are currently on hold pending development of a more favourable economic climate.

2.3.3. Potential Effects of Mining

The Red Sea represents a unique environment characterised by highly diverse faunal assemblages, particularly among the coral reefs along the coastline and in near surface waters throughout its basin. Benthic and pelagic diversity is less notable, but specially adapted to the environmental extremes that differentiate the Red Sea from other oceanic regions (Klausewitz 1989, 1994; Türkay 1986, 1996). The entire ecosystem is subjected to severe environmental conditions including intense irradiation from the sun, consistent and hot winds, limited inflow of freshwater, and restricted exchange with Gulf of Aden, Arabian Sea, and Indian Ocean waters over a shallow sill at the narrow southern end near Bab al Mandeb. These conditions bring about increases in temperature and salinity, while oxygen and nutrient levels are lower in comparison to other seas of the world (Abu Gideiri 1984a). Pollution influences such as oil and other wastes along the coasts and reefs were already noted in the 1960s, and will originate in part from high ship traffic along this waterway. Environmental conditions will undoubtedly worsen as the population of the region continues to grow and industrialise.

Concern over potentially detrimental environmental effects to the delicately balanced and relatively ill-understood Red Sea ecosystem associated with mining the metalliferous mud deposits of the central graben resulted in the Red Sea Commission requiring that environmental impact studies be performed as part of the exploration and development contract (Nawab 1984, Thiel 1991). The German Federal Ministry for Research and Technology (Bundesministerium für Forschung und Technologie or BMFT) and the Red Sea Commission jointly funded the MESEDA (*ME*talliferous *SED*iment *A*tlantis II Deep) programme during the years 1977 - 1981. Three extensive and combined exploration/environmental cruises were conducted using the RV *SONNE* (MESEDA I, Oct. 1977 - Jan. 1978) and the RV *VALDIVIA* (MESEDA II, March 1979 - June 1979, and MESEDA III, Oct. 1980 - March 1981). Environmental studies included characterisation of the physical, chemical, biological, and geological milieu and focused particularly upon the potential for harmful effects to biota from increased heavy metal loads in the water column and on the sea floor as a result of tailings disposal from the mining platform (Thiel *et al.* 1986, Thiel 1991). A comprehensive listing of publications based on these environmental studies is provided by Thiel (1991).

Unlike the situation with manganese nodule mining (Chapter 2.1), sea floor impacts as a direct consequence of mining metalliferous mud deposits are believed to be unlikely. With the exception of high temperature and pressure adapted anaerobic microbial species (Fiala *et al.* 1990), life does not appear to exist within the metalliferous mud and hot brines of the Atlantis II Deep. Mining activity would be confined to the approximately 60km^2 of the Deep, a relatively small areal extent in comparison to the huge surfaces required for commercially viable nodule mining. Although some ambient seawater would be withdrawn from above the hot brine layer and introduced as a dilutant to help mobilise the mud at the base of the brines, the affect on biota would likely be minimal since few animals appear to live in immediate proximity to the brine layer surface. Disturbances to the brine-seawater interface caused by inserting, moving, and withdrawing the pipestring with its attached suction head would also be expected to have minimal influence. Once taken up by the suction head, the mud is transported vertically through the pipestring to the mining vessel, again without any anticipated deleterious environmental consequences unless a catastrophic failure occurs. Even then, the system would not be able to continue to pump sediment and any environmental impact would be limited to loss of material already within the pipe. It is when the diluted metalliferous mud slurry arrives aboard the mining platform and requires concentration and preliminary processing to be an economically viable source of ore that the greatest potential for deleterious environmental impacts is encountered.

In order to be economically sustainable and meet the planned annual commercial production rate of $6 \cdot 10^4$ tons of Zn (and proportional amounts of other value metals and by-products), approximately $100 \cdot 10^3$ tons of original mud would be mined on a daily basis. This material would be fluidised and diluted with a like mass of bottom water before being pumped to the mining vessel. The $200 \cdot 10^3$ tons of diluted mud arriving aboard the mining platform would then be homogenised and further diluted with an additional $200 \cdot 10^3$ tons of near-surface water in preparation for the concentration and filtration processes. This on-board beneficiation procedure would produce about $2 \cdot 10^3$ tons of concentrated metal filtrate for shipment to the shore-based processing plant, leaving $398 \cdot 10^3$ tons of rejected slurry (containing $98 \cdot 10^3$ tons of original mud) to be discharged as tailings. Since about 10% (range: 8% - 13%) of the original mud is solid material in the form of particulates, this means that roughly $10 \cdot 10^3$ tons of particulate matter would be discharged on a daily basis, an amount equivalent to the daily particulate load discharged by the Rhine River into the North Sea (Thiel *et al.* 1986, Thiel 1991).

An analysis of the physical and chemical characteristics of tailings generated during the Red Sea PPMT has shown that, on average, they have a density of $1.1 g \cdot cm^{-3}$, a temperature of $35^{\circ}C$, a particulate content of $25 g \cdot l^{-1}$, and contain dissolved salts equal to $150 g \cdot l^{-1}$ (Abu Gideiri 1984, Thiel *et al.* 1986). With about 60% of the particulates having a grain size of less than 0.002mm and containing trace levels of such elements as Pb, arsenic (As), vanadium (V), Co, Ni, antimony (Sb), Ag, Cd, and mercury (Hg), it is obvious why the fate of the discharged tailings was the primary environmental concern of the scientists charged with conducting the mining risk assessments.

Limited toxicological testing has demonstrated that tailings at concentrations above a certain threshold would be hazardous to biota (Karbe *et al.* 1981, Abu Gideiri 1984a). Short duration experiments utilising standard test organisms indicated that this threshold is approximately 10mg solids/litre. When a typical, if somewhat arbitrary, safety factor of 1000 (or 0.1%) is applied to exclude possible chronic effects, this would indicate that a concentration above 0.01mg solids/litre would be unacceptable (Karbe *et al.* 1981).

A typical approach to eliminating such a massive volume of waste material containing partially toxic suspended particulates might have been to simply discharge the tailings overboard into the surface waters of the Red Sea in the hope that immediate mixing with ambient seawater would reduce solid concentrations to below critical levels. Surface disposal was, however, not deemed to be acceptable to the Atlantis II Deep project (see Abu Gideiri 1984a). Concerns had been voiced by the oceanographers, warning of possible lethal and sub-lethal effects on the plankton community including a potential for food chain biomagnification which could result in an increase of concentration levels of toxic materials in fishes and ultimately affect human health. Similarly, there was a concern that near-surface discharge plumes could form and spread laterally more rapidly than they would sink. Driven by currents and wind into highly productive near-shore waters, these plumes could impact the delicate coral reef environment and reefs. Other potential influences of surface discharge were also foreseen such as: (i) fine suspended matter clogging the filter-feeding apparatus of zooplankton, (ii) particle adhesion to the outer surface of plankton reducing their floatation capabilities, (iii) increases in suspended matter causing decreased light penetration and subsequent reduced photosynthetic activity in the euphotic zone, (iv) changes caused by increases in salinity and temperature, and decreases in pH, (v) decreases in oxygen availability due to oxidation of sulphide materials in the tailings, and (vi) disturbances in the levels of essential nutrients like silicates and phosphates (Abu Gideiri 1984a, b). Such considerations determined the need for sub-surface discharge in the case of Red Sea mud mining.

Biological studies of Red Sea plankton, nekton, and benthos had shown that there was a low standing stock and limited productivity in most oceanic pelagic and benthic locations (Thiel 1979, Weikert 1982). Indeed, most regions of the Red Sea may be classified as oligotrophic and resemble the central gyre regions of the world oceans in this regard. Exceptions to this generalisation occur only in narrow zones around the Sinai Peninsula in the north, in the transition zone between Red Sea and Indian Ocean waters in the south, and in the highly diverse and productive shallow coral reef areas along the periphery (Thiel *et al.* 1985).

The Red Sea is a uniquely two-layered environment with sharp temperature and salinity gradients between 75 - 180m separating an upper shallow zone of very warm surface waters (temperature of 27.7°C and salinity of 39.73‰ in Nov. 1977) from the isothermal and isohaline deep waters (temperature of 21.7°C and salinity of 40.67‰). The major portion of zooplankton diel vertical migration activity occurs above 750m, although some species penetrate to depths of up to 1100m. Below 1100m, no significant vertical migrations could be detected due to zooplankton depletion. Furthermore, if there were migrations at these depths, they could be considered insignificant and unlikely to materially affect mass transport (Weikert 1982).

The risk assessment studies were also concerned with the formation, behaviour, and ultimate fate of any discharge plumes associated with introduction of tailings into the water column. From a mining perspective, returning the tailings directly into the Atlantis II Deep was not desirable since, over time, this could dilute the deposit and reduce the average ore grades. On the other hand, it was considered desirable from an ecological standpoint to confine deposition of tailings to the central graben with its low levels of benthic life and minimal biological activity. The discharge at 400m depth was a compromise between surface and great depth release of the tailings, a compromise made between economists and ecologists.

Two models of possible plume dispersal were proposed: (i) a gravity flow model and (ii) a momentum-jet-flow model. Under the gravity flow model which assumed little or no dispersal of tailings and a lack of eddy development as the material sinks through the unstratified waters, the tailings were expected to fall to the seabed nearly undisturbed with plume development occurring only in proximity to the brine pycnocline and to the sea floor. The momentum-jet-flow model assumed development of turbulent flow with plume creation about 100m below the discharge point. This plume would attain a horizontal length of 2000m after only two hours and its final distribution would be determined by horizontal water currents. Particles would sink out of the plume and fall to the bottom in accordance with their settling characteristics. Since currents may flow in different directions at various water levels, the plume would not be expected to be stable in location and dimension, but would be altered and could shift its direction of movement by 180° within the central trench system. Even under stable conditions and assuming unidirectional transport, the plume would be massive, extending over 1500km^2 with sedimentation of particles over a similar sized area.

Observations of the PPMT tailings discharge at 400m depths were conducted using echosounding profiles and turbidity meter readings. A portion of the tailings dropped rapidly to the 800-1200m levels, more than anticipated by modelling, due to the higher density of the highly saline brines and pump pressure from aboard the mining vessel (Mustafa and Amann 1980), but there was evidence of plume development at 700-800m where the highest concentration of particulate matter was found (Thiel *et al.* 1986). No data were obtained on the fate of the dissolved components of the tailings.

Although some field observations and measurements of tailings discharge using iridium as a tracer during the PPMT supported several aspects of the momentum-jet-flow model, behaviour not predicted by the model was also detected. It is obvious that the models so far developed do not fit well with field observations of actual events. Furthermore, neither model makes adequate predictions on plume behaviour and dispersal after formation. Mean current transport in the Atlantis II Deep area is northward and, indeed, iridium-labelled sediment was recovered from up to 90km north and only 12km south of the mining area, but the number of samples was too small for a full plume dispersal evaluation. Even with the intermittent and limited discharge of only 12,000m^3 during the PPMT, these results already indicate a larger dispersal and sedimentation area than was predicted under the momentum-jet-flow model. Furthermore, plume formation, dispersal, and sedimentation, as derived from the model, show differing behaviour when discharge occurs at 400m, 800m or 1100m. Consequently, since many assumptions and conditions will differ, study of discharge at 400m may be of limited predictive value and is certainly not extrapolatable to commercial mining levels.

2.3.4. Research Required to Evaluate the Effects of Mining

At least three potential sources of pollution effects upon the pelagic layer of the Red Sea associated with metalliferous mud mining have been identified. Organic waste dumping from the mining ship while at the mine site is undoubtedly a minor concern and can be entirely avoided through use of shipboard treatment facilities or transport of wastes to shore for disposal along with the ore concentrate on a shuttle vessel. The vast requirements for uptake of ambient seawater to be used during the beneficiation process will certainly destroy the entrained members of the plankton community. The greatest threat to the Red Sea ecosystem is the requirement for massive volume discharge of tailings containing numerous toxic substances.

Although a review of selected publications on the 1979 Red Sea PPMT could lead one to obtain the impression that all potential environmental effects associated with mining the metalliferous mud deposits of the Atlantis II Deep have been satisfactorily addressed and sufficiently evaluated, this is clearly not the case as indicated by the scientists who conducted the environmental studies (Thiel *et al.* 1986, Karbe 1987, Thiel 1991). The limited temporal and spatial nature of the 1979 test mining campaign and the associated environmental monitoring efforts could not possibly provide the comprehensive data set required for a complete and definitive evaluation. Consequently, a number of outstanding questions remain to be answered.

Thiel (1991) identified at least five areas in which the data collected during the 1979 PPMT were insufficient for a definitive evaluation of the environmental risks associated with metalliferous mud mining in the Red Sea:

- discharge plume models and field observations of the test mining plume do not yet provide a coherent representation of plume formation, dispersal, and particulate sedimentation.

- nothing is known about the ultimate fate of the dissolved components of the tailings discharge.

- metal concentrations in the tailings are high and are toxic to animals after being leached into solution; to date, limited and occasionally contradictory data have been derived from laboratory experiments on standard test organisms, but long-term toxicological effects upon Red Sea fauna remain unknown.

- effects of tailings discharge upon the plankton community are not yet understood; secondary impacts via the food chain and through bioaccumulation may occur.

- effects of tailings discharge upon the benthos community are also not yet predictable; final plume development of both the particulate and dissolved phases and their settlement to the seabed are not known or understood.

Each of these unanswered questions or gaps in knowledge and understanding of potential impacts needs to be addressed through large-scale investigations as part of an environmental impact assessment in order to achieve results that would allow extrapolation to commercial mining. The preliminary work accomplished to date has identified the areas of greatest environmental concern.

As with nodule mining, the best, and perhaps only, opportunity to conduct appropriate risk assessment studies before major decisions on commercial mining are made is during a pilot mining operation (PMO) which has been identified by the ocean mining industry as the next developmental step in preparation for full-scale mining (Nawab 1984, Thiel 1991). Risk assessments delayed until after industry has made the necessary investments to initiate full-scale mining will be too late to allow effective environmental input into the decision making process and design of mining parameters. Although the Red Sea Commission had decided from the outset that if "the activities of the Commission in mining and/or treating the metalliferous deposits would result in destroying or severely damaging the marine environment, then the whole programme would be eventually frozen until such time it became possible to both mine the muds and preserve the environment" (Mustafa *et al.* 1984), it may become difficult to justify "freezing" when detrimental effects are detected only after huge sums have been invested and commercial mining has begun.

The at-sea portion of a metalliferous mud PMO is anticipated to be on a 1:5 scale and have a duration of 200-300 days (Mustafa and Amann 1978, Blissenbach and Nawab 1982, Mustafa *et al.* 1984), which should be ample to permit long-term monitoring of tailings discharge. An adequate monitoring effort will require dedication of considerable financial and human resources to study the effects primarily associated with discharge plume formation, dispersal, and effects upon biota, particularly the members of the plankton and benthos communities.

However, in preparation for this large-scale endeavour, a number of laboratory experiments and theoretical considerations should be conducted. These studies would not only add to impact evaluation but would also help to define and conduct effective investigations for the environmental part of the PMO phase. These pre-PMO studies should be concerned with

- laboratory experiments on the toxicity of the tailings on plankton and nekton species,
- baseline studies on metal concentrations in Red Sea organisms, and
- the further development of plume models for the fate of the tailings.

As pointed out above, the numerical models developed earlier were not in agreement with the observations made on the short-term discharge during the PPMT, and they predicted the plume fate for only a few hours. New insights in plume modelling can be expected after a period of more than 15 years and specific efforts in plume modelling (Jankowski and Zielke 1997, Segschneider and Sündermann 1997). Model verification and refinement can be achieved during the PMO-mining. At the same time, it must be realised that the plume must be followed at sea, probably for a long time span - months or years - following plume creation. During the earlier investigations it already became evident that tissues of certain crustacea and fish collected from the water column and seabed near the Atlantis II Deep normally exhibit mercury concentrations well above organisms from other deep sea regions and what is considered a safe limit for human consumption (Karbe *et al.* 1981). Had such a finding been made after commercial mining had begun, there is little doubt that the ocean mining industry would have been wrongly blamed for what is a pre-existing and normal condition. Organisms inhabiting the Red Sea may well have adapted to their unique environment in ways that may appear alarming although exhibiting natural conditions.

Further development of Red Sea metalliferous mud mining is currently being hindered by some of the same economic conditions that are delaying ocean mining in general, primarily low prices of the value metals on world metal markets due to increased use of recycled materials, substitution, and discovery of new terrestrial sources. These conditions will, however, eventually change and ocean mining economies may become favourable. In the further advancement of the Atlantis II Deep mine, however, it is important not to forget the needs of the environment. Pre-

PMO studies and the large-scale environmental PMO investigations are essential steps in this development.

2.4. Polymetallic Sulphides: Massive Consolidated Sulphides

2.4.1. Resource Description

Consolidated sulphide deposits (also referred to as hydrothermal mineralizations or massive polymetallic sulphides with reference to their origins or variety of metals they contain, respectively) were known from terrestrial occurrences well before the first discoveries of fossil and active localities in the oceans had been made. Land-based deposits, such as the well-known Cyprus sulphides which formed on the seafloor about $8 \cdot 10^7$ years ago, represent some of the most significant sources of copper (Cu), zinc (Zn), lead (Pb), silver, and gold and have been mined for centuries (Oudin and Constantinou 1984, Scott 1983, 1985, 1992a). Other important terrestrial occurrences of massive sulphides are under commercial production in Canada, particularly in the provinces of Ontario and Quebec where the Timmins and Noranda regions hold ores that originated as seabed deposits about $2.7 \cdot 10^9$ years ago (Malahoff 1982a). Additional significant accumulations on land are found in Oman and Japan and have supplied mankind's need for several important metals for many hundreds of years (Ballard 1984).

With better understanding of the geological history of the earth, it was recognised as early as the turn of this century that such sulphide deposits had been formed in association with submarine volcanism and hot springs on the floors of ancient seas, and had become part of the continental crust through processes associated with movement and uplifting of oceanic crustal plates. Pieces of oceanic crust which have been shifted onto adjacent continent or island margins in a process called obduction are termed ophiolites and may contain fossil metal-enriched deposits originating in association with ancient seafloor hydrothermal events (Marchig *et al.* 1987, Rona 1975, 1982). It was not until the 1960s, however, that the first actively forming polymetallic sulphide accretions were found within the axial region of the Red Sea (Degens and Ross 1969, Amann 1983, 1985). These unconsolidated sulphides are referred to as metalliferous muds and are discussed in Subchapter 2.3.

Major discoveries of actively forming and more consolidated polymetallic sulphide deposits were made toward the end of the 1970s when hot springs were found and sampled at the Galapagos spreading centre of the eastern equatorial Pacific in 1976 (Edmond 1984, Lonsdale 1984) and the first high temperature "black smokers" were located during a joint French-American expedition to the East Pacific Rise off Mexico in 1979 (Francheteau *et al.* 1979, Haymon and Kastner 1981, Hekinian *et al.* 1980, MacDonald *et al.* 1980, Spiess *et al.* 1980).

New discoveries continue to be made as ocean scientists further explore rift or spreading centre environments and subduction zones (Cronan *et al.* 1992, Rona 1992, Scott *et al.* 1992).

At present, actively forming polymetallic sulphide deposits are known from slow-, medium-, and fast-moving spreading centres along the mid-ocean ridge (MOR) system (Ballard and Francheteau 1982, Hekinian *et al.* 1983, Malahoff 1982a, Normark *et al.* 1982, Plüger *et al.* 1990), from the flanks of intraplate and axial seamounts and volcanoes (Lonsdale *et al.* 1982), from relatively young ocean basins (Lonsdale *et al.* 1980, Rona 1982), and from back-arc subduction zones (Batiza 1985). In addition to the Red Sea and the North Pacific localities already mentioned, important discoveries have been made in such diverse areas as along the mid-Atlantic Ridge (TAG hydrothermal field and Snakepit area), the Mediterranean Sea (Palinuro Seamount), the Juan de Fuca (Normark *et al.* 1982, Koski *et al.* 1985), Gorda (Malahoff 1981, Beauchamp 1984), Explorer (Tunnicliffe *et al.* 1986), and Endeavour (Tivey and Delaney 1985) Ridges of the eastern North Pacific, the Guaymas Basin of the Gulf of California (Lonsdale 1984), the Galapagos Rift (Corliss *et al.* 1979, Edmond 1984), the Florida Escarpment in the Gulf of Mexico, the Lau Basin (von Stackelberg *et al.* 1985), Okinawa Trough (Halbach *et al.* 1989), Mariana Trough, and other back-arc areas of the South Pacific, as well as recent discoveries on the Central Indian Ridge of the Indian Ocean (Halbach *et al.* 1994, Hannington *et al.* 1994). Depths of occurrence for these deposits range from about 3700m to less than 1400m. At present, approximately 130 localities with active hydrothermal deposits have been documented (Scott 1992a, 1995, Hannigton *et al.* 1994) and more discoveries of both active and fossil (inactive) sites continue to be made as oceanographic expeditions focus on likely areas using ever more sophisticated techniques.

Massive sulphides represent yet another type or expression of oceanic metal deposition and ore formation traceable to a hydrothermal origin (Edmond *et al.* 1979) (see Subchapters 2.2 and 2.3). Even the metals found in manganese nodules (Subchapter 2.1) may be considered to originate ultimately from hydrothermal sources. Hydrothermal sulphide and oxide mineralizations are end members of a continuous series of oceanic mineral deposit types. Polymetallic sulphides of Cu, iron (Fe), Pb and Zn are generally formed as precipitates from unmixed, primary, high-temperature (about 350°C or higher), acidic, and reducing hydrothermal solutions. On the other hand, metallic oxides of Fe and manganese (nodules and crusts) form as precipitates when the hydrothermal solution has been well mixed with ambient seawater in a low temperature, alkaline, and oxidising environment (Rona 1982).

The types and relative amounts of metals that are carried by and deposited from the heated seawater to some extent also depend upon the structure and make-up of the rock through which the water has passed during its convective circulation. Extremely porous rock strata may result in a more dispersed deposition pattern and may favour precipitation of some metals in layers beneath the seabed before the hydrothermal fluids can emerge and mix with bottom waters promoting deposition at the surface of the seafloor. Less porous rock may force the hydrothermal fluids to pass through restricted cracks or fissures that conduct the fluid to the seafloor more rapidly and in more confined localities. Such channelisation may be responsible for the formation of mounds topped with chimneys and spires that are typically found in more rapidly spreading regions. The mounds may contain collapsed chimneys formed during prior episodes of active hydrothermalism. Passage of fluid through evaporite lenses may increase the salinity of the solution and enhance its ability to dissolve certain metals from host rock. High brine solutions may also form stable layers covering disseminated deposits and help to keep them from becoming oxygenated. Another factor affecting the metal composition of a deposit is the type and thickness of sediment overlying the locality where the heated water emerges. Existing polymetallic sulphide samples may provide somewhat biased estimates of metal concentrations since many have been collected by breaking off pieces of chimneys or other recently formed surface expressions of hydrothermal deposition. There is a need to obtain samples from older, sub-surface deposits so that the effects of oxidation and dissolution of deposited metals over time can be better evaluated. The combination and interaction of all these conditions gives rise to great variability in massive sulphide ore tenors, both between and within deposits (Morgan and Selk 1984).

As with other hydrothermal deposits, massive sulphide formation requires proximity to a source of heat such as a magma chamber a few kilometres below the seafloor. Seawater that has permeated downward through pores, cracks and fissures in the hard crustal rock is heated by the magma and forced to rise again. This convective circulation of heated seawater through layers of sufficiently permeable rock allows dissolution and other chemical reactions to occur that enrich the hot seawater with metal ions and other substances. The metal-rich fluid may break out at the seafloor in relatively small vents and, if conditions are appropriate, may form the chimney-like features associated with many "black" and "white smokers" as a result of precipitation once the heated ascending water mixes with the cooler ambient water. Such active chimneys or pipes may be elevated up to 20m above their surroundings (Edmond 1984) and are constructed of crystallised metal sulphide compounds. "Black smoker" refers to the dark exhalations of iron and copper compounds that characterise high temperature (more than 350°C) vents, while "white smoker" is a term applied to emissions at lower temperatures containing predominantly Zn compounds (Malahoff 1982a). The "smoker" phenomenon is not restricted to emanations from chimney-like features, however. Even mounds, talus piles, or fractures in layered rock may exhibit "smoker"-like exhalations (Rona *et al.* 1986). Because of structural variability of subseabed strata, the heated and metal-enriched rising water meets cooler bottom water over a wider area and may form mound-like features or may simply enrich the overlying or nearby sediment as the dissolved metals are precipitated as a result of changes in the ambient physical and chemical conditions (comp. Subchapter 2.3).

Some distinctions may be made between hydrothermal deposits associated with slow-spreading centres (half-rate of spreading - i.e., movement of the crust away from the rift on one side of the spreading centre - of less than or equal to 2cm per year) and medium- to fast-spreading centres (half-rates in excess of 2cm per year). The former are characteristic of much of the mid-Atlantic Ridge and western Indian Ocean portions of the MOR system, as well as the newly opening ocean basin of the Red Sea (Rona 1983, 1985), while the later are typical of the Pacific and eastern Indian Ocean portions of the MOR system, as well as the newly forming ocean basin in the Gulf of California. The slower opening rates are associated with lesser heat flux and may favour formation of disseminated deposits consisting of dispersed mineral crystals or of stockwork deposits which consist of networks of mineralised veins within the host rock at some distance below the sea floor, while more rapid opening with higher heat flux appears to lead to formation of sulphide deposits in the form of mounds and chimneys directly on the seabed.

Back-arc basins such as the marginal seas of the western Pacific Ocean, where subduction zones and rifts exist between volcanic island arcs or between an island arc and the Asian continent, have also been demonstrated to harbour active and fossil massive sulphide occurrences on or beneath the seabed. In addition, axial and intra-plate volcanic seamounts are known to provide conditions necessary for development of sulphide deposits (Batiza 1984, Lonsdale *et al.* 1982). Many such areas are currently undergoing further exploration which should result in greater understanding of seabed mineralization processes as well as in enhanced predictive capability for locating sulphide deposits, both on land and beneath the sea.

To date, relatively few seafloor sulphide deposits have been shown to be "significant", that is, of sufficient size and quality to be potentially considered for commercial exploitation. Scott (1992a) listed six large known seabed sulphide deposits (the unconsolidated mud deposits in the Atlantis II Deep of the Red Sea, and consolidated massive sulphide occurrences of the Escanaba Trough, Middle Valley, southern Explorer Ridge and a seamount at 13° N in the Pacific Ocean, and the TAG (from Trans-Atlantic Geotraverse) site on the mid-Atlantic Ridge) which are of sufficient size to attract probably commercial interest. The Escanaba Trough and Middle Valley deposits are estimated to each hold several tens of millions of tons, while the other three locations may each contain up to 5 million tons, but are still of the same size order as most consolidated terrestrial deposits currently being mined. He further pointed out that discoveries of seafloor sulphide occurrences are relatively recent events and that more and larger deposits are expected to be located as more of the ocean floor is surveyed.

Information is especially needed on the thickness of sulphide deposits in order to make more meaningful economic judgements. Visual observations have allowed estimates based only on visible dimensions of mounds which may considerably underestimate their full extents. The TAG mound, for example, was judged to be about 250m in diameter and 50m in height, with an estimated 5 million tonnes of sulphides. Drilling by the Ocean Drilling Program in the Middle Valley area of the Juan de Fuca Ridge has shown that a sulphide mound was at least 94m in depth (Duckworth *et al.* 1994). The basement rock had not yet been reached and the entire core length was uniformly composed of sulphide compounds and showed no evidence of major intercalated sediment layers. On the other hand, drilling at the Snake Pit deposit of the mid-Atlantic Ridge (Ocean Drilling Program Leg 106 Scientific Party 1986) indicated a deposit thickness of at least 13m at the base of a "black smoker", which rapidly thinned to 3 - 6m above basement rock about 17m away. Some large fossil deposits on the seabed may have been covered by sediment while other sulphide mineralizations may have formed in porous layers below a capping structure well below the sediment surface. Until recently, only specially equipped drill ships such as the *JOIDES RESOLUTION*, were able to provide information on the depth of sulphide occurrences.

Commercial mining of consolidated sulphides on the seabed, appears to be a speculative activity at the present time, but under aspects of environmental research, these ore types must also be considered within this report.

2.4.2. Description of Mining Techniques

Only four basic approaches to mining exist: (i) scraping ore from a surficial deposit, (ii) excavating ore from the subsurface in an open pit, (iii) fluidising ore in a solution or slurry through a borehold or pipe, and (iv) tunnelling into the ore and removing it from beneath the surface (Cruickshank 1990). These methods may be utilised singly or in some combination, and are employed in many variations in accordance with the characteristics of the ore deposits.

The use of a scraping technique (i) is limited to removal of a relatively thin surficial layer (up to about 1m) from the surface being mined. Bucket dredges, the continuous line bucket (CLB) system, and the specialised active and passive seabed mining devices attached to hydraulic risers designed to harvest nodules are examples of this approach. It is conceivable that specialised scraping machinery that would permit exploitation of exposed consolidated sulphides could be developed. The crushed material might then be brought to the surface using hydraulic risers.

Excavation methods (ii) involve extraction of ores from an increasingly deeper work face, usually in an open pit-like feature. The specialised suction dredging devices designed to mine the unconsolidated sulphide material in the Red Sea are the only known example of a deep water application of this strategy. Some possible approaches to utilising this approach in exploiting deep seabed consolidated sulphide deposits have been suggested (Kaufman 1985), but improvements are required in rock penetration, fracturing, and fragmentation techniques before more conventional dredging of the ore can be applied to bring it to the surface.

There are at least two sub-types of the fluidising approach (iii) to harvesting metallic ores: (1) slurry mining and (2) solution mining (Cruickshank 1990). Slurry mining (1) requires *in situ* reduction of ore material to a slurry form and its removal to the surface via a borehole. Only a limited number of seabed sulphides might be amenable to such a mining method. Relatively soft sedimentary or exhalative beds overlain by hardened sediments or rock layers would be suitable, but few such deposit forms are known.

Solution mining(2) involves *in situ* extraction of preferentially dissolved value metals from an ore deposit brought into solution. Such method would allow implementation of deep oil drilling techniques to mining metallic ore deposits well beneath the sea floor and may prove to be the most viable procedure of mining metalliferous sulphides associated with crustal plate boundaries (Cruickshank 1990). There have been suggestions for using contained nuclear explosions to produce porous and shattered caverns suitable for introducing chemical solvents to differentially extract desired metals from the often complex sulphide mixture. An adequately large and non-porous flow path to allow introduction of solvent and withdrawal of the enriched solution is also required. Even selective *in situ* bacteria mediated leaching was brought in discussion.

The tunnelling (iv) approach is probably unsuitable to most known sulphide deposits since they occur at and below the sea floor at great depths (commonly between 1500 - 3500m) and often far from any nearby emergent feature such as an island or continent. If sufficiently attractive seabed massive sulphide deposits are found in proximity to island or continental shorelines, it may be possible to utilise mine shafts and lateral tunnels to reach them much as other stockwork deposits on the continents are currently exploited. In order to reduce risk to human miners, remotely-controlled robot mining machinery could be employed to separate and crush sulphide ores for transport through the tunnels and to the surface on shore.

Since some of the richest known deposits occur in the form of several meter tall spires and chimneys precipitated around exhalation vents, it would be possible to utilise large, TV-guided grabs to grasp the vent tubes, break them off, and bring them aboard a mining platform. Test sampling of a small massive sulphide deposit utilising a high capacity, TV-guided grab samplers, followed by observation of the site from a submersible, has been advocated as a possible means to learn about and mitigate potential environmental impacts associated with recovery of seabed sulphides at larger scales (Scott 1992b and pers. comm.). Such a system could also be utilised in recovering massive sulphides occurring in other forms, for example, in mounds of partially consolidated materials that are suitable for wire-line grab recovery. However, it is difficult to envisage such an approach ever to be utilised at a commercial scale since this requires vastly larger volumes of material to be retrieved per haul to justify the expense associated with offshore mining.

A particularly innovative mining approach that has been informally discussed among some members of the ocean mining community (Malahoff 1982b) would make use of technology currently being employed in the offshore oil drilling industry. Suitable vent openings could be fitted with re-entry cones that would allow a pipestring to be placed into a tube or fissure from which hydrothermal fluids are being actively discharged. The heated and metal-enriched hydrothermal fluid would then be pumped aboard a mining platform for processing and removal of value metals while the remaining solution is returned to the seafloor and released through a second or discharge pipe. There have even been suggestions made that mineral extraction could be combined with some form of ocean thermal energy conversion (OTEC) which utilises water temperature differences to drive generating equipment for production of electrical energy. Such a concept sounds simple, but has to overcome a variety of technical hurdles before practical application could even be considered. Of particular concern in utilising such a technique is possible damage to the mining and/or energy conversion equipment through corrosion and thermal effects (Scott 1992b). Even scientific sampling in proximity to a high temperature vent has occasionally proven to be problematic (Lonsdale 1984). In addition, precipitates may build up within the pipestring, eventually blocking fluid movement. Typical vent fluids also contain metals at too low a concentration to permit efficient recovery. Given these factors, mining sulphides using direct precipitation from vent fluids is currently considered to be unrealistic (Scott 1992a).

2.4.3. Potential Effects of Mining

As with concepts of how one might approach and conduct massive sulphide mining, a discussion of potential mining effects is by necessity speculative at best. The lack of information derives from the currently held view that massive sulphide mining is unlikely or certainly less likely than the exploitation of other ore on the seabed. Little effort has so far been expended on development of massive sulphide deposit mining techniques and, consequently, little can be definitively stated on possible environmental impacts.

The nature of massive sulphides and their restricted occurrence does allow some gross evaluations which would be common to any potential mining approach that might be employed. Just as with known occurrences of hydrothermal sulphide deposits on land, ocean deposits found to date are three-dimensional and occupy relatively small areas of the sea floor. Environmental effects resulting from mining such deposits should therefore also be on more limited scales, particularly when compared to the large surfaces that would be involved in mining a two-dimensional deposit such as nodules. Thus tailings disposal, plume development and the settlement of particulate matter from the plume would be more restricted. In some aspects, mining massive sulphide deposits will be similar to terrestrial "hard rock" mining in that it will require means of separating the ore from inert material and possibly some preliminary size reduction or crushing depending on the tpe of seabed to surface transfer.

A certainty again is that the unique community of chemoautotrophic and associated organisms occuring at active hydrothermal vents will not survive, when mining at or near such a locality is conducted. It should be remembered, however, that active vents are by nature a relatively short-lived phenomenon and that the faunal communities associated with them are also negatively impacted by natural events. Hydrothermal sites that have recently entered a quiescent phase have shown evidence of massive die-off of the community dependent on chemosynthesis, while newly active sites appear to be colonised rapidly by these opportunistic species. Vent biota have been found near hydrothermal cold seeps on the Florida Escarpment as well as in areas where hot hydrothermal fluids emerge (Rona *et al.* 1986). Even massive foodfalls such as whale carcasses create reducing microenvironments that temporarily accommodate organisms usually thought of as vent fauna (Smith *et al.* 1989). It would seem reasonable to assume that losses of some of these communities brought about by human exploitation of seabed ores could also be tolerated and that there is little chance of species loss. Vent biota appear to be well adapted to making use of the ephemeral nature and oasis-like distribution of active hydrothermal sites.

2.4.4. Research Required to Evaluate the Effects of Mining

If ocean mining of massive sulphide deposits ever makes the transition from the realm of imagination to reality, there will certainly be considerable need for evaluating potential mining effects, and consequently, a considerable need for knowledge on the ecosystem. Two different scenarios need to be considered: mining and impacting an area (i) at or near a hydrothermal vent and (ii) distant enough from a hydrothermal vent not to affect it.

Hydrothermal vents are characterised by their prominent communities discovered only 20 years ago. Made up mainly of new, large species they show dense populations of pogonophorans, bivalves, decapods, snails and some fish, as well as smaller species less visible on photographs. The variety of vents is wide and so is the distribution of animals. It is certain that most of these species require the vent habitat and do not live at other locations, not exhibiting the specific vent conditions. It is predominantly the intrusion of warm to hot waters, containing hydrogen sulphide from the crust below, on which the total biome exists.

Chemoautotrophic bacteria in the water, on inorganic substrates and organisms or as symbionts are the mediating key species gaining energy from the hydrogen sulphide and producing organic matter as the food basis for many other species as well as for the total community. These highly specialised communities occur where sulphidic particulates are produced and settle to the seafloor, probably leading to mineable ore and to protectable life at the same locality. The discovery of the vents and their biota have stimulated intensive research on life at sulphide biomes in shallow and deep-sea regions, and much has been learned about the vent ecology of species and assemblages. Quite good knowledge is already available (Hessler and Smithey 1984, Jones 1985, Tunnicliffe 1988, 1991, Cavanaugh 1994, Fisher 1995, Van Dover 1995) and will further increase since these fascinating systems will remain in the top rank of deep sea research. It thus seems that basic research on hydrothermal communities is already and will continue to be well underway.

It is difficult to conceptualise the future mining technique and its impact potential, but since vent areas are mostly of limited extent and at some distance from each other, the basic knowledge gained may be sufficient to decide whether a specific vent must be protected or may be sacrificed for commercial mining.

General questions for which answers are essential:

- Are vents characterised by endemic species or rare genotypes?

- Do gene flows exist between neighbouring vents?

These questions are of general inerest and they are already on the project lists of vent deep-sea ecologists.

It seems more difficult to propose research for the other scenario at greater distance from any hydrothermal vent. Unlike ore deposits which are known (Subchapter 2.3) or existing mining claims (Subchapter 2.1), consolidated sulphides could be mined wherever a resource-sized ore body exists. Basic knowledge on the oceanography of the region will be needed, the area of influence from plumes created and tailings discharged as well as their potential effects, must be studied, and general knowledge on the distribution of species in the deep sea must be available.

This last topic is of general interest and deep-sea biologists are already working on it. Due to the problem of species definition, purely by morphological characteristics or by genetic differences, one cannot expect a simple answer. Since the importance of species distribution for impact assessments is not generally understood, no specific research programmes are directed to this question. These should be initiated for key taxa, i.e., those animal groups that promise the best general results on this problem.

In a later phase, when industry approaches the mining of consolidated sulphides, new thoughts should be given to environmental studies. At minimum industrial PPMTs and PMOs must be intensively used for environmental impact assessments. But before these tests occur and as soon as potential mining locales have been identified and delineated, the local oceanography with all its disciplines, including short-term research and yearly variabilities need to be assessed, to establish a basis for comparison against which environmental changes that may occur as a result of the mining process can be both detected and evaluated.

2.5. Phosphorites

2.5.1. Resource Description

Offshore phosphorite nodules were first dredged on the continental shelf off South Africa during the Challenger Expedition in 1874 (Murray and Renard 1891b). Since that time, numerous near-coastal phosphorite deposits have been located on the sea floor and on the tops of seamounts (see Baturin and Bezrukov 1979, Baturin 1982, Rowland 1985, Burnett *et al.* 1987 for reviews and Fig. 2.5.1, Cook 1988).

The term "phosphorite" is applied to sedimentary rocks of marine origin as well as bird or bat guano deposits. Phosphorites differ from common sedimentary rocks by virtue of their increased phosphorus pentoxide (P_2O_5) contents. Most authors place the limit at values between 5 and 40% P_2O_5 (Riggs 1979, Bentor 1980). In contrast, typical sedimentary rocks and sea floor sediments contain less than 0.3% P_2O_5 (Riggs 1979).

The major phosphorus-fixing mineral in marine phosphorites is carbonate-fluorapatite (CFA), also termed francolite. The variable composition of offshore phosphorites can be generalised by the chemical formula: $Ca_{10-x-y}\,Na_x\,Mg_y\,(PO_4)_{6-z}\,(CO_3)_z\,F_z\,FOH$ (McArthur 1990).

Quantitatively important phosphorites occur as authigenic precipitates and as replacements of carbonates in the shape of irregular masses, nodules, conglomerates, grains or crusts. They are usually located on the continental shelves and the upper part of the continental slopes and are typically associated with terrigenous, calcareous or siliceous sediments.

Experimental investigations have shown that a concentration of 0.1ppm of orthophosphate (PO_4^{-3}) in saline pore solutions is sufficient to initiate the replacement of carbonates (Ames 1959). Compared to a total concentration of 0.07ppm PO_4^{-3} in seawater (Brewer 1975), a radical change in the phosphorus content is not necessary for the initiation of carbonate replacement, provided that the physico-chemical conditions are suitable (Birch 1980). The estimated average PO_4^{-3} content of upwelled water is 0.2ppm (Spencer 1975) and, therefore, sufficient for the replacement of carbonates.

Even at present, the question of the phosphate source at the sediment-water interface has not been satisfactorily answered. Potential sources are organic matter, dissolution of fish debris, and release of phosphate sorbed onto hydrous iron oxides such as limonite (FeOOH) (for detailed discussion see Froelich *et al.* 1988). Additionally, phosphogenesis occurs during *post mortem* mineralisation of phosphorus-bearing bacteria (O'Brion *et al.* 1981, O'Brion and Veeh 1983).

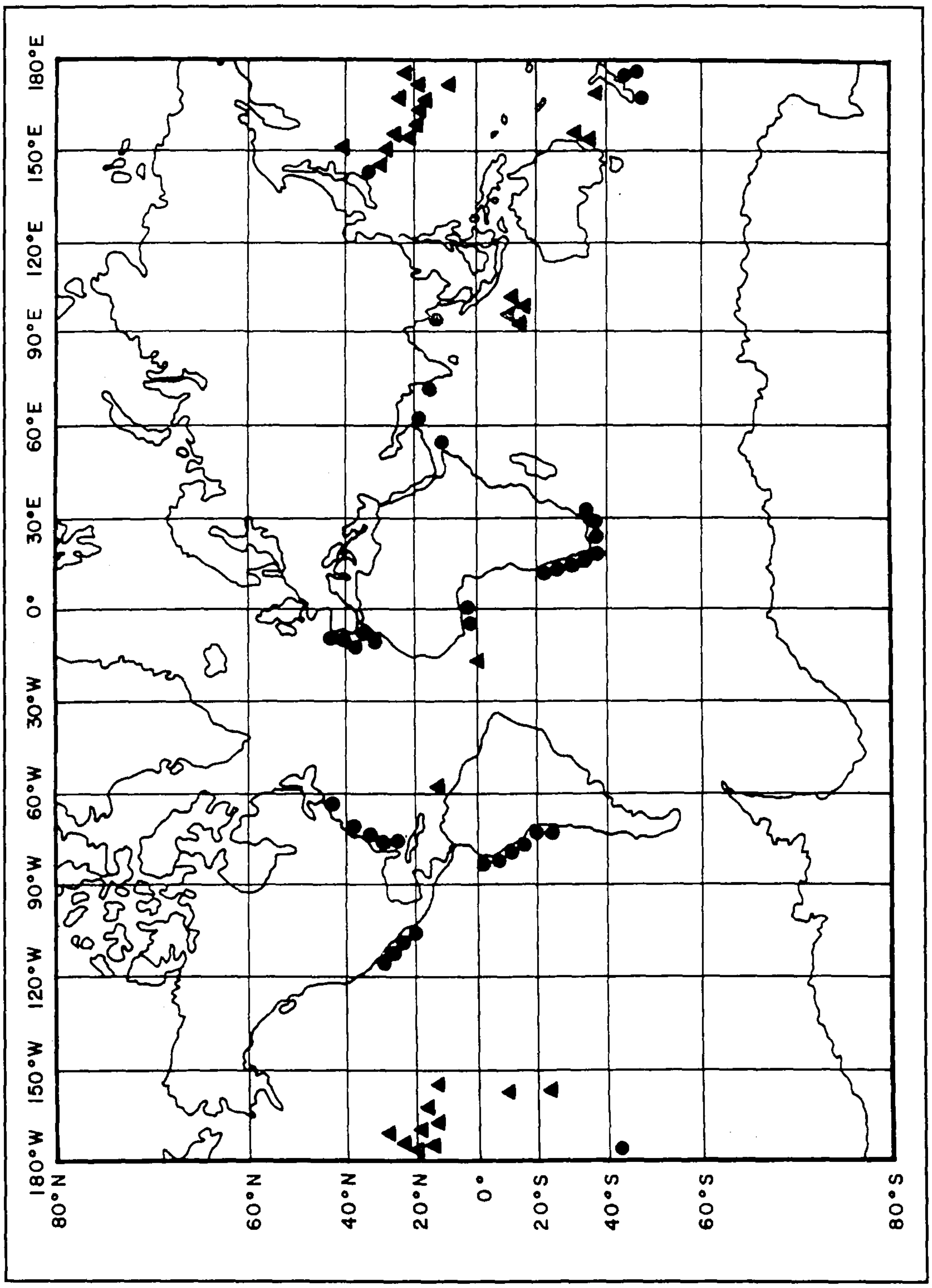

Figure 2.5.1 Distribution of marine phosphorites on continental shelves (●) and seamounts (▲) (redrawn from McKelvey 1986 after Bezrukov and Baturin 1976).

Phosphorites are present in the sedimentary rock column of almost all ages since the early Proterozoic (2200 million years before present (BP)). But only distinct time intervals in the geologic past such as the later Proterozoic, early Cambrian, Ordovician, Permian, Jurassic, late Cretaceous-Eocene, and Miocene were favourable for quantitatively important phosphorite depositions (Cook and McElhinny 1979, Sheldon 1980). Most phosphorites on the present sea floor are relicts and of Miocene age. Only phosphorites from four localities are of recent age. These include (Fig. 2.5.1) the deposits off Peru and Chile (Veeh *et al.* 1973, Burnett 1977), off Namibia (Baturin *et al.* 1972, Veeh *et al.* 1974), off eastern Australia (O'Brion and Veeh 1980, O'Brion *et al.* 1981), and off Baja California in Mexico (Jahnke *et al.* 1983).

In the recent literature, phosphorite pellets from the continental margins of Peru and Chile (Baker and Burnett 1988, Burnett 1990), of south-west Africa (Birch 1979b, Thomson *et al.* 1984), and of Baja California, Mexico (D'Anglejan 1967) are viewed as modern analogues of large ancient phosphorite deposits because the bulk of terrestrial phosphate reserves are also of the pelletal type (Burnett and Froelich 1988). When comparing the small modern with the large ancient ore deposits, the variation in magnitude remains an open question. Burnett and Froelich (1988) pointed out that the differences in upwelling areas are more a matter of time scales rather than of distinct environments or processes. Investigations in non-upwelling areas suggest that phosphorite formation is not a question of episodic removal of phosphorus from seawater, but rather a process of physically localised concentration of CFA precipitates (Ruttenberg and Berner 1993). Concentration of precipitates by reworking or winnowing is assumed to be one important step in the formation of economically significant deposits (Burnett 1977, Sheldon 1980, Baturin 1982). Recently, the *in situ* concentration of phosphatic pellets without winnowing on the continental margin of Peru was also discussed (Baker and Burnett 1988, Burnett *et al.* 1988).

2.5.1.1. East Atlantic Continental Margin (Namibia and South Africa, Morocco, Spain)

The phosphorites of the Atlantic coast of Africa can be divided into authigenic and replacement deposits (Fig. 2.5.2). The authigenic deposits prevail on the shelf off Namibia and occur only at isolated localities off the western coast of South Africa. Extensive areas of replacement phosphorites are characteristic for the western and southern continental margins of South Africa (Birch 1979a, 1979b, 1980, Thomson *et al.* 1984, McArthur *et al.* 1988, Bremner and Rogers 1990). Along the east coast of central Africa, only isolated and less-enriched replacement carbonates and phosphatised faecal pellets occur (Baturin 1982). On the shelf of North Africa (Morocco), both authigenic and replacement phosphorites are present (Summerhayes *et al.* 1972, Baturin 1982).

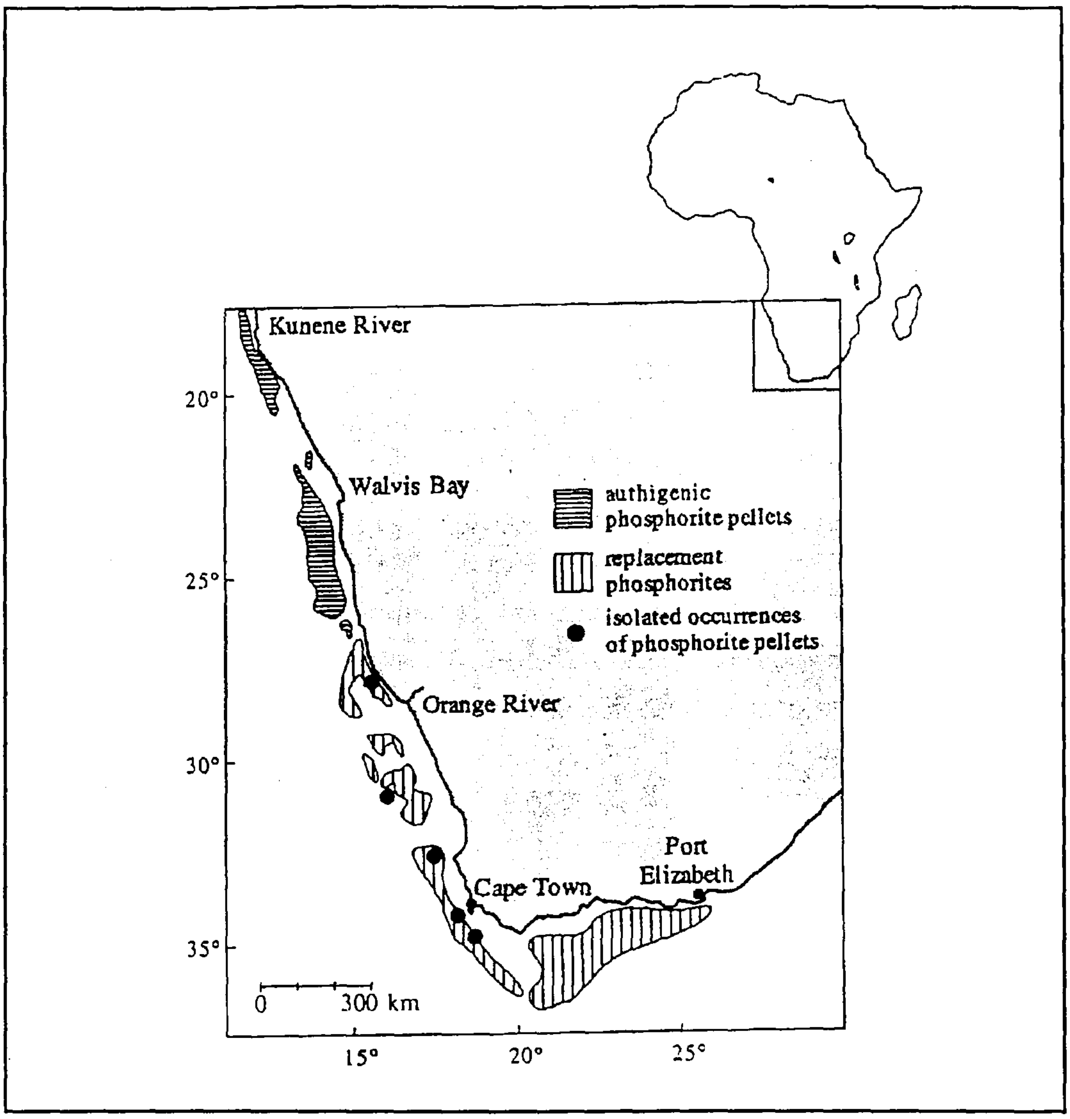

Figure 2.5.2 Locations of phosphorite resources of southern Africa (from Birch 1980)

Authigenic phosphorites off Namibia occur as concretions in water depths shallower than 78m and as pellets on the middle shelf mixed with diatomaceous, calcareous, clastic, and glauconitic sands. The concretions are irregular, discoidal and elongated, and exhibit a dull matted or polished surface. Only small amounts of diatom shells and aluminosilicate debris can be found in the CFA matrix. Locally, the whole rock is comprised of 24.9 to 31.4% P_2O_5. Age determinations have shown that the concretions are entirely Holocene with an estimated maximum age range from 500 to 3800 years (Thomson *et al.* 1984). Comparing the concretion growth and accumulation rates of 0.3cm $\cdot$ yr^{-1} over the past 5000 years (Bremner 1980), winnowing of fine sediment components must be assumed for continuous growth (Burnett *et al.* 1982).

Pellets and glauconitised pellets are found on the present sea floor in water depths greater than 170m. The dull, brown-coloured pellets are ovoidal or spherical, and the sizes vary generally between 130 and 350µm. The pellets are very well to well rounded and very well sorted. Most of the pellets contain nuclei made up of quartz, feldspar or biogenic debris. Concentric banding around the nuclei is frequently observed, whereas the CFA layers are separated from one another by organic matter. The pellets are thought to be precipitated in a shallow lagoon or estuary adjacent to an area of intense upwelling and high biological productivity, and then redeposited on the middle shelf (Birch 1979b, 1980, Thomson *et al.* 1984). Recent age determinations by strontium isotope methods yielded ages of 1.15 and 4.7 million years (McArthur *et al.* 1990).

Along the coast off South Africa, phosphatised limestone, glauconitic sands cemented by phosphatised carbonate mud, and reworked material of the latter as conglomeratic phosphorites are present in water depths between 100 and 500m (for a detailed description see Birch 1979a, 1980, McArthur *et al.* 1988). Most of the phosphatised limestones are rounded or tabular, dull-surfaced, green-coloured pebbles and cobbles of primary foraminiferal packstones. The cemented glauconitic sand consists of shiny, dark green, rounded to irregular pebbles and cobbles which exhibit bioturbated internal fabrics with burrow fills. The phosphorites probably formed by replacement of limestones in oxic or post-oxic, organic-poor environments (Birch *et al.* 1983, McArthur *et al.* 1988). Phosphogenesis off South Africa has occurred in four episodes around 63, 83, 250 to 800kyr BP, and 800kyr BP and the mid-Eocene during the reworking of Tertiary limestone related to sea level fluctuations (Birch *et al.* 1983, McArthur *et al.* 1988).

On the shelf off Morocco, phosphatised limestone, glauconite-rich conglomerates (reworked limestone) and authigenic pelletal phosphorites occur between 31° N and 33°45' N latitude (Summerhayes *et al.* 1972, Baturin 1982, Summerhayes and McArthur 1990). The foraminiferal limestones and conglomerates were phosphatized by replacement of calcareous material in regions remote from detrital sources. The conglomerates comprise concretions of irregular shape, angular fragments and blocks with a size range of less than 1cm up to 8cm. They are found in 150 to 300m water depths. The conglomerates represent the quantitatively important phosphorite deposit in this region (Baturin 1982). The sand-sized pellets are of authigenic accretionary origin. Around nuclei of quartz, calcite, dolomite and foraminifera tests, a commonly structureless coating of CFA is present. No active phosphogenesis is documented at the sediment-water interface in the Moroccan shelf region. Absolute determinations yielded Miocene and mid-Miocene ages (Summerhayes *et al.* 1972, Baturin 1982).

The phosphorites on the continental margin off northern Spain are situated in water depths between 300 and 1000m at Cape Ortegal near the shelf edge and on top and flanks of Le Danois Bank (Lucas *et al.* 1978, Lamboy and Lucas 1979). Phosphorites occur as nodules in a variety of more or less conglomeratic form. At Cape Ortegal, the nodules are associated with sediment rich in quartz and glauconite grains, whereas on the Le Danois Bank, fine glauconite-poor sediment prevails. Ferruginisation and manganese encrustations were observed only on top of the Bank. Phosphogenesis took place by replacement of lithified calcium carbonates and by cementation of rock fragments, bioclasts, tests, fine matrix glauconite grains or structures associated with organic matter. It is assumed that phosphogenesis began during the Miocene. Evidence also exists for Quaternary CFA precipitation (Lamboy and Lucas 1979).

2.5.1.2. West Atlantic Continental Margin (the Carolinas, Georgia and Florida)

The Blake Plateau off South Carolina and Georgia (U.S.A.) is the best known phosphorite deposit on the west Atlantic continental margin. Additional phosphorites have been found along the coast off Florida on the upper and lower Miami Terrace and the Pourtales Terrace, an erosional bench structure off the Florida Keys (for review see Manheim *et al.* 1980, Riggs 1984, Rowland 1985).

The Blake Plateau is situated between Cape Hatteras and the Bahama Islands on the west Atlantic continental shelf. Four different types of consolidated deposits occur on the Plateau. Well-sorted carbonate foraminiferal sands with locally varying amounts of deep water corals dominate the platform. Phosphorite nodules, phosphate-manganese pavements and slabs, and finally, ferromanganese concretions occur over an areal extent of 22,000km^2 off the south-eastern coast of the United States. Water depths increase eastward ranging from 250 to 1000m. The phosphorites are composed of pellets, granules, pebbles, and conglomerate aggregates. At some localities, masses of phosphorite grains cemented by CFA form a slab-like pavement.

At the present time, the Blake Plateau is an area of erosion and the phosphorites are of relict origin. Precipitation of the primary phosphorites probably started in the Oligocene and reached its maximum during the early to middle Miocene. Less CFA may have precipitated during the Pliocene and Pleistocene (Manheim *et al.* 1980, Riggs 1984). It is assumed that the former phosphorites extended from the south-eastern U.S. continental shelf to the Blake Plateau (Manheim *et al.* 1980). The Tertiary phosphogenesis took place in a highly productive environment accompanied by limited detrital input and anoxic conditions at the sediment-water interface. Riggs (1984) pointed out that upwelling was present during the Neogene.

Deposits similar to the Blake Plateau phosphorites occur off the continental shelf of the Florida Peninsula. The upper and lower Miami Terraces, found in 200 to 375m and 600 to 700m water depth, respectively, and the Pourtales Terrace off the Florida Keys at depths of 20 to 350m are characterised by phosphorite pavements and slabs of lower to middle Miocene age (Gorsline and Milligan 1963, Mullins and Neumann 1979, Rowland 1985).

Other potentially significant deposits occur about 95km offshore North Carolina in 15 to 26m water depths of Onslow Bay (Riggs *et al.* 1982, Rowland 1985, Hobbs 1991). This highly phosphatized Miocene section of the coastal plain contains neither uniform nor continuous phosphate beds, but outcrops extend offshore in a NE-SW trending belt of about 145 by 40km that continues into the subsurface both east and south. The area includes eight beds with substantial phosphate concentrations of which at least five are thought to be economically viable. These contain an estimated 1.36 billion metric tons of phosphate concentrates ranging from 28 to 30% phosphate (Rowland 1985).

2.5.1.3. East Pacific Continental Margin (Peru, Chile, California and Baja California)

The occurrence of modern phosphorite nodules and pellets on the sea floor along the continental margin off Peru and Chile (Fig. 2.5.3) is well documented (Veeh *et al.* 1973, Burnett 1977, 1990, Burnett *et al.* 1982, 1983, 1988).

The phosphatic material found in the sediment off Peru and Chile consists of nodules, grains (pellets), crusts, foraminifera and fish debris. Nodules are characterised by numerous phosphatically cemented layers which bind size- and/or morphologically-sorted phosphatic, biogenic and siliciclastic grains. They are irregularly shaped, dull and earthy in appearance (Burnett 1977, Glenn and Arthur 1988). In some areas, the sediment mass is dominated by pellets making up more than 80% of the total. The authigenic pellets are black, spherical to ovoidal in shape, and show a nucleus of inorganic mineral grains. The size of the pellets generally varies between 125 and 500µm (for detailed description see Baker and Burnett 1988, Glenn and Arthur 1988).

Pellets were sampled from the continental margin (100 to 600m water depth) in an area of intense biological activity mediated by coastal upwelling processes (Baker and Burnett 1988). At the depositional centre, an oxygen minimum layer is developed (Suess 1981), resulting from the oxidative consumption of the high organic matter content of the surface sediment layer. The growth site of the phosphorites is generally located at the upper and lower boundary of the oxygen minimum layer (Burnett *et al.* 1983). Pellets form authigenically near the sediment-water interface in a period of less than 10 years after growth is initiated (Burnett *et al.* 1988). The nodule growth rate of less than 1 to 10mm · kyr^{-1} is much slower (Burnett *et al.* 1982). Comparing growth and sedimentation rates, winnowing of fine sediment components is a presupposition for continuous growth at the sediment-water interface (Burnett *et al.* 1983).

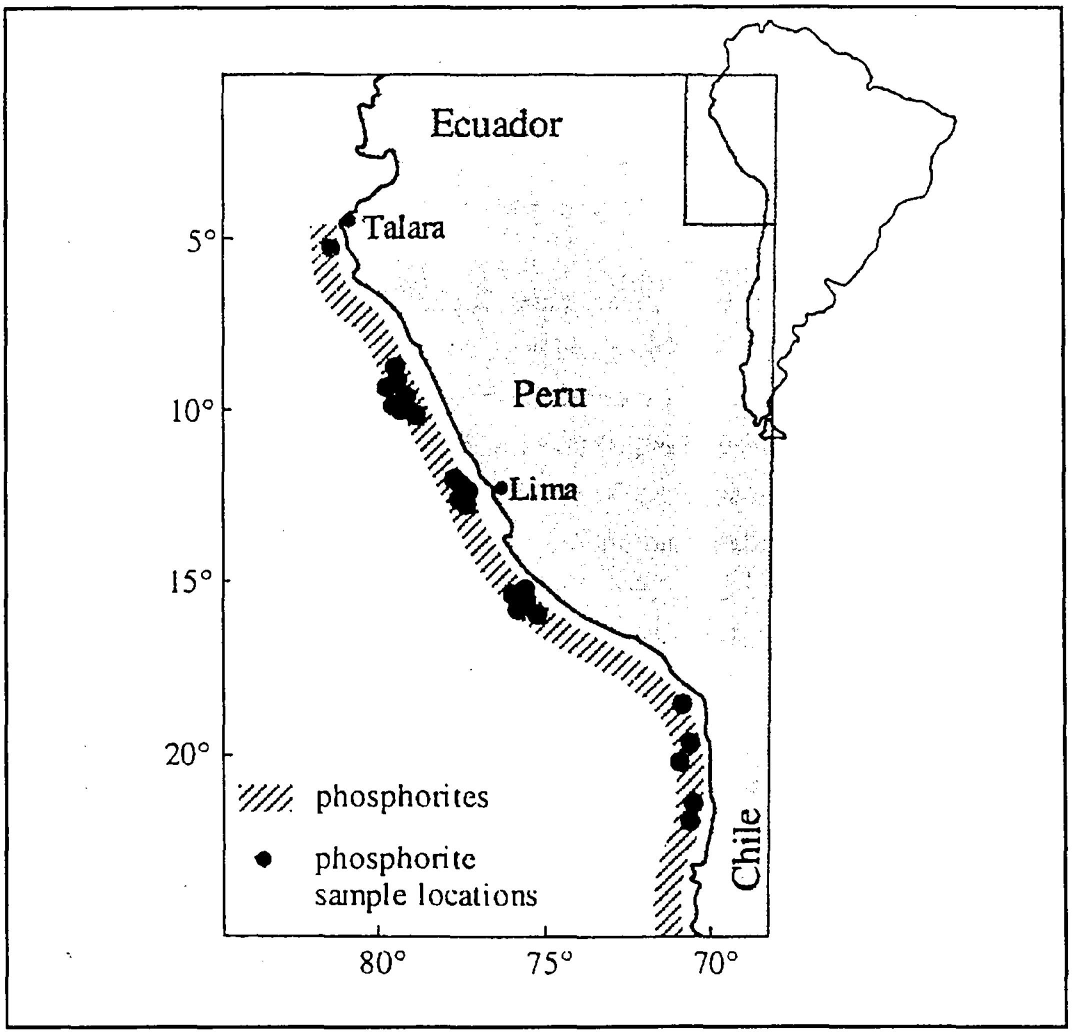

Figure 2.5.3 Locations of phosphorite resources off Peru and Chile (modified from Burnett 1977, Burnett and Froehlich 1988, Burnett 1990)

The California phosphorite province extends between 21° N and 34°30' N along the western continental margin off California (U.S.A.) and Baja California (Mexico), a distance of more than 2000km (Emery 1960, D'Anglejan 1967, Baturin 1982). The continental margin off California between 31° N and 34°30' N is characterised by a narrow shelf and an upper continental slope divided into a series of trenches and rises. Phosphorites occur on the outer shelf and on the flanks and hills of the upper slope in water depths of 80 to 2800m. However, most of the phosphorites (95%) do not occur deeper than 330m. They accumulate only in areas of slow sedimentation and are absent on the bottom of trenches (Roberts 1989). These phosphorites occur as grains, sheets and concretions of various shapes with an average diameter of 5cm. Some grains exhibit an oolithic structure around nuclei of foraminiferal tests, glauconite grains or clastic material. The diameter of the grains varies between 0.1 and 0.3mm. These deposits are usually associated with quartz-mica and glauconitic sands. In the area off California, two stages of phosphogenesis can be differentiated extending from the middle to late Miocene and late Pliocene to early Pleistocene (Emery 1960). The smooth and polished surfaces of the phosphorites and the manganese oxide coatings on top of the concretions in greater water depths are evidence of ceased phosphogenesis.

Patchily distributed phosphorites are also found on the continental margin and slope off Baja California between 21° N and 26° N (D'Anglejan 1967, Jahnke *et al.* 1983). These phosphorites formed on an up to 80 km-wide, shallow erosional platform of little relief and on the upper slope where high biological productivity in the summer leads to development of an oxygen minimum layer. Authigenic pelletal phosphorites and phosphatised biogenic detritus occur on the platform. The black, structureless pellets are ovoidal-shaped and occur as sand-sized particles. These phosphorite pellets are composed of 90% CFA and varying amounts of detrital inclusions, syngenetic sulphides, organic matter, opaline silica and disseminated calcite. The calculated average P_2O_5 content lies at 30.2%. On the platform, the pelletal phosphorite deposits are generally associated with the sand-sized sediment facies at water depths of 50 - 100m. The surface distribution of the deposits is patchy, with concentrations fluctuating over short distances between 15 and 40% by weight of total sediment (D'Anglejan 1967). Nodular phosphorites were dredged at water depths of 100 - 200m along the offshore banks and the upper continental slope. Miocene foraminiferal limestones were partially to completely replaced by CFA, and subsequently reworked by currents (D'Anglejan 1967). At greater water depths (200 - 390m), crystalline CFA was detected only within the 4 - 6cm interval of the sediment column (Jahnke *et al.* 1983). The age of the pelletal and replacement phosphorites is uncertain. Fossil evidence and radiometric measurements suggest that phosphogenesis took place from Pliocene to Recent (D'Anglejan 1967). Age determinations of the disseminated CFA yielded maximum ages of 3000 - 4000 years (Jahnke *et al.* 1983).

2.5.1.4. East Australian Shelf and the Chatham Rise East of New Zealand

The Holocene phosphorites off eastern Australia are an example of phosphogenesis in an environment of only limited seasonal upwelling, low biological productivity and low carbon flux into the sediment (O'Brion and Veeh 1980, 1983, O'Brion *et al.* 1981, 1986, Cook and O'Brion

1990, Heggie *et al.* 1990). Along the continental margin of east Australia between latitudes 28° S and 32° S, phosphorites have been found in water depths between 200 and 455m. This area is characterised by a relatively narrow shelf (25 - 40km) and an unusually steep shelf break at varying water depths between 210 and 450m (Marshall 1979). The phosphorites occur in the form of nodules on the shelf and upper slope and may be divided into ferruginous and non-ferruginous nodule types. Non-ferruginous nodules are typically 1 - 5cm in diameter, invariably friable and earthy in appearance. They have been recovered at water depths between 275 and 455m on the upper slope associated with unconsolidated, iron-poor, glauconitic, foraminiferal sands. The sedimentation rate of the sands is less than $1cm \cdot kyr^{-1}$, and the content of organic carbon is lower than 0.5% in the sediment (O'Brion *et al.* 1981). The phosphorites probably originated through *post mortem* alteration of phosphorus-rich bacterial cells to CFA (O'Brion *et al.* 1981, O'Brion and Veeh 1983). Recent investigations indicate that the phosphorus is trapped by scavenging within the zone of phosphogenesis and nodule growth (upper 15cm of the sediment). Phosphorus, released by rapid degradation of organic matter supported by intense bioturbation, is scavenged by iron recycling between oxic and anoxic sediment (Heggie *et al.* 1990). The relatively low phosphate content of the nodules (less than 15% P_2O_5) can be related to the dilution of CFA by allogenic and authigenic debris and minerals. Dating using the uranium/thorium method resulted in late Pleistocene and Holocene ages being determined. It is assumed that phosphogenesis has probably continued for the last 100kyr and is not related to sea level fluctuations (O'Brion *et al.* 1986). The ferruginised nodules are believed to be a ferruginised and extensively reworked variety of the Quaternary non-ferruginous nodule type. These goethite-rich, well-indurated and often conglomeratic nodules exhibit diameters between 1 and 15cm. They are found on the outer shelf in shallow water between 200 and 275m in current swept areas.

The Chatham Rise phosphorite deposits east of New Zealand (Fig. 2.5.4) are one of the best investigated submarine replacement phosphorites (Cullen 1980, Rad and Kudrass 1984). The deposits extend along the crest of the Chatham Rise between latitudes 43°19' S and 44°05' S and longitudes 177°08' E and 177°27' W. The largest nodule accumulations have been found between 179° E and 179°50' E in 350 - 450m water depth (Kudrass 1984). The phosphorite nodules are distributed in patches. Iceberg scouring probably disturbed the primary phosphorite distribution on the crest of the Chatham Rise, but only on a local scale (Kudrass and Rad 1984a). The vertical distribution of nodules within the foraminiferal glauconite sand is uniform with the exception of some nodule enrichment in the lower portion of the sediment column (Kudrass 1984). Bioturbation is an important factor influencing the vertical distribution of the nodules (Cullen 1980). All phosphorite nodules are of replacement origin and exhibit a distinct concentric zonation (Cullen 1980). Sub-angular to sub-rounded nodule shapes prevail, while the markedly irregular shapes of larger nodules may develop through selective solution and borings. Nodule size varies between 0.5 and 200mm with the majority falling between 8 - 50mm. The average P_2O_5 content of the nodules is 22% (Kudrass 1984). The source of the supply of phosphorus initiating the phosphogenesis is still in some doubt. The phosphorus for replacement was either released by degradation of organic matter accumulated by upwelling processes or obtained by direct uptake of phosphorus from seawater (for discussion see Kudrass and Rad 1984b). The phosporite nodules are found in and on unconsolidated Miocene and Quaternary foraminiferal glauconite sands intermixed with a considerable input of eolian volcanogenic and glacial marine components. Most of the unconsolidated sands were sub-marine eroded during the Pliocene and Pleistocene, forming a karst-like topography (Kudrass and Rad 1984b). Based on strontium isotope dating, the CFA replacement of carbonates occurred about 4.9 million years BP (McArthur *et al.* 1990).

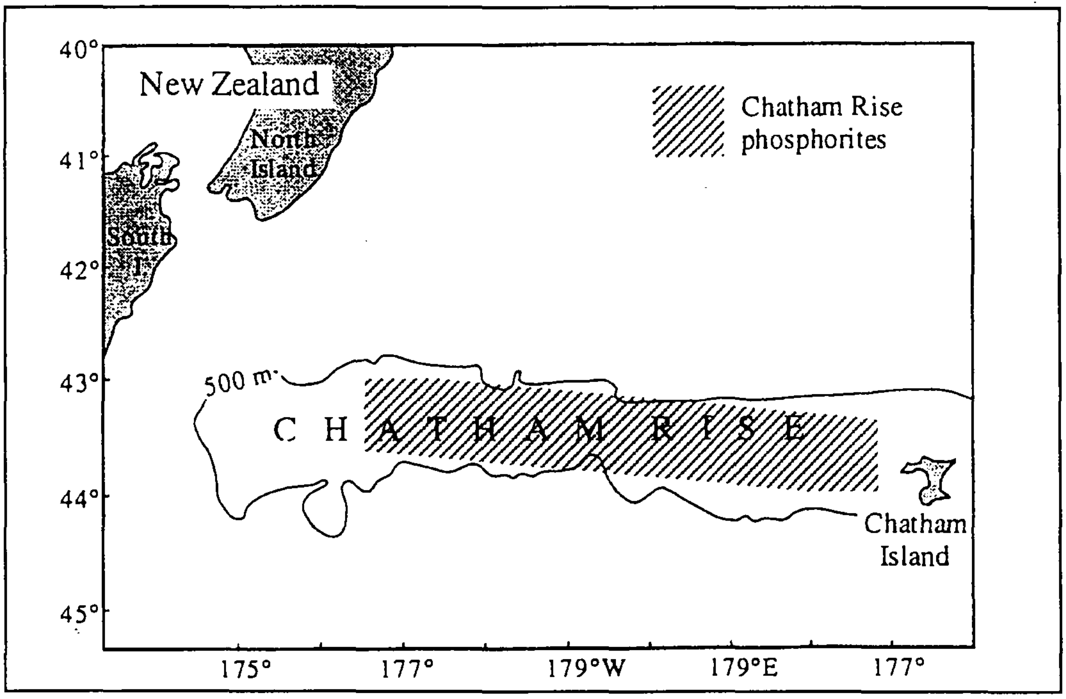

Figure 2.5.4 Area of phosphorite resources on the Chatham Rise east of New Zealand (from Rad and Kudrass 1984).

2.5.1.5. Economic potential

The P_2O_5 reserves found on the present day sea floor in three regions along the Atlantic coast off Africa have been estimated. With a presumed thickness of 1m for phosphatized rock on the Agulhas Bank, a reserve of 140 million tons of P_2O_5 was arrived at (Baturin 1982). With respect to the entire continental margin of South Africa, a P_2O_5 reserve of 10,000 million tons was calculated (Birch 1979a). More recently, the reserves on the Namibian continental shelf were estimated to be 3020 million tons of P_2O_5 (Bremner and Rogers 1990). It has been suggested that the Moroccan continental shelf comprises a reserve of 430 million tons P_2O_5 (Summerhayes *et al.* 1972).

The tonnage of the phosphorite concretions on the Blake Plateau were approximated at 2000 million tons, corresponding to about 440 million tons P_2O_5 (at an estimated average phosphorus pentoxide content of 22%) (Manheim *et al.* 1980). The potential reserves contained within the manganese-phosphorite pavement were not taken into account. The Onslow Bay phosphorite beds could yield an additional 1.5 billion tons of phosphate concentrate or 435 million tons of P_2O_5 at an average grade of 28-30% (Riggs *et al.* 1982).

The coverage of phosphorites off California, as estimated through dredging, is highly variable. The most recent phosphorite resource assessment was undertaken by Hess (1978) who estimated reserves of 65 million tons of phosphorite nodules and 52 million tons of phosphorite sands. The estimated area of maximum phosphorite concentration on the continental margin off Baja California is approximately $1800km^2$. Based on this areal distribution and an average sediment CFA content of 5%, a minimum reserve of 1.5 million tons of P_2O_5 was calculated. A maximum estimate of 4 million tons was also given (D'Anglejan 1967).

Because of the high regional variability of the phosphorite nodule concentration on the Chatham Rise, its quantitative assessment is difficult. Between 179°08' E and 179°42' E longitude, about 25 million tons of P_2O_5 occur in an area of $378km^2$, amounting to an average of $66kg \cdot m^{-2}$ (Kudrass 1984). Additional reserves with only $11kg \cdot m^{-2}$ are expected to be found in an area of about the same size (Kudrass and Rad 1984b).

About 88% of the phosphate produced in the world is utilised for agricultural purposes with the remainder consumed for various industrial purposes in the form of chemical products (Marvasti and Riggs 1989). Although potential marine reserves of phosphorites are extensive and widely distributed, and will at some point contribute to the world-wide demand for this important ore material, they will probably not be exploited until terrestrial sources become uneconomic or are exhausted.

2.5.2. Description of Mining Techniques

The economic interest in marine phosporites and the development of adequate mining techniques depends on the world marketprice of phosphate. At present, the known magnitude of the reserves on land is much greater than in the oceans and efforts for developing ocean mining targeted to this ore are low. However, once traditional terrestrial phosphorite sources are sufficiently depleted or potential land areas become protected, the interest in ocean mining for phosphatic material can be expected to increase correspondingly.

In general, the mining technique eventually utilised must be capable of operating under various conditions because of the high variability of phosphorite deposits which differ in water depth, morphology of the ocean floor, extent of phosphorite coverage, shape and size of phosphatic material, and content of P_2O_5. Moreover, several working steps during the total process must be taken into account, i.e., the collection of the phosphatic material on the ocean floor, its transport to the sea surface, and its subsequent handling and processing including transportation to processing facilities on land or at sea.

Since phosphates have not been mined at sea, no specific techniques have been developed and environmental disturbances have not been observed. However, various types of other mineral resources have been mined from the sea floor in shallow and deep water, and it seems justified to assume that one or another technique or component thereof could be adjusted and employed for phosphorite mining. One parameter determining potential mining techniques is water depth because not all of the possible techniques are suitable for great depths.

Sediment pumping has been and continues to be used regularly for the mining of unconsolidated sand from shallow waters in order to obtain materials for coastal protection and for construction. It is also employed in placer mining for such materials as precious metals. This method would be suitable as well for deep water mining as was demonstrated in a pre-pilot mining test pumping metalliferous muds from more than 2000m depth in the Atlantis-II-Deep of the Red Sea (see Subchapter 2.3). Ore taken by suction strongly depends on particle size and the grade of sediment consolidation, and probably also on additional techniques to loosen or break up the ore.

Air lift systems have the same limitations. They are used in shallow water diamond mining and have been employed in several pre-pilot mining tests in the Atlantic and Pacific Ocean to demonstrate their suitability for harvesting manganese nodules from the deep sea (see Subchapter 2.1). However, whereas air-lifting of phosphorites will be possible, different ore types may need additional techniques to prepare them for uplifting.

The cutting wheel technique has been employed in offshore diamond mining where gravel excavation is required (Garnett 1995). A 1.25m diameter wheel was used mounted on a remotely-controlled crawler cutting 1.5m deep furrows into fine to medium sheet gravels. Horizontally rotating steel tables up to 7m diameter, fitted with tungsten carbide-tipped drill heads were deployed down to about 200m depth, but they could be adapted as well for greater depths. These systems are able to penetrate and digest cemented gravel and even irregular bed rock. They penetrate the sea floor through the ore-bearing layers.

Bucket-ladder dredges have been successfully used, for example in tin and gold placer mining. The dredge vessel BIMA (Billiton Marine) was originally used off Indonesia for tin and later, off Alaska, for gold mining. Such a type of dredge is limited by length and maximum operating angle of its dredge ladder. Although the BIMA was the largest, having an 88m long ladder, it could work only down to about 40m of water depth, and the cut below the seabed was limited to 10m (Garnett and Ellis 1995, Garnett 1996). Thus the bucket-ladder system would allow mining only the shallow phosphorite deposits. Furthermore, this rigid system would be more vulnerable to higher sea state conditions than equipment suspended on a more flexible recovery pipe used in pumping or air-lifting the desired material.

The Continuous Line Bucket (CLB) system (see Subchapter 2.1) would only scrape off the material from the sediment surface to some uncontrollable depth. Although this technique was strongly advocated by Masuda (1985, 1987) for use in crust and nodule mining, little experience is available and doubt exists that the loop can be steered effectively. The CLB system is not expected to work efficiently, especially on the continental shelf with rapidly changing environmental conditions such as depth and sediment type or outcrops of basement rock.

Further potential mining techniques are the mechanical and the hydraulic collector systems or both combined in the form of a hybrid collector (see Subchapter 2.1). The mining vehicles are passively towed or actively crawl along the seafloor. The collector is carried on its front side and takes up the material mechanically or is aided by hydrodynamic forcing of the particulate material.

Among these different collector types, the basic technologies needed for phosphorite mining should be available. The variety of ore types, grain and nodules size, thickness of ore layers, amount of inert tailings, and degree of material consolidation will determine the most suitable technique for each minesite. Whereas manganese nodules may be harvested from the sediment surface with the aid of mechanical and/or hydraulic collector systems, a phosporite collector could be a sand mining system in the case of fine grained ore (grain size off Peru and Chile 125 - 500µm), a nodule collector (pebble size) or a scraper, cutter or breaker for more consolidated material.

Trawling has recently been introduced as a potential method for mining of manganese nodules off the Cook Islands. It has been suggested to use beam trawls having a rigid front frame and a foot chain of heavy steel to cause the nodules to bounce up in front of the trawls to facilitate harvesting (Bechtel Corporation 1996).

Another potential method for mining marine phosphorites not accessible to surface mining methods is application of borehole mining techniques in hydraulic slurry mining. This involves drilling a borehole into the sub-surface ore deposit, lining it with a suitable casing, and reducing the phosphates to a slurry form using a cutting tool fitted with a high speed water jet. The resulting slurry is then pumped out through the casing at a rate of 35 - 80 tons $\cdot$ hr^{-1} (McKelvey 1985, Marvasti and Riggs 1989). Although the borehole technique was first experimented with by several terrestrial phosphate mining companies in North Carolina during the 1960s, the technology was not further developed until renewed testing was conducted in north-eastern Florida in 1980. Such an approach could also be applied to mining phosphorites in shallow marine situations and may be extended into deeper localities.

2.5.3. Potential Effects of Mining

Inevitably, mining will impact the oceanic environment. The total destruction of the benthic system will be unavoidable, a near-bottom plume will be created by each of the different techniques requiring contact with and excavation on the seafloor, and plumes will develop in the water column. This is the same for all the different mining techniques, but the extent of their environmental disturbance will vary.

Compared to metalliferous resource mining as described in Subchapters 2.1 - 2.4, phosphorites will be mined from the upper reaches of the deep sea and from the continental shelf. This shallow mining depth introduces new aspects into the environmental considerations. All the actions occur rather close to the sea surface and even within the epipelagic zone with its primary production, its rather high standing stocks of all size classes of plankton and nekton, its fish breeding and nursery grounds, and its living resource production for human nutrition. This indicates that "shallow deep-sea" and shelf mining has the potential for greater interference with other environmental and commercial interests. This potential conflict situation needs to be recognised and carefully considered well before any mining activities are initiated.

All the mining techniques to be employed will destroy some life on and in the sea floor. Some of the organisms may escape immediate death by active migration or by resettling to the sediment surface after disturbance, but it may be assumed that their survival is not guaranteed because of the destruction of the normal sediment zonation and the nutritional base supplied by an undisturbed surface. The degree of disturbance varies with the mining technique utilised for sediment surface scraping (such as by trawling) to deep sediment excavation (such as by the cutting wheel technique). The latter will disturb the benthic system to a much larger extent depending on the depth of penetration below the sediment surface, the layer with strong changes of oxygen and nutrients, pH and Eh, with life in the upper part and fossil traces deeper down.

In all cases, a near-bottom plume will be created and its extent will depend on the technique applied. The rotating wheel could stir up a considerable amount of sediment. Depending on the grain size, much of the material may resettle relatively quickly, but other components may drift off with the currents for some unknown distance.

Phosphorites predominantly occur near or in upwelling areas where vertical mixing processes are stronger than in other ocean regions. Consequently, there will be more likelihood of the transport of fines into the productive surface water layer where they may impact organism of all size classes and the production of the system.

The transport of the mined material from the mine to the surface will leave plumes in the water column to various extents. Whereas closed transport systems (e.g. pump and airlift systems) may separate and reject fine particulate material close to the seafloor, transport to the mining platform would release no additional material from the pipe. In contrast, open transport systems like the buckets of the bucket-ladder or the CLB, and the trawl nets may release sediments throughout the path from the seafloor to the sea surface. The volumes of mined, but unwanted, sediments may be rather large and the different mining systems can not separate them from the phosphoritic material. Thus, much may be washed during hoisting through the water column and may turbidise the ambient water.

A further plume source will be the mined, but inert material which will be led back into the sea. All mining systems will produce these tailings and their volume will depend on the ability to separate this material while still near the sea floor. Brought back into the ocean, the coarser components will rapidly re-deposit on the sea floor, but the fine particulate material may drift for long periods and may adversely interact with the organisms. The extent of these impacts will certainly depend on the depth of tailings discharge. The closer to the seabed that the tailings are released, the less widespread will be their influence on the organisms and on the water column ecosystem.

Thus, as in the mining of metalliferous ores, sea floor and water column impacts must be considered. However, the rather shallow and continent-near occurrence of phosphorites gives the impacts another dimension of potential for interference. The euphotic zone may be impacted disturbing the production cycles, but additional conflicts may arise with other users of the oceans, specifically with the users of living resources. Fishery activities in many areas already penetrate down to 1000m water depth, occasionally even deeper, and cover all regions of potential phosphorite mining.

The combined sediment plumes may disturb the planktonic and nektonic food sources for the commercially valuable fishes. The particulate matter may clog the gills of larvae and may reduce the energy content of the food source, thus slowing down growth rates and production. With the disturbance of the sea floor, food resources and breeding/nursery grounds of demersal fishes may be lost. The recovery of the benthic community after mining may bring about a new species composition and community structure so that the original fish stock composition may be altered severely. Related to the impact conditions, a broad spectrum of adverse reactions can be hypothesised for the various potential mining areas.

2.5.4. Research Required to Evaluate the Effects of Mining

Before marine mining of phosphoritic ore starts, studies are required to evaluate the potential effects on the environment. Although the mining technology to be employed may not yet be known, and the eventual effects can only be estimated at best, precautionary considerations of disturbances are important. These will vary with the local conditions to an extent that no specific research plan can yet be recommended, but some general statements may be given.

For general effect evaluation, the ecosystem must be known and this should include an area much wider than the projected mining locality. Sediment transport and sedimentation will reach far beyond the direct impact field and may even leave the national economic zone and impact neighbouring ones.

As indicated above, phosphorites are found predominantly in upwelling regions where the Coriolis effect and the prevailing (trade) winds transport surface water towards the open sea, resulting in the transport of water from several hundred metres of depth to the surface. This nutrient rich upwelling water is responsible for high primary production followed in the food web by high standing stocks of pelagic and benthic animals as well as fishes. Upwelling regions are, therefore, considered to be among the most important fishing areas in the world, and the mining plumes may have a strongly disturbing influence.

An understanding of the general ecology of a mining area and of possible commercial conflicts with other existing, well established exploitation interests should be gained well in advance of any commercial mining. The necessary research for an analysis of the hydrographic system and the related ecology would remain within the normal range of oceanographic recording, observation and sampling programmes.

However, for the precautionary evaluation of the mining, even in the pre-mining phase, experimental work should be conducted on two different levels. Numerical modelling of the current system and of plume transport would not touch the ecosystem, but for the testing and validation of the models and for the prediction of effects, a disturbance experiment should be envisioned. It would be essential to conduct a large-scale experiment that allows extrapolation to full-scale mining. Numerical modelling should aid in experimental design and set the framework for creation of a large-scale plume which, in the practical phase, has to be monitored carefully in its extent in space and in its persistence over time, as well as in its effects on plankton and benthos. Intensive recording and sampling programmes must be conducted to follow the plume, to observe reactions of the plankton, to register the intensity of resedimentation in the near field of plume creation, and to look for short- and long-term effects upon the benthos.

No suitable natural event or artificial disturbance is known that would allow detailed evaluation of the environmental impact of phosphorite mining as would such a precautionary experiment. Since regions with mineable phosphorates have their own characteristic oceanographic settings, environmental impact assessments must be conducted for each mining area.

3. Wastes in the Deep Sea

The oceans have received waste materials from daily life for as long as human populations have settled along coastlines. At one time, all ships discarded their garbage, rubbish and other waste materials into the sea and much of it sank to the deep-sea floor. Clinker and unburned coal on the sea-bed mark the routes of steamships all over the oceans. Considerable tonnage is lost every year, mostly in coastal and shelf areas, but many other wrecks, including their fuel and cargo, are widely scattered over the deep sea-bed.

During the last four decades increasing pressure has been exerted on the oceans as a waste recipient resulting from the increasing human population of the world and concomitant industrial and technical developments. These wastes range from high volume, relatively low toxicity wastes such as sewage sludge and offshore installations, to relatively low volume, but potentially highly toxic wastes such as radioactive materials. Some wastes have already been disposed of in certain deep-sea regions, but further discharges are banned by international regulations.

However, waste disposal problems are not solved. The human population of the earth is currently approaching 6 billion and rising. It is predicted to exceed 10 billion before the end of the 21st century and possibly as early as the year 2050 (Merrick 1990), inevitably leading to simultaneous increases in industrialisation and the inherent problems of waste management. In developed countries the annual production of domestic waste alone currently amounts to 2 to 2.5 tonnes per capita, whereas in developing countries it is around 0.25 tonnes per capita (Spencer 1991). As more countries become industrialised, their average production of wastes can be expected to increase faster than their population growth.

Even at today's level of world population, if the area covered by ice-caps, deserts, agricultural land and managed forests is discounted, only 1 hectare of natural terrestrial ecosystem per capita remains (McAllister 1993). There must be a critical point at which this figure falls below the minimum needed to sustain the "ecological services" which are maintaining the qualities of land, water and atmosphere as well as biodiversity on the planet. In all aspects of our socio-economic activities, the protection of land and fresh waters, and the quality of the atmosphere, will have to be given a much higher priority. In particular, waste management practices must cease to degrade these environments further and, as a consequence, the high degree of protection being currently afforded to the oceans will probably be reduced as demands to exploit the oceans for waste disposal increase.

There will need to be minimisation of waste production, maximisation of recycling, continued use of land disposal (albeit greatly curtailed), destruction of toxic substances, cessation of manufacture of particularly troublesome compounds, and finally, consideration of the possibilities of deep-sea disposal. It is unlikely that there will be a single best solution to any waste management problem since the environmentally least-damaging option will vary according to the geographical and socio-economic context in which the waste is generated. But because of the size and assimilative capacity of the oceans, there is a *prima facie* case that their use for the disposal of at least some kinds of wastes may result in an overall reduction in environmental impact compared with the continuation of presently accepted methods of disposal on land.

Some of the waste types which may be considered for deep-sea disposal have already been dumped (munitions, radioactive wastes, and sewage sludge) while others came into discussion only recently (offshore installations, dredge spoils, and carbon dioxide). Therefore research needs will vary, and the available experience may be applied, with modifications, to other materials.

As in the previous chapter, Chapter 3 is divided into a number of subchapters, each reviewing a major waste material category which has already been dumped into the sea or is under consideration for ocean disposal. Subchapter 3.1 reviews the dumping of munitions and related war materials, while Subchapter 3.2 examines the disposal of radioactive wastes. Disposal of large offshore structures is discussed in Subchapter 3.3. Depositing sewage sludge and dredge spoils into the oceans is addressed in Subchapters 3.4 and 3.5, respectively, while Subchapter 3.6 looks at the potential for using the sea to reduce the vast quantities of carbon dioxide released into the atmosphere by industrialised societies.

The single subchapters are structured similarly to those on potential mining of the deep sea: (1) description of the wastes, (2) their transport and disposal techniques or options, (3) the expected impacts on the oceanic environment, and finally, (4) the research that will be needed to assess the acceptability of the ocean disposal options, considered to be *prima facie* cases for undertaking research programmes.

The problems of environmental acceptability of the research proposed will be discussed in Chapter 4 together with the research activities assumed necessary to evaluate environmental mining impacts.

3.1. Munitions

3.1.1. Description of Waste Materials

For over two centuries, sea disposal of defective, obsolete and surplus munitions has been regarded as an acceptable practise (Länder Government Working Group 1993) although presumably not much thought was given to the environmental problems involved. The vast majority of the dump sites have been either in deeps on the continental margin or in close proximity to the continental slope. There are many shallow areas in European waters where war materials have been dumped in the past. The dumping often occurred at imprecisely known positions, partly because at the time navigational techniques were poor, but also because of inadequate or lax control of the dumping procedures. Moreover, poor records seem to have been kept of precisely what was dumped. In addition to the munitions which were dumped purposely, many other munitions and war material have finished up on the seabed through losses from wrecked naval vessels, from firing ranges or as unexploded devices left over from hostilities. This report is restricted to the at-sea disposal of predominantly British and German munitions to demonstrate the extent of these activities, but other countries have certainly also dumped additional war materials. The majority of reported dump sites are rather close to shore and in relatively shallow waters. It seems that most of the dumping activities were kept confidential and that for many years the world-wide military forces were exempted from the regulations of the London Dumping Convention (LDC). In UK waters despite the dumping of military equipment being banned under the Dumping at Sea Act 1974, except under emergency conditions, devices are continually being washed or trawled up. For example, between 1973 and 1977, no less than 10,301 articles were made safe by British Royal Naval Explosive Ordnance Teams, and many other European countries have encountered similar problems. Table 3.1.1 lists the devices made safe by these Royal Naval teams in 1982-83.

In addition to environmental impacts, disposal and loss of military equipment and munitions at sea, particularly in coastal waters, presents problems ranging from risk to human safety and resources, and the security problem of ensuring that none of the material and equipment is reusable. Chemical weapons present special difficulties. Since the use of the most dangerous chemical weapons has been banned under international convention and others have become dangerously unstable, stocks of these materials have had to be disposed of and were in part dumped at sea. After the last World War, the Allies were faced with the problem of disposing of large quantities of weapons and munitions including chemical weapons captured from the Axis powers. This was mostly undertaken by disposal at sea, in areas which at the time were considered to be away from fishing grounds and where there was no potential for future industrial activity. This has left a large and dangerous legacy that now hinders other legitimate uses of the sea.

In the autumn of 1995, a 60cm deep trench was ploughed between Scotland and Northern Ireland just to the north of a feature known as Beaufort's Dyke in preparation for the laying of a gas pipeline. Despite passing well clear of an area previously designated as a munitions dumpsite, the trenching operation appears to have resulted in about 4500 decaying 30 pound (13.6kg) incendiary devices being washed up on the west coast of Scotland, beginning on 6 October 1995 (Anon. 1995). The incendiaries, dislodged from the sea-bed, floated to the surface and were then blown by the prevailing winds into the Firth of Clyde. Once beached and dried, the phosphorus spontaneously ignited.

Table 3.1.1 Explosive articles made safe by Royal Naval Explosive teams in 1982-1983. Region 1 is the UK coastline from Portland to Hull; region II from Hull to Barrow-in-Furness (including South-west Scotland) and region III from Barrow-in-Furness to Portland.

Article		**Region**	
	I	II	III
Bombs < 226.8kg	6	-	2
> 226.8kg	-	1	-
unspecified	-	2	1
Depth Charges	-	2	1
Hand Grenades	1	-	1
Pyrotechnics*	367	1256	581
Rockets	1	-	-
Shells < 76.2mm	12	2	1
> 76.2mm	26	-	-
unspecicified	6	1	14
Torpedoes	1	11	3
Others	2	8	7
TOTALS:	422	1283	611

* Pyrotechnics are either flares or devices producing smoke or flames used during military exercises.

From the end of the last World War until around 1976, Beaufort's Dyke - a 50km long feature, 5km wide and about 250m deep - was used extensively as a dumpsite for a reported 1.17 million tonnes of chemical and conventional weapons, and may even have been used as a munitions dump as early as 1920. The munitions disposed of included 14,000 tonnes of 5-inch artillery shells filled with phosgene gas which were dumped in autumn 1945. A further 135,000 tonnes of conventional munitions were disposed of during the next three years, and then each year a further 20,000 tonnes were discarded, until the late 1950's when the quantities dumped began to decline, falling to 3,000 tonnes a year by the early 1970's (Edwards 1995). Munitions dumping at this site ceased in 1973 apart from an emergency dump of a small number of 40mm shells in 1976. Over the past five years about 700 anti-tank grenades have been washed up on the coast of the Isle of Man and Northern Ireland.

A recent preliminary report describes the results of a survey carried out in that region in Spring 1996, using side-scan sonar, towed underwater television, and grab and trawl samples (SOAEFD 1996). There was no contamination of sediment or faunal samples (including such commercially important seafood as cod, haddock, whiting, hake, plaice, monkfish, queen scallops, and scampi) that could be attributed to the disposal of munitions. However, the side-scan sonar and television survey revealed large numbers of munitions and boxes, presumably containing other munitions, lying on the seabed not only in the area designated as a dumpsite, but also strewn along the track the vessels followed on the way to and from the dumpsite. The majority of the boxes were less than one metre in length, but some were considerably larger. Munitions were seen loose, and packed in crates or open racks. The loose material included small arms cartridges, small canisters, land mines, cylinders, metal drums, shells, mines, hand grenades, depth charges and rockets. Many were in an advanced state of decay and the majority had been colonised by a variety of sessile organisms.

Other disposal of munitions in UK waters have involved the scuttling of 24 redundant ships packed with a reported 137,000 tonnes of chemical weapons, at three localities: at at depth of 4500m some 625km south-west of Land's End, 100km north-west of Northern Ireland, and 80 - 160km to the west of the Hebrides in and at the edge of the Rockall Trough (500-3000m) (Ukooa 1996). In 1956 for example, during Operation Sandcastle 3 ships loaded with 250kg nerve gas bombs containing "tabun" were scuttled at the edge of the Rockall Trough (about 500m depths), 112km west of Bloody Foreland, County Donegal in Northern Ireland (ACOPS 1988, Hencke 1995). These disposals not only included 17,000 tonnes of German bombs filled with "tabun", but also other munitions containing sarin, phosgene, mustard gas and tear gas. On release sarin, phosgene and tabun are quite rapidly hydrolysed in sea-water into relatively harmless by-products. Mustard gas, however, can be more persistent since it is comparatively insoluble and is heavier than water, so it can form an oily layer on the seabed (HELCOM CHEMU 1994). Fishermen working over one of the Baltic dumpsites are reported to have been severely burnt by mustard gas although this report was not confirmed by Theobald and Rühl (1994). In the Firth of

Clyde and the North Channel between Scotland and Northern Ireland trawlers frequently drag up munitions.

In September and October 1966 several underwater explosions were reported by merchant vessels in the vicinity of Beaufort's Dyke. Since 1982, in this region 25 explosions have been detected, 14 occuring in 1995 alone. Elsewhere, seismic monitoring has registered 200-400 underwater explosions per year in Britain's offshore waters, about the same number as there are earthquakes (Browitt 1995), though most of these are thought to be the direct or indirect result of military activities rather than of dumped material.

In the Baltic, 42,300 to 65,300 tonnes of chemical munitions containing up to 10,000 tonnes of chemical agents were dumped by the Allies to the south of the Little Belt (30m), around Bornholm and south of Gotland (70 - 105m) (Theobald and Rühl 1994). The chemical agents produced by Germany during World War II included (HELCOM CHEMU 1994):

Tear gas	chloroacetophenone
Nose and throat irritants	Clark I, Clark II, Adamsite
Lung irritants	phosgene, diphosgene
Blister gases	mustard gases, Lewisite
Nerve gases	tabun
Additives	monochlorobenzene

After World War II, about 296,000 tonnes of chemical munitions and warfare agents were found on German territory. Records of what happened to these materials, of how they were disposed of (on land or in the sea) or destroyed, are poor and incomplete. An estimated 34,000 tonnes of chemical munitions, containing about 12,000 tonnes of chemical agents were dumped to the east of Bornholm and near Gotland (70 - 120m) by the Soviet Authorities, and a further 200 - 300 tonnes, discovered after 1952, were dumped by the German Democratic Republic (GDR). In addition there are unconfirmed reports that in 1946 vessels containing a further 15,000 tonnes of chemical munitions were dumped south-west of Rønne (Bornholm) (20 - 40m) and in 1956 a further four decommissioned coastal patrol boats each loaded with around 50 tonnes were scuttled in the same area. Most of the munitions dumped off Bornholm were loaded at the Baltic port of Wolgast (Theobald and Rühl 1994). Numerous witnesses reported (Länder Government Working Group 1993) that dumping of the chemical munitions started while the vessels were still en route to the dumpsite.

In 1960, about 69,000 shells containing tabun were recovered from the entrance to Little Belt (to the south-east of Als), set in concrete blocks and dumped west of the Bay of Biscay. However, about 5000 tonnes of phosgene and nerve gas bombs remained, probably now well covered with sediment at a water depth of about 30m. In 1964, 462 tabun shells were recovered from Wolgast Harbour (Baltic), set in concrete and dumped at a depth of 3100m in the Norwegian Sea (64°42' N, 1°36' W) (HELCOM CHEMU 1994).

Between 1945 and 1948 the Allies loaded up to 35 confiscated German merchant ships with an estimated 130,000 tonnes of conventional and chemical weapons and sank all but one in the Skagerrak 25 nautical miles south-east of Arendal (depth not given) in the Norwegian Channel. Another ship was sunk in the Norwegian Sea (3100m). An additional group of nine vessels containing an estimated 20,000 tonnes of munitions including weapons containing mustard gas were sunk in an area west of Måseskär Lighthouse (200m) in the Skagerrak (at 55°07' N, 10°47' E). Another two were sunk close to Skaw in the Skagerrak containing about 1,500 tonnes of chemical munitions (HELCOM CHEMU 1994).

Munitions continue to be disposed of in the deep ocean. In 1994 the Portuguese Government notified the Oslo Convention that it would sink a vessel loaded with nearly 2,200 tonnes of a wide range of redundant munitions (including a variety of grenades, chemical grenades, ammunition, fuses, pyrotechnics, depth charges, rockets, bombs, missiles and mines). The vessel "modified specifically for the transport of ammunition during the Gulf War" was scuttled in 4000m water depth close to 39° N, 14° W at the very edge of the Portuguese EEZ (SEBA 1995). It should be remembered, however, that although such dumps may be made within the bounds of a given country's EEZ, bottom currents will undoubtedly transport substances freed by the breakdown of dumped materials well beyond their original disposal sites.

In November 1994 the South African authorities notified the LDC (IMO 1994) that it had issued a permit for 77,863kg of obsolete ammunition to be dumped at 33°30' S, 15°30' E about 150nm (275km) south-west of Cape Town at a depth of 3,500m. Although alternative disposal methods were available then, the munitions had been packaged in blocks of concrete. A further 14,000 tonnes of munitions (including, bombs, shells, cartridges, mines, mortar bombs, grenades and smoke devices) were licensed to be dumped at an unspecified depth near 30°28' S, 31°21' E, some 40nm (73km) south-east of Durban. In the notification it was pointed out that despite "a request for technical assistance on alternatives to sea disposal, information is needed on the comparative environmental effects of land-based and sea-disposal options for obsolete ammunition ..... very little information has been forthcoming".

3.1.2. Description of Disposal Techniques

There have been no special disposal techniques employed for many of the munitions other than the simple method of dropping boxes off ships. For the chemical weapons the materials were sealed in the holds of redundant ships which were then scuttled. The shells containing nerve gas recovered from the Baltic were embedded in concrete before re-disposal in deeper water. Otherwise little effort was made to ensure the materials were isolated from the sea-water. Even more serious is the fact that much of the dumping was poorly supervised and considerable amounts of materials were dumped outside the confines of the designated sites. This was amply demonstrated by a video-survey of the Beaufort's Dyke site, which revealed munitions strewn along the ships' tracks to the site, and the finding of war materials in shallow waters in the Baltic between the port and the dumpsite. Similarly, war materials have been recovered along the routes from the loading ports to the dumpsites in the Baltic, and there are eye-witness accounts of materials being thrown over the side well outside the dumpsites either during bad weather or to ensure scheduling targets were achieved for the disposal (Länder Government Working Group 1993).

At present the main alternative method for the disposal of munitions is by controlled incineration in rotary kilns (See SEBA 96/8/2), and it appears that this method is generally accepted. At least the signatory states of the IMO currently refrain from further munition dumping. Therefore, we conclude that there is no rationale for ocean disposal continuing to be an option.

3.1.3. Impacts of Disposal

Dumped munitions clearly pose a threat to other users of the sea. Fishermen trawling on the sea-bed are particularly at risk in shallow water and munitions are recovered regularly in certain areas, including the Baltic. If munitions have been dumped where they can eventually be washed inshore, as has happened with waste material dumped in the Barrow Deep in the Thames Estuary (20m water depth with tidal currents of up to $1.4m \cdot s^{-1}$), they will represent quite a serious risk to members of the general public and to divers. The spontaneous underwater explosions that have been recorded in UK waters pose a potential threat both to human safety and to marine organisms. Relocation by water currents of war materials appears to have occurred from around the site following disturbance during the laying of a pipe-line. Most dumping of munitions seems to have occurred at depths that have been sufficient to make relocation by natural processes unlikely, but fishing activity can result in the material being dredged up and then dropped back outside the dumpsite.

Chemical warfare agents pose threats which vary with their specific characteristics:

Chloroacetophenone hydrolyses in water slowly but can be biodegraded to non-toxic substances in sea-water. This tear gas is probably of relatively minor concern.

Clark I and II are arsenical substances that degrade slowly in sea-water to form tetra-phenyldiarsine oxide, either hydrochloric acid or hydrogen cyanide (Clarke II). The acids are rapidly neutralised or detoxified by the buffering action of the sea-water. The arsenical organic residue will probably biodegrade leaving an inorganic arsenical residue (200g of Clark contain 75g of arsenic). This is potentially environmentally dangerous if it is accumulated in any way. If it is dispersed, its concentrations will soon be lost in the background concentrations. Abdullah *et al.* (1995) report total arsenic levels of 10-15nm $\cdot$ l^{-1} in the surface waters of Oslofjord and up to 25nm below the pycnocline (1nm = 0.075µg), and summarise literature reports giving ranges of total arsenic of 1 to 53nm $\cdot$ l^{-1} for estuaries and open ocean stations. These probably pose only a slight environmental threat and are only dangerous to humans if an intact device is recovered.

Adamsite has low solubility in water and is very slowly hydrolysed to phenarsazinic acid and hydrochloric acid. These will pose similar threats to those from Clark compounds.

Phosgene is an extremely reactive compound which is rapidly broken down in sea-water to hydrochloric acid and carbon dioxide. Its environmental impact will be highly localised and ephemeral.

Mustard gas in its raw form is a relatively insoluble substance and the addition of thickeners has rendered it far less soluble. Even though mustard gas is broken down slowly in solution in sea-water, its main danger lies in its long term persistence as lumps and crusts on the seabed. Its main danger is to fishermen who trawl up lumps in their nets and then get severely burnt through inadvertently handling it. Mustard gas easily penetrates rubber, leather, textiles, wood and concrete as well as human skin. Symptoms of itching and burning appear some time after penetrations of the skin, and then painful blisters erupt. Secondary infections of burst blisters are common as the mustard gas reduces resistance to infection. Inhalation causes severe breathing difficulties and large doses can lead to death. Ten of thirteen incidents, reported (1957-1991) by German fishermen operating in the vicinity of Bornholm, involved burns from mustard gas. Nearly 850 Danish fishermen work in the Baltic and since 1976, there have been 439 incidents reported when contaminated catches have had to be destroyed. Up until 1990, there were between 5 and 48 incidents reported each year, but in 1991 the number rose inexplicably to 101.

Lewisite reacts with water to form chlorovinyl arsine oxide which breaks down to form arsinic acid and acetylene. The acetylene presents no threat. The possible impact of the arsinic acid is much the same as any arsenical compound.

Tabun dissolves rapidly and is rapidly broken down to phosphoric acid and hydrogen cyanide which is in turn rapidly converted to sodium formate. Hence this material presents no long term danger to the environment, but could pose a serious risk if a shell or container was inadvertently handled by a fisherman.

Cyclone B consists of cyanide salts which are highly soluble and rapidly broken down.

Munitions that were well packed prior to disposal pose no immediate threat unless they are inadvertently recovered and improperly handled, though additional danger comes from some munitions being under internal pressure. Munitions left on the sea-bed and exposed to the sea-water will eventually corrode through. Corroded casings have been recovered which no longer contain any traces of their original contents. In 1989 an investigation conducted by the Research Institute of the Norwegian Ministry of Defense investigated the state of munitions loaded in vessels scuttled in the Skaggerak (Tornes *et al.* 1989). Most bombs in the wrecks were intact but a few were perforated by corrosion. No traces of chemical agents were detected in the water or sediment around the wrecks. No evidence has been found of fish being poisoned by warfare agents.

There seems to have been little considerations given to the impact of any heavy metals contained in the weaponry or the scuttled vessels in which they have often been dumped. While it is not envisaged that these materials will present any significant environmental threat in the deep ocean, monitoring is needed to verify this conclusion.

3.1.4. Research Required to Evaluate Impacts

There appears to be no valid argument to consider further disposal of munitions in the deep ocean since adequate techniques are available for the disposal of redundant munitions with minimal environmental risk on land. Thus, no field experiment is necessary to investigate the future impact of weapons and munitions. Laboratory studies, using material from shallow water dump sites, may be needed to examine the direct and indirect impact of some of these materials, for example, in determining if there is any evidence that the arsenical breakdown products produced by the degradation of the Clark tear gasses will bioaccumulate. It is possible that much of the information needed is already available, but is classified. In this case any non-public information that is directly relevant to environmental assessments of material dumped in the ocean should be made available.

However, monitoring studies of dumping sites are urgently needed. For oceanographers, it is difficult to conceive that large impacts such as the dumping of munitions, either in small sealed containers or by scuttling of loaded ships, is carried out with secrecy and seemingly never properly monitored. Indeed, as far as we know, the vessels sunk by British military authorities after World War II in the North Atlantic have never been monitored at all. The Natural Environment Research Council "Scientific Group on Decommissioning Off-shore Structures" (NERC 1996) stated: "Military waste has been dumped at a number of sites in the deep ocean. Much of the waste was disposed of at sea apparently because it was perceived as being toxic, but the nature, quantities and positions of such disposals are not generally available. Further work on such sites could be useful if the dispersal of toxic material in the deep ocean, and its impact on local fauna, is to be understood". Munitions dumping subjects the deep-sea floor to large-scale impacts, and these should be carefully examined and monitored. The scuttling of the Portuguese vessel should have been exploited as an analogue of large-scale experiments for risk analysis. It would have helped in evaluating the potential effects of dumping large structures, e.g. the Brent Spar. Many other impacts of similar extent exist and a certain number should be revisited to obtain a general overview on the degree of impact. Some should then be carefully studied and monitored, to assamble basic knowledge for this mode of using the deep sea and for risk assessments. All available information should be declassified and made available to scientists and the general public.

3.2. Radioactive Wastes

3.2.1. Description of Waste Materials

The history, development, controls and scientific activity associated with the disposal of radioactive wastes in the NE Atlantic have been described in detail and summarised in various documents. European Nuclear Energy Agency (ENEA) (1968), Nuclear Energy Agency (NEA) (1980, 1981, 1983a,b, 1985, 1986a,b, 1989, 1990), Holliday (1984) and Her Majesty's Stationary Office (HMSO) (1986) essentially concern the dumping of low and intermediate level wastes; NEA 1984, 1988a-h cover the feasibility studies associated with the disposal of high level wastes. There are numerous other sources ranging from general overviews (NEA 1982, Saunders 1992), annual reports of national groups (e.g. the Radioactive Waste Management Committee (RWMC)), ethical considerations (NEA 1995a), annual updates on waste management policies and programmes (e.g. NEA 1995b), evaluations of research programmes (e.g. Guillaumont *et al.*, 1989), consultation documents (e.g. Department of the Environment (DoE) 1994) and no doubt hundreds of individual papers and reports of specific issues by individual authors in many different countries. This review attempts to summarise the general principles. It is clearly not an exhaustive treatment of the complex issues involved. Before describing the various types of radioactive waste and the relevant definitions it is appropriate to outline the process of radioactive decay since some of the subsequent waste categories depend upon the type of energy emitted by the waste.

During radioactive decay atomic nuclei disintegrate and may emit various forms of radiation. These can be as alpha or beta particles or as gamma rays. *Alpha (α) particles* are positively charged particles emitted at high speed from the nucleus. They are easily stopped, e.g. they cannot penetrate the skin, but are long lived and can be very harmful if taken into the body by swallowing or breathing. *Beta (β) particles* are negatively charged, and are smaller, lighter and faster than α particles. They can pass through skin but can be stopped by thin sheets of lead or aluminium. *Gamma (γ) rays* may be emitted during the emission of α and β particles. They are short wavelength rays which are very powerful and require thick sheets of lead and many feet of concrete or water to stop them.

Alpha waste is predominantly associated with nuclear fuel processes and the production of specific radionuclides. These processes produce some uranium isotopes but mainly the transuranic nuclides plutonium and americium. Beta/gamma waste has 4 main sources. Reprocessing nuclear fuel gives plutonium-241; nuclear power plant operations create mixed fission products, especially strontium-90 and caesium-137, and activation products, mainly iron-55, cobalt-58 and -60. Production of specific nuclides for research creates carbon-14 and iodine-125, and decontamination and decommissioning of redundant nuclear plants gives rise to a range of nuclides, mostly as surface contaminants on building rubble, etc. Beta wastes include tritium, which since about 1974 has been estimated separately (see below). Tritium waste arises from the production and use of labelled compounds for medical research and industrial applications; it occurs in a variety of inorganic and organic forms, is highly soluble and of low radiotoxicity relative to other nuclides.

Radioactive wastes arise from nuclear electricity generation, from the use of radioactive materials in medicine, agriculture, industry and research, and from military nuclear programmes. All member countries of the EU produce such wastes in varying quantities, as summarised in Saunders (1992). Different types of wastes are classified into low, medium (intermediate) or high level waste according to their composition, concentration of radioactivity and whether or not they generate heat as they decay. The differences between these categories vary slightly between different countries but can be described broadly as follows:

Low level - waste containing low concentrations of radioactive material which are so low as to require no special shielding and can be handled with simple protectors like rubber gloves. Typically such wastes includes paper towels, clothes, used syringes, air cleaning filters, and similar materials used in medical research or practice.

Intermediate level - waste containing higher concentrations of radioactive material than low level waste and require shielding, usually by metal or concrete, and so need to be handled using special techniques. Typically it consists of metal scrap, sludge, resins and used radioisotope sources from nuclear power stations and industrial activities.

High level - waste with high concentrations of radioactive material such that the intensity of radiation causes the waste to become physically hot. Such wastes require cooling, shielding and remote handling. They arise from reprocessing plants and as ash from the burning of nuclear fuel.

The above descriptions are neither quantitative nor specific. Since 1965 the disposal of radioactive wastes into the NE Atlantic was co-ordinated by the ENEA (later the NEA), and an international group of experts concluded that the disposal of solid radioactive waste on the bed of the Atlantic at a rate of approximately 10,000 Curies $\cdot$ yr^{-1} would "give rise to intakes by man of radioactivity many orders of magnitude below those currently recommended as permissible by the International Commission on Radiological Protection (ICRP)". They further concluded that there would be "no effects on biological organisms, except perhaps for a few individual members of species in the immediate vicinity of the disposal area" (ENEA 1968). This voluntary regulation developed into an international dumping operation under the auspices of the NEA and was later subsumed into the Convention on the Prevention of Marine Pollution from Dumping of Wastes and other Matter, i.e. the London Dumping Convention (LDC) in 1972. This treaty was ratified in 1975, and uses the International Atomic Energy Agency (IAEA) for the provision of scientific advice on radioactive wastes. The LDC prohibits the dumping of high-level wastes as defined by the IAEA (1978), but allows dumping of other radioactive wastes subject to a range of restrictions and the issue of special permits by the competent national authorities.

However, in 1983 a non-binding resolution of the Contracting Parties to the LDC introduced a voluntary moratorium on the dumping of low level radioactive wastes at sea. This was followed by a ban in November 1993 when, at the 16th Consultative Meeting of the Contracting Parties to the LDC, a prohibition on sea disposal of low level radioactive waste was adopted by a vote of 37 to 42. Five countries abstained - Belgium, China, France, the Russian Federation and the UK (through the UK abandoned this option in September 1997). The ban on low-level radioactive waste ocean disposal took effect from February 1995. The Russian Federation in not accepting the amendment to the Convention, undertook to "endeavour to avoid pollution of the sea by dumping wastes and other matter" (Significantly, the 16th Consultative Meeting received a report by the Russian Federation (Yablokov *et al.* 1993) on the extent of disposal at sea of radioactive materials carried out by the former USSR and requested at the 15th meeting; knowledge in the west of the extent of this activity had been previously unsubstantiated). The amendment also states that "after 25 years the Contracting Parties shall conduct a scientific study into the effects of radioactive substances in the marine environment and review the position of low-level radioactive wastes ...... in the light of the study findings".

The IAEA definition of high-level waste is based on radiological protection principles and sets out to define high-level waste quantitatively so that dose limits for members of the public from such waste are not exceeded. The definition is in terms of units of concentration in the waste, assuming an upper limit to mass dumping rate of 100,000 $t \cdot yr^{-1}$ at a single site; and because of the impossibility of ensuring long term containment in packages the concentration refers to the concentration in the material as it is dumped. The 1978 definition is:

"...... high level radioactive wastes or any other high-level radioactive matter unsuitable for dumping at sea means any waste or other matter with an activity per unit gross mass (in tonnes) exceeding:

- 1 Ci/t for α emitters but limited to 10^{-1} Ci/t for ^{226}Ra and supported ^{210}Po

- 10^2 Ci/t for ß/γ emitters with half-lives of at least 0.5 years (excluding tritium) and ß/g emitters of unknown half lives; and

- 10^6 Ci/t for tritium and β/γ emitters with half-lives of less than 0.5 years. The above activity concentrations shall be averaged over a gross mass not exceeding 1000 tonnes (see NEA 1985 for further details)".

Radioactive wastes disposed of in the NE Atlantic between 1949 and 1982 were not in the high level category. Their concentrations of radioactivity were lower than those specified by the IAEA. To demonstrate compliance with the regulations, from 1967 countries submitted waste inventories to the NEA. Between 1967 and 1974 the waste was divided into two categories: α activity and β/γ activity including tritium; from 1977 onwards the amount of tritium activity was recorded separately, but had been estimated between 1974 - 1977.

Dumping occurred in the NE Atlantic between 1949 and 1982. The total amounts dumped are shown graphically in Figure 3.2.1 and as Tables 3.2.1 - 3.2.6. These tables are summarised in Table 3.2.7. To these European data must be added the waste dumped by the USA in the NW Atlantic. Johnson *et al.* (1984) report that the total amount of mixed activity at the time of packaging was $3.7 \cdot 10^{15}$Bq, about 95% of which was disposed of at a position 38°30' N, 72°06' W in a depth of 2800m. (1Bq = $27 \cdot 10^{-12}$Curies). A further addition is the dumping in the seas bordering the former Soviet Union and reported in Yablokov *et al.* 1993. Further disposals of radioactive wastes occurred in the Pacific Ocean (Dyer 1976).

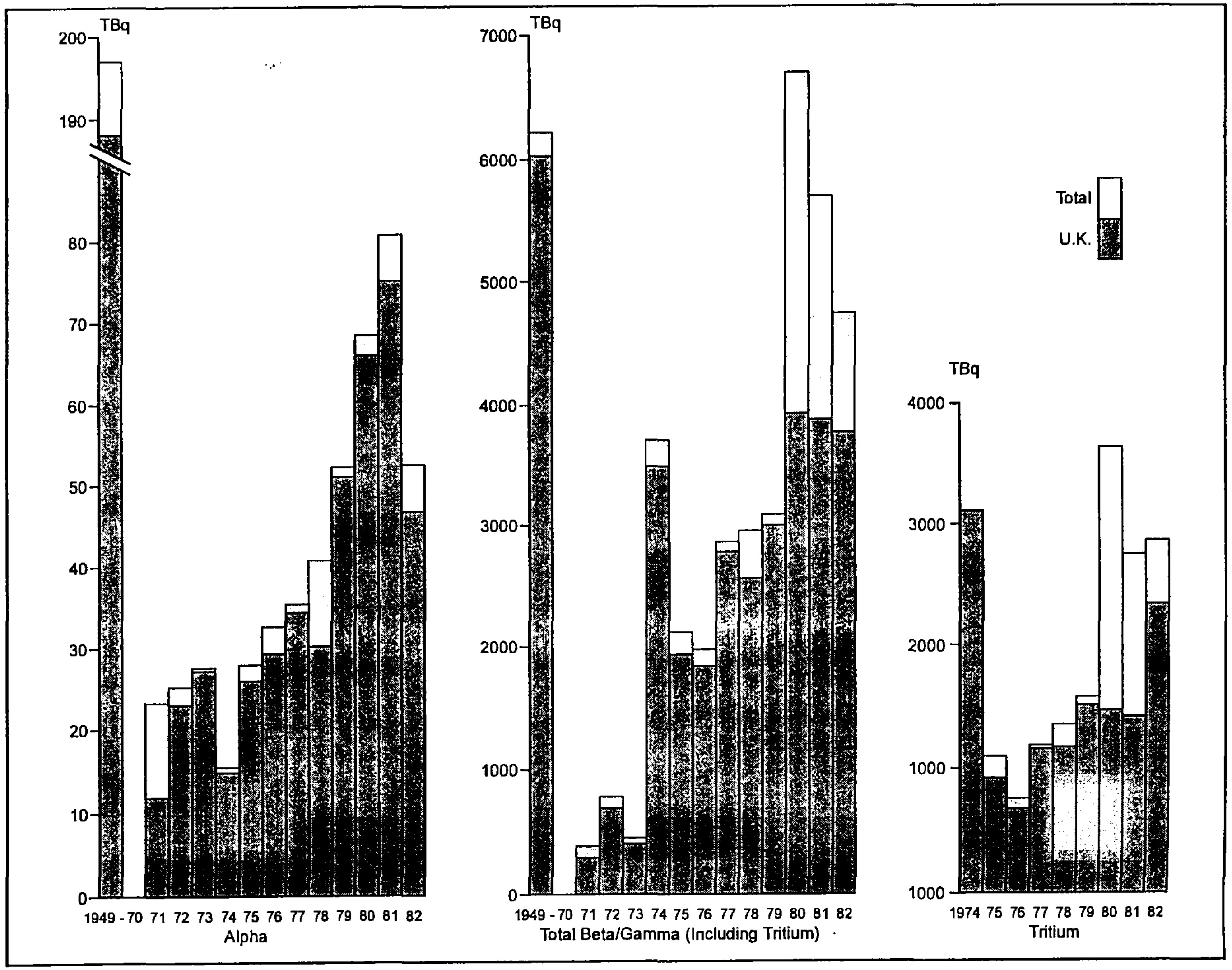

Figure 3.2.1 Quantities of radioactive waste dumped by year (from Holliday *et al.* 1984)

Table 3.2.1 Dumping in the NE Atlantic under NEA/ENEA supervision, 1967-82. (Belgium) [From NEA, 1985]

Year	Gross mass (tonnes)	Radioactivity (TBq) Alpha	Beta-gamma (excl. Tritium)	Tritium
1967	600	-	7.03	-
1969	600	-	17.95	-
1971	1768	11.10	13.69	66.34
1972	1112	0.04	1.11	70.30
1973	2250	-	2.96	32.19
1975	2001	1.37	8.47	111.93
1976	2243	2.70	23.42	51.80
1978	3671	9.44	47.03	112.89
1979	871	1.17	5.96	34.82
1980	3512	2.28	592.00	64.59
1981	4450	4.55	332.56	8.81
1982	5100	5.65	237.49	3.30

Note: This table refers to dumping in the deep waters of the NE Atlantic off the continental shelf. It is based on NEA/ENEA records. The number of digits quoted does not give an indication of the accuracy of the data.

Table 3.2.2 Dumping in the NE Atlantic under NEA/ENEA supervision, 1967-69. (France, F.R. Germany, Italy, Sweden). [From NEA, 1985]

Country	Year	Gross mass (tonnes)	Radioactivity (TBq) Alpha	Beta-gamma
France	1967	9184	5.96	213.01
	1969	5015	2.52	131.91
Germany	1967	180	0.02	0.18
Italy	1969	50	0.07	0.11
Sweden	1969	1081	0.93	2.04

Note: see Table 3.2.1

Table 3.2.3 Dumping in the NE Atlantic under NEA/ENEA supervision, 1967-82. (Netherlands). [From NEA, 1985]

Year	Gross mass (tonnes)	Radioactivity (TBq) Alpha	Beta-gamma (excl. Tritium)	Tritium
1967	207	0.003	0.07	-
1969	303	0.01	1.00	-
1971	360	0.01	0.74	-
1972	626	-	2.03	-
1973	657	-	1.85	-
1974	501	0.04	0.63	20.35
1975	901	0.06	3.33	14.80
1976	1911	0.04	32.63	3.70
1977	3015	0.33	14.76	7.10
1978	1562	0.19	39.41	17.46
1979	2122	0.05	19.86	11.40
1980	1885	0.02	16.09	3.85
1981	2063	0.24	65.53	3.07
1982	3049	0.12	37.58	17.70

Note: see Table 3.2.1

Table 3.2.4 Dumping in the NE Atlantic under NEA/ENEA supervision, 1969-82. (Switzerland). From NEA, 1985]

Year	Gross mass (tonnes)	Radioactivity (TBq) Alpha	Beta-gamma (excl. Tritium)	Tritium
1969	244	-	2.62	-
1971	375	0.07	12.95	-
1972	508	0.22	14.80	7.10
1974	508	0.59	142.12	67.41
1975	202	0.89	26.83	15.73
1976	349	0.56	9.36	17.50
1977	450	0.70	24.57	13.54
1978	733	1.00	150.66	14.43
1979	409	0.003	5.14	-
1980	301	0.06	27.64	1876.31
1981	404	0.57	72.89	1324.86
1982	847	0.29	147.09	499.43

Note: see Table 3.2.1

Table 3.2.5 Dumping in the NE Atlantic, 1949-70. (UK and Belgium). [From NEA, 1985]

Country/year	Gross mass (tonnes)	Radioactivity TBq Alpha	Beta-gamma
UK			
1949	9	-	0.04
1951	33	0.04	0.19
1953	57	0.07	0.07
1955	1453	0.44	1.22
1957	4404	35.33	29.90
1958	2715	25.71	40.15
1961	4361	20.83	60.31
1962	253	0.63	6.03
1963	5809	13.62	261.63
1964	4392	16.43	558.33
1965	1760	4.22	508.90
1966	1044	2.89	101.45
1968	3164	27.05	2768.97
1970	1674	8.62	748.29
Belgium			
1963	789	0.04	2.22

Note: see Table 3.2.1

Table 3.2.6 Dumping in the NE Atlantic under NEA/ENEA supervision, 1967-82. (UK). [From NEA, 1985]

Year	Gross mass (tonnes)	Radioactivity (TBq) Alpha	Beta-gamma (excl. Tritium)	Tritium
1967	724	3.37	62.223	-
1969	1885	14.43	650.83	-
1971	1465	12.03	318.75	-
1972	1885	24.94	704.81	-
1973	1443	27.38	431-42	-
1974	1256	14.76	3482.66	-
1975	1350	26.05	985.68	956.08
1976	2269	29.19	1148.74	693.01
1977	2140	34.41	1609.57	1159.14
1978	2080	30.12	1354.46	1209.90
1979	2014	51.10	1483.29	1516.67
1980	2693	66.27	2438.67	1486.25
1981	2517	75.18	2459.24	1414.99
1982	2697	46.77	1410.77	2345.17

Note: see Table 3.2.1. Prior to 1975, tritium activities are included the beta-gamma figures

Table 3.2.7 Locations of dumping areas and quantities of waste dumped in the NE Atlantic, 1949 - 1984 (No dumping after 1982).

	Approximate location		Annual quantity dumped					
				Activity (Ci)				
Year	lat. °N	long. °W	mass [tonnes]	α	β/γ	^{3}H	NEA supervision	Countries supplying wastes
1949	48°30'	13°00'	9	0	1			UK
1950	49°50'	2°18'	350	2	20			UK
1951	49°50'	2°18'	319	1	18			UK
	55°26'	11°20'	33	1	5			UK
1952	49°50'	2°18'	534	2	29			UK
1953	55°08'	12°10'	57	2	2			UK
	49°50'	2°18'	758	10	39			UK
1954	49°50'	2°18'	1145	23	55			UK
1955	49°50'	2°18'	1164	35	44			UK
	32°37'	14°05'	1453	12	33			UK
1956	49°50'	2°18'	1038	44	33			UK
1957	49°00'	2°18'	1537	109	161			UK
	32°42'	19°30'	4404	955	88			UK
1958	32°42'	19°30'	2715	695	1085			UK
	49°50'	2°18'	1011	58	57			UK
1959	49°50'	2°18'	1197	4	74			UK
1960	49°50'	2°18'	2551	74	218			Belgium, UK
1961	49°50'	2°18'	1967	20	308			UK
	32°38'	20°05'	4361	563	1630			UK
1962	46°27'	6°10'	253	17	163			UK
	49°50'	2°18'	1444	5	76			Belgium, UK
1963	49°50'	2°18'	1543	3	44			UK
	45°27'	6°16'	5809	369	7131			Belgium, UK
1964	45°27'	6°36'	4392	444	15090			UK
1965	48°15'	13°15'	1760	114	13754			UK
1966	48°15'	13°15'	1044	78	2742			UK
1967	42°50'	14°30'	10895	253	7636		Yes	Belgium, France, Germany, Netherlands, UK
1968	48°20'	13°16'	3164	731	74837			UK
1969	49°05'	17°05'	9178	485	22066		Yes	Belgium, France, Italy, Netherlands, Sweden, Switzerland, UK
1970	48°20'	13°16'	1674	233	20224			UK
1971	46°15'	17°25'	3968	627	11148		Yes	Belgium, Netherlands, Switzerland, UK
1972	46°15'	17°25	4131	681	21626		Yes	Belgium, Netherlands, Switzerland, UK
1973	46°15'	17°25'	4350	740	12660		Yes	Belgium, Netherlands, UK
1974	46°15'	17°25'	2265	416	100356		Yes	Netherlands, Switzerland, UK
1975	46°15'	17°25'	4454	767	57374	29690	Yes	Belgium, Netherlands, Switzerland, UK
1976	46°15'	17°25'	6772	878	53518	20703	Yes	Belgium, Netherlands, Switzerland, UK
1977	46°00'	16°45'	5605	958	76451	31886	Yes	Netherlands, Switzerland, UK
1978	46°00'	16°45'	8046	1101	79628	36613	Yes	Belgium, Netherlands, Switzerland, UK
1979	46°00'	16°45'	5416	1414	83166	42240	Yes	Belgium, Netherlands, Switzerland, UK
1980	46°00'	16°45'	8391	1855	181227	98135	Yes	Belgium, Netherlands, Switzerland, UK
1981	46°00'	16°45'	9434	2177	153566	74372	Yes	Belgium, Netherlands
1982	46°00'	16°45'	11693	1428	126988	77449	Yes	Belgium, Netherlands, Switzerland, UK

Note: The number of digits quoted for mass and activity in Curies reflects accounting procedures and does not give an indication of the accuracy of the data. The α-activities include ^{226}Ra and the β/γ-activities include $^{3+}$H unless not specific $^{3+}$H value is quoted (years 1975 - 1982).

3.2.2. Description of Disposal Techniques

Low and intermediate level wastes were dumped in the NE Atlantic between 1949 and 1982. Figure 3.2.2 shows the location of all sites, with dates, to which should be added the USA site(s) mentioned previously. Guidelines for site selection are included in the LDC (see NEA 1985 for details). High level wastes have not been dumped in the sea.

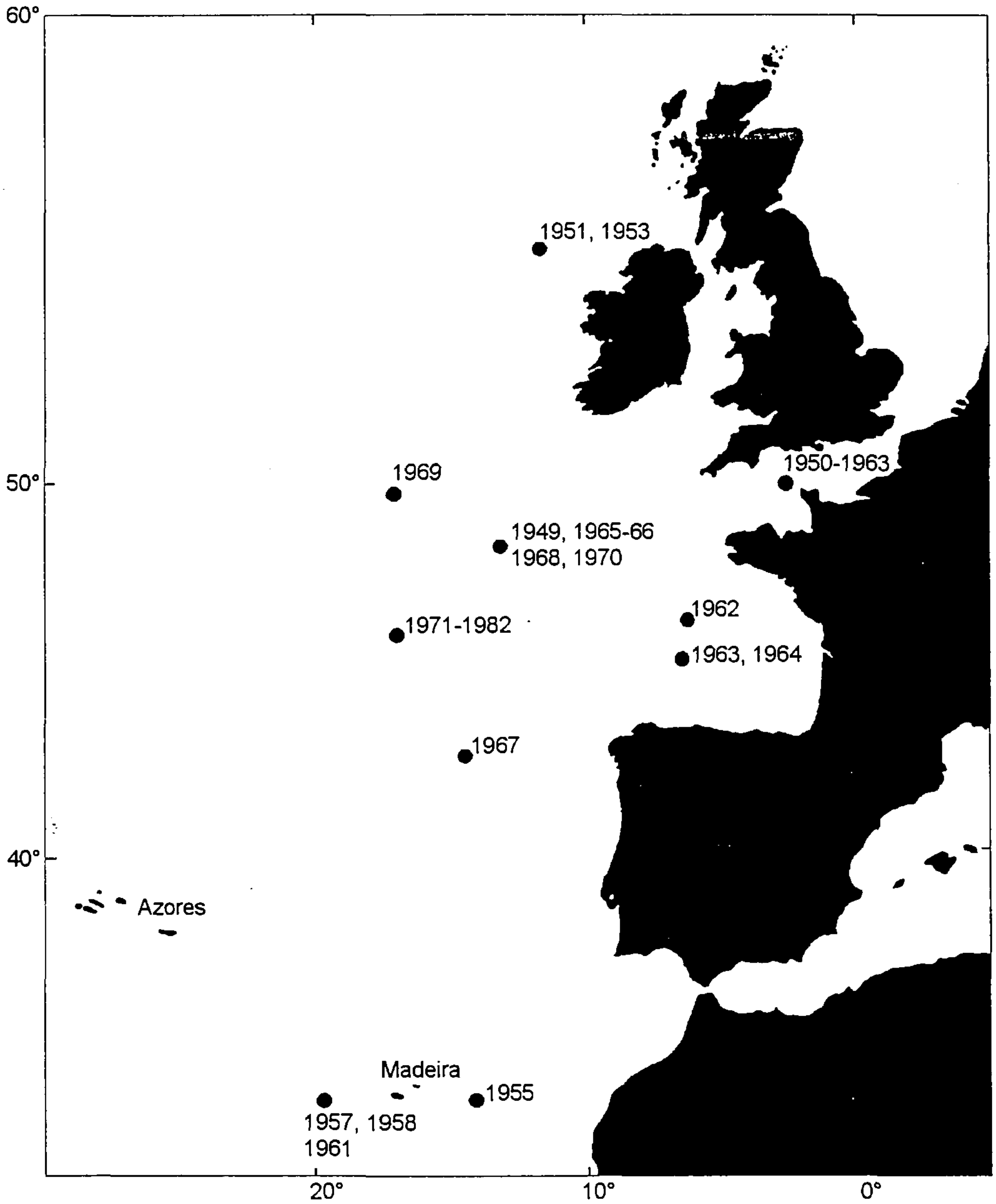

Figure 3.2.2 North-East Atlantic radioactive waste disposal sites (from NEA 1985).

3.2.2.1. Low/Intermediate Level Wastes

Considerable quantities of low level radioactive waste have been disposed of in the deep ocean. Between 1971 and 1982, about 123,000 drums and other containers full of low level nuclear waste were disposed of by several European nations, of which the UK was the largest contributor, in the outer part of the Bay of Biscay over an area of 5000 square kilometres near 46° N, 17° W. A survey of the site (Sibuet *et al.* 1985) discovered six containers, of which one appeared to be leaking.

Between 1946 and 1965, over 75,000 drums of US low-level radioactive waste were dumped at three sites in the Atlantic and Pacific. Atlantic dumping took place near 38° N, 71° W at depths between 2800 and 3800 metres. Surveys of two of the sites have revealed some leakage of radioactive caesium and plutonium into the environment (Dyer 1976), and further monitoring of the sites could provide valuable information on the effects of low-level radioactivity on the deep-sea environment and on the dispersion of contaminants.

Prior to 1966 the design of waste disposal packages for low and intermediate level wastes was the responsibility of individual countries (effectively only the UK disposed of radioactive wastes in the sea prior to this). Packaging specifications were laid down by the ENEA for the first international dumping operation in 1967 - described in detail in ENEA (1968), and various revisions were issued subsequently. The LDC specifies that waste should be in the form of solid packages, no significant quantities of liquids or gases are permitted and sludge must be solidified prior to packing. Prior to the LDC, the UK disposed of low level sludge in the Hurd Deep in the English Channel between 1950-1963. Disposal was in containers which were designed to implode at depth resulting in direct release and dispersion in the sea (Holliday 1984). Sludge was also disposed of during the first international operation (ENEA 1968).

Waste packages were designed to provide shielding and containment of waste during handling and transport, and as a means of delivering the waste to the seabed without loss on the way down. Various experiments using pressure chambers were carried out (Pearce and Vincent 1963, Seki *et al.* 1980). Containers were also photographed *in situ* (Sibuet and Coic 1989) or recovered (Colombo *et al.* 1983, Feldt *et al.* 1985).

Waste container and waste were integrated together in two basic designs, monolithic or with voids (air spaces). Figures 3.2.3a and 3.2.3b show examples of both types; all containers had a specific gravity of >1.2 to ensure that they sank. In monolithic packages waste was incorporated in a matrix of cement, bitumen or a polymer. The matrix was usually enclosed in a sealed container, typically a mild steel drum, and sometimes surrounded by a concrete lining for further shielding. Most waste disposed of in the NE Atlantic was in void containing packages which consisted of a concrete lining around the inner waste container(s) with a device to ensure pressure equalisation as the drum descends. The whole was contained within a steel drum. Typical steel drum thickness was 3mm; concrete caps were 80mm thick with reinforcing bars; outer concrete containers were ca 50mm thick. Drums used by the UK ranged in size from 220 - 690 litres (NEA 1985).

Details of dumping operations and the monitoring of this as regards crew safety are contained in ENEA (1968). Essentially the drums were dropped overboard from the ship and sank to the bottom. Impact velocity on the seabed was equivalent to a drop of <4m, and so it was considered to be very unlikely that the drums were damaged by the impact on the sea floor. Drum integrity, therefore, depends on corrosion, or failure of the seal on lidded drums. Estimated lifetimes of the steel drums vary from 15 - 150 years; for the concrete caps perhaps as little as 3 years. Concrete and bitumen waste forms are assumed to remain intact for 10^3 years before breaking up, with complete disintegration in 10^5 years. These rates will be higher for wastes containing lumps of corrodible material. Recoveries of intact drums (see earlier references) do nothing to negate these estimates. Models of release rates of radionuclides with different containers and different corrosion scenarios are discussed in Holliday (1984) and NEA (1985) (Fig. 3.2.4).

3.2.2.2. High Level Wastes

The dumping of high level radioactive wastes in the oceans is prohibited under the 1972 LDC. However feasibility studies and associated scientific research are allowed since these do not involve the actual disposal of any nuclear waste (NEA 1984). In 1975 the RWMC of the NEA decided to organise a workshop to determine the interest, nature and scope of possible international co-operation in the field of waste disposal option investigations. The first workshop was held in 1976 to consider the disposal of reprocessed high-level radioactive wastes or spent fuel in geological formations below the ocean floor. This workshop led to the formation of the Seabed Working Group (SWG) of the NEA in 1977, which subsequently met every year until 1987. Membership grew from 4 countries in 1977 to 10 plus the Commission of the EC in 1987. The final reports of the SWG were published in 1988 (NEA 1988a-h), an interim report was published in 1984 (NEA 1984) and an overall summary by Murray *et al.* (1991). The following account draws heavily on these earlier reports.

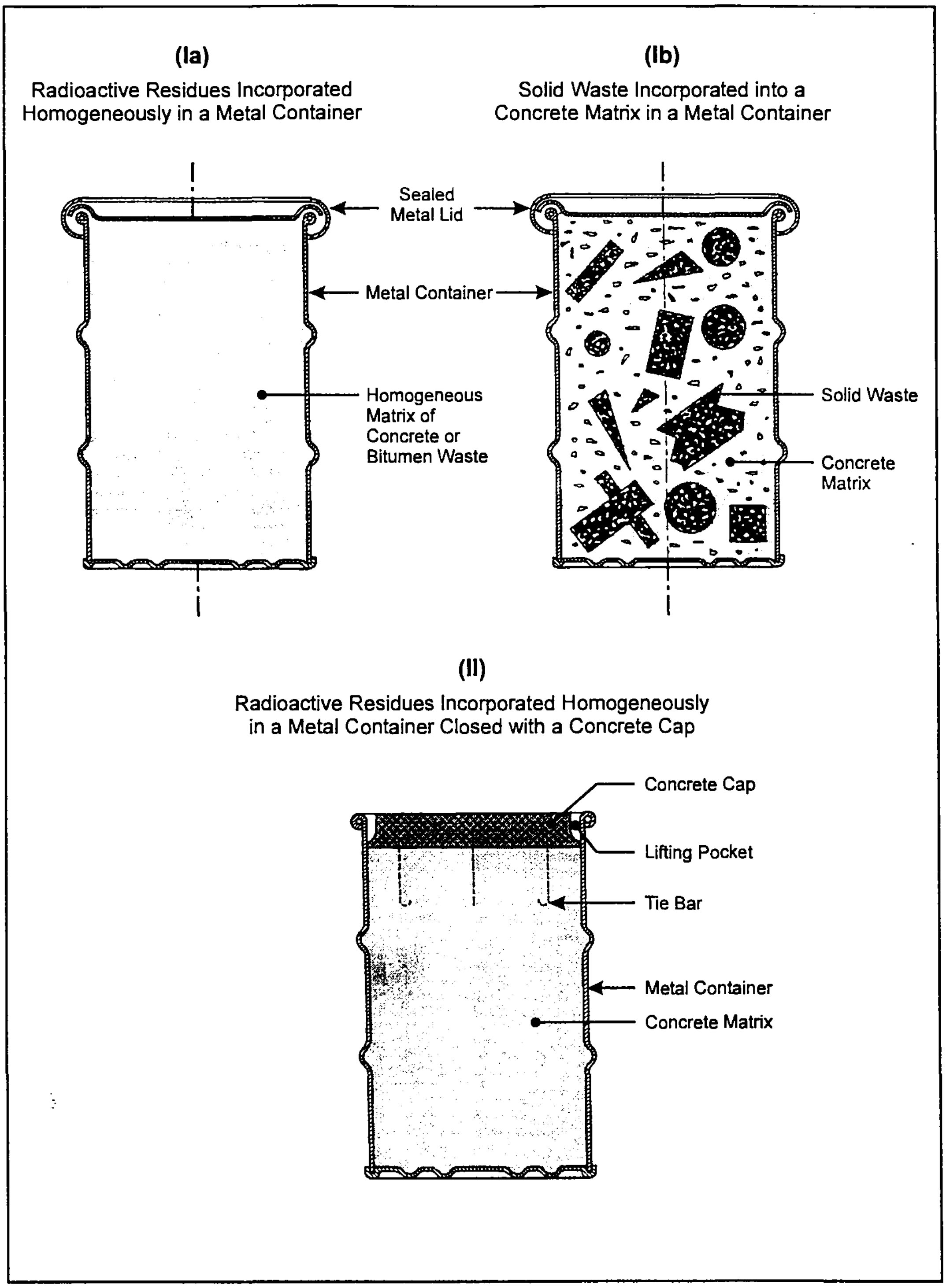

Figure 3.2.3a Types of monolithic packages (acceptable under NEA guidelines for sea dumping packages) (from Holiday *et al.* 1984).

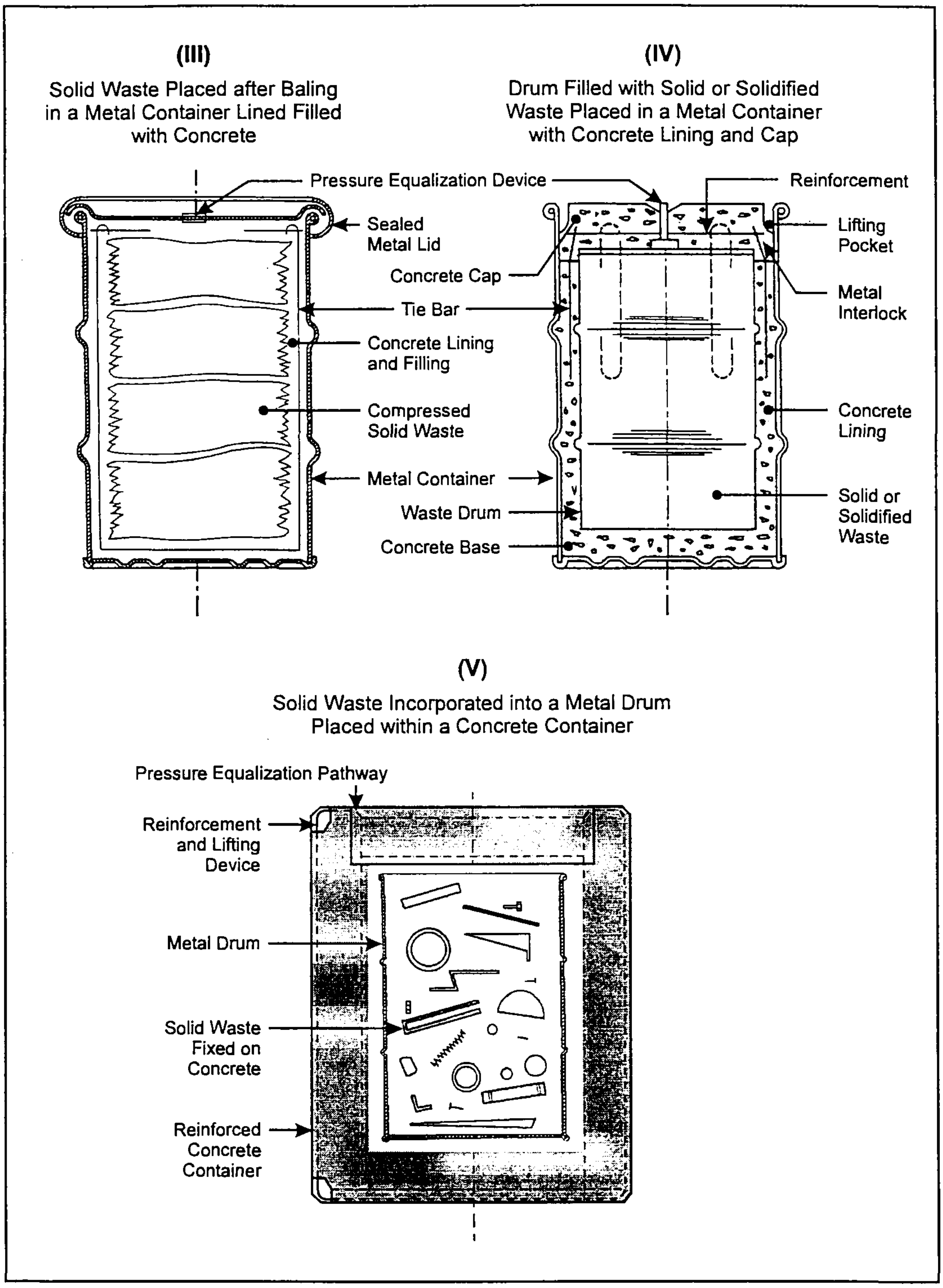

Figure 3.2.3.b Types of void-containing packages (acceptable under NEA guidelines for sea dumping packages) (from Holiday et al 1984).

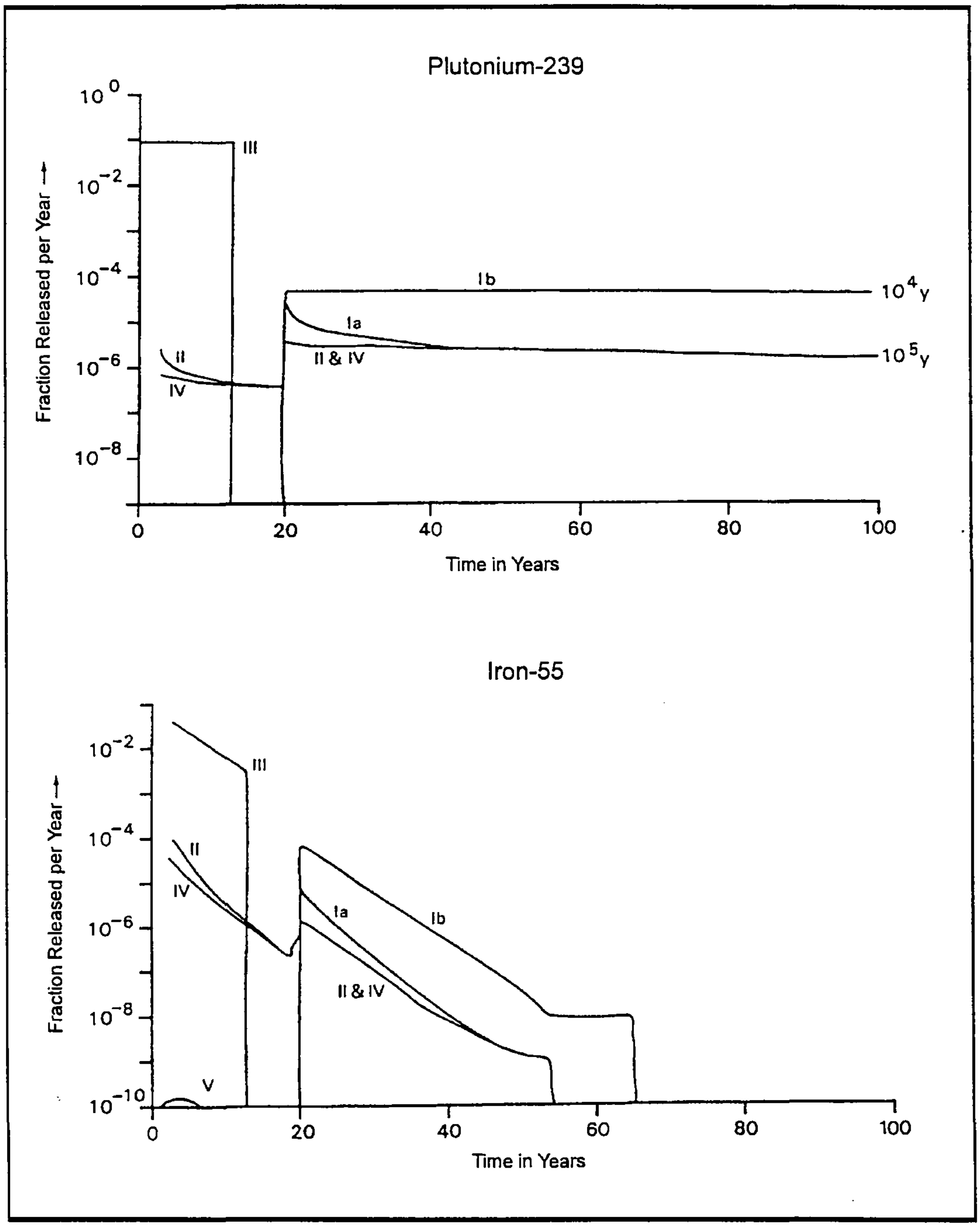

Figure 3.2.4 Fractional release rates from the package types as calculated by the model (from Holliday et al. 1984).

Ia	Monolithic, in concrete steel lid (see Fig. 3.2.3a)
Ib	Monolithic, in bitumen, steel lid (see Fig. 3.2.3a)
II	Monolithic or multistage, in concrete, concrete cap (see Fig. 3.2.3a)
III	Vented, loose packed (see Fig. 3.2.3b)
IV	Vented, encapsulated in concrete (see Fig. 3.2.3b)
V	Solidified in concrete, concrete container (see Fig. 3.2.3b)

The disposal of high-level radioactive waste beneath the seabed has as its prime objective the isolation of such waste "from the biosphere in suitable geological strata for a period of time and in conditions such that any possible subsequent release of the radionuclides in the environment will not result in unacceptable radiological risks, even in the long term" (NEA 1988a). The concept differs from sea dumping of low level wastes which does not involve isolation of waste within geological strata. A multibarrier concept was envisaged in which suitably processed and packaged waste was implanted into suitable sediments. The task of addressing the packaging and disposal techniques fell on the Engineering Studies Task Group (ESTG, NEA 1988d).

Figure 3.2.5 illustrates some of the >20 concepts for emplacing high level waste which have been considered. Other ideas, e.g. a seabed repository made of concrete and weighing ca. 2000 tonnes, were also investigated (Hemming and Freeman 1988). The ESTG chose 2 methods for detailed study, drilled emplacement of assemblages of canisters and emplacement of single or multiple canisters by free falling penetrators. Waste within the canisters is either reprocessed fuel waste dispersed within a matrix of borosilicate glass, or spent fuel rods. The waste is encased within canisters made of stainless steel (European Reference Waste) or titanium (European Spent Fuel and USA waste) each designed to ensure containment for at least 500 years. Drilled emplacement relies on the ocean drilling technology developed in the International Deep Sea and Ocean Drilling Projects. Essentially strings of waste packages would be lowered into predrilled holes, the holes backfilled and subsequently sealed. The Penetrator option relies on free falling penetrators, filled with multiple (European) or single (USA) waste canisters, embedding themselves up to 70m deep in the sediments. Both techniques require custom built transport ships, and, for emplacement, a drilling platform. Laboratory and field experiments were carried out to verify the concepts. Cost and risk analyses were also made. Further details of the proposed techniques, waste canisters and the various modelling and verification studies carried out in the programme are given in NEA (1988a, d).

3.2.3. Impacts of Disposal

Section 3.2.2. makes clear that dumping of low level radioactive wastes occurred in the NE Atlantic from 1949 to 1982, and that high-level waste has not been dumped or disposed of in the sea at all. It follows that "actual" effects of dumping high-level wastes are zero, but for completeness a short outline of the feasibility study carried out by the SWG of NEA is given below.

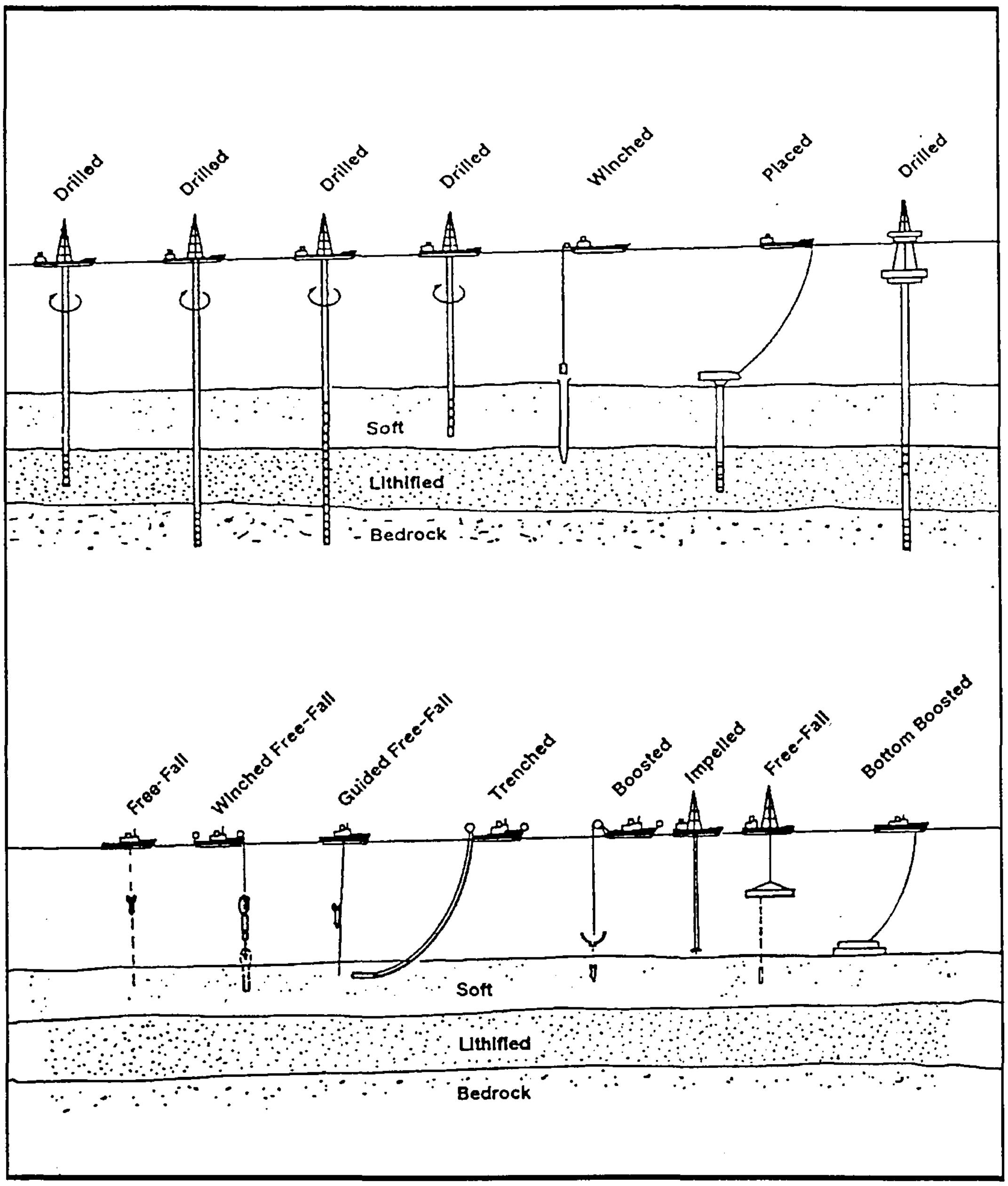

Figure 3.2.5 Representative emplacement methods (from NEA 1988).

3.2.3.1. Low Level Wastes

Since 1967, the disposal of low level radioactive wastes was controlled by the ENEA, later the NEA. To further the objectives of the LDC, in 1977 the Organisation for Economic Co-operation and Development (OECD) created a Multilateral Consultation and Surveillance Mechanism for Sea Dumping of Radioactive Waste to provide for international consultation and scrutiny. This Mechanism requires the NEA to assess the suitability of dumping sites at a minimum of 5-yearly intervals, and this assessment includes hazard assessment and environmental evaluation (see NEA 1980 for further details).

Under the terms of the OECD decision, a review of the suitability of the Atlantic dump site was carried out (NEA 1980) in which it was confirmed that the site would be suitable for dumping waste during the next five years at annual rates comparable to those reached in the past. It was also recommended that future assessments should be based on an increased scientific data base and that a site specific model of the transfer of radionuclides to the marine environment should be developed.

To achieve these objectives a group of NEA experts established a 4 year Co-ordinated Research and Environmental Surveillance Programme (CRESP) relevant to the Atlantic dump site (NEA 1981). Results from the initial CRESP were reported as interim descriptions of the NE Atlantic site (NEA 1983a, 1986a), as a major review (NEA 1985), and as progress and activity reports (NEA 1983b, 1986b). The 1985 review concluded that from a radiological protection viewpoint the NE Atlantic site would be suitable for continued dumping at existing rates since this would not give significant doses to man or marine organisms.

In parallel with the CRESP activities, the IAEA asked the joint Group of Experts on the Scientific Aspects of Marine Pollution (GESAMP) to provide advice on the most suitable oceanographic modelling technique to be applied to the deep-sea dumping of both radioactive and non-radioactive substances. The resulting report (GESAMP 1983) summarises relevant physical, geochemical, and biological processes, describes existing and potential models, and outlines future research needs in both processes and models. CRESP was extended in 1986 for 5 years with the additional remit to consider land based sources of radioactivity. The results of the second phase of CRESP were summarised in NEA (1990), with a further interim description of the site in NEA (1989). NEA (1990) contains further tables, summarising progress in the four areas of research into geochemistry and physical oceanography, biology, model development and radiological assessments. The 1990 report concluded that there was no scientific evidence to suggest that the NE Atlantic dump site could not be used within the next five years for dumping at rates similar to those previously adopted. Within the body of the report the different task groups confirm that the improved data collected since 1985 did not invalidate the earlier hypothesis or estimates. The biological task group found that earlier estimates of radiation exposure were too high and concluded that "all available evidence indicates that incremental radiation exposures of this magnitude (from dumping) could not result in any observable effects in the populations of deep-sea organisms at the dump site". The group also found no evidence to support an artificial reef effect of faunal concentration around and on the drums of waste.

It follows from the preceding account that few, if any, actual effects of dumping low level wastes have been found. The various summary reports quoted earlier are all supported by detailed scientific reports on particular aspects, many of which have been published in the open literature. For example, Feldt *et al.* (1985) described radioecological results from cruises of the *Walther Herwig* between 1980-1984. They found that the levels of radioactive contamination in the sampled fauna could be attributed to fall out from nuclear weapons testing - except in seven cases where the trawl had passed close to an old dump site, suggesting that any contamination will probably be restricted to the environment very close to the drums. However, in 1989, Feldt *et al.* concluded that the elevated levels of Cs^{137} found in Actiniaria resulted from a number of biogeochemical processes and was not the result of contamination from waste drums. They concluded that contamination from waste drums dumped in 1967 could not be established.

In 1990, CRESP activities were extended for a further five years until 1995. The report on these activities is not yet available.

All these reports are based on sound scientific knowledge and many data collected in monitoring programmes. However, the drums were never closely inspected except for a photographic system gliding a few meters above the seafloor (Sibuet and Coie 1989). We regard this to be insufficient for an environmental impact assessment or a monitoring programme.

3.2.3.2. High Level Wastes

The feasibility studies into the disposal of high level wastes into the seabed have been referred to earlier. The SWG was asked to address the following questions:

- Are there locations in the oceans which have the geological stability and barrier properties suitable for disposal?
- Is it possible to implant waste containers in the seabed sediments and what effect does this have on the barrier properties of the containment system?
- What are the radiological consequences of seabed burial?

To address these questions the SWG set up 7 task groups - Radiological Assessment, Site Assessment, Sediment Barrier, Engineering Studies, Near Field, Physical Oceanography and Biology, each of which produced a final report (NEA 1988a-h) and each of which contains recommendations for further work. In general the reports established the feasibility of seabed disposal.

As with the low level waste reports the SWG reports were underpinned by many years of research into various specific aspects. The results of these studies are summarised in the NEA final reports, and have been summarised overall by Murray *et al.*, (1991). Details are to be found in national reports on specific activities (Roe *et al.* 1987), characterising the biology of the Great Meteor East Site, and these results in turn have been given in the open literature, e.g. Roe (1988), Roe *et al.* (1990). Subsequent to these reports there are national listings of research priorities for future work. For example, the UK's Natural Environment Research Council (NERC) lists dispersion processes, dispersion models, monitoring of existing dumpsites, impact assessments, and studies on the immobility of naturally occurring radionuclides as subjects for further research (NERC 1995).

3.2.4. Research Required to Evaluate Impacts

There are enough existing "experiments" for reasonable validation of deep ocean disposal of radioactive wastes. Either the impact of old dumpsites or of sunken nuclear-powered vessels can be monitored. The earlier monitoring concentrated on radiological impacts and on the spread of radioactive contamination around the disposal sites. The ultimate fate of anthropogenic isotopes was investigated with modelling, but mainly to investigate the worst case scenario for radiological doses to Man. The results of these were generally optimistic about the probability of significant mass or critical-pathway doses reaching the human population (GESAMP 1983). However, little has been undertaken as to the impact on living communities either on the seabed or in the water column, and any effects induced changes in the biological communities might have on chemical exchanges, particularly across the sediment/water interface. It can be predicted that the impacts will generate concentric rings of decreasing effects, or perhaps ellipsoids with their long axes oriented along the predominant ambient current direction, centred around each disposal site analogous to the effects seen around hydrocarbon installations in shallow-water (see Olsgard and Gray 1995).

The advantage of monitoring a sunken nuclear-powered vessel is that the exact date of the accident and a reasonably precise inventory of the radioactivity on the seabed would be available. However, monitoring one of the old dumpsites would be more directly analogous with any future disposal that might take place and the isotopes involved would be less of a threat to the researchers involved if significant leakage has taken place. Monitoring these sites would not only provide information about the environmental safety of using the deep ocean for disposal of radioactive wastes, but would also provide the general public with information about the risks to deep ocean communities and ecological processes, to living resources, and to the human population, associated with these existing inputs. The monitoring would also provide useful scientific information about the sensitivity of deep-living communities to anthropogenic disturbances. Improvements in sampling techniques, the development of more reliable benthic landers (Tengberg *et al.* 1995) and the greater availability of submersibles and remotely controlled vehicles (ROVs) will enable researchers to achieve the fine-scale and close proximity of sampling that may be required. These techniques will also allow collection of more reliable data series. However, since the level of radioactivity and leakage of the barrels is assumed to be slow, and since damage to the barrels at the time of impact on the seafloor was presumably rare and corrosion in the deep sea is a slow process, studying the general water body at the dump site or the wider surroundings of the seabed may not reveal any measurable effects which could easily remain hidden within background values. Close-up inspection and targeted sampling are the key methods to be employed. Both these techniques are available today. It is essential to utilise ROVs and/or submersibles for close visual inspection, for probing corroded material of the barrels, for sensor measurements directly at possible leakage points, and for sediment/animal sampling directly adjacent to the barrels and at selected distances, following a potential downstream gradient. Only such near-field studies will deliver unequivocal results when a number of barrels and dump sites are investigated. Between 1967 and 1982, the NE Atlantic dump site was shifted from the northern Iberian deep-sea plain further to the north-north-west (Fig. 3.2.2). Thus older barrels, i.e., earlier dumps, are well separated from more recently dumped, younger barrels, and their respective levels of degradation may well be compared. This year is the 30th anniversary of the first common European dumping, and 15 years ago dumping was terminated. Such large-scale dumping over time and space should be evaluated as if it represented a large-scale "experiment". The experiment has already been initiated and an experimental basis has been laid, but adequate investigations have never been conducted. Such close-up research is essential for evaluation of the experiment and for risk assessment. Future generations of oceanographers and other members of the human community may criticise the scientists and politicians of today for failing to utilise properly the "experiment" and therefore not obtaining optimum information from these early dumping events. Detailed and close-up investigation of the barrels are demanded in the interest of both science and the larger community.

The radiological models, now a decade old, need to be updated, both to include the improvements in modelling programmes and the increases in our understanding of physical dispersion in the deep ocean which have occurred during the last ten years.

Should the outcomes of the monitoring and modelling continue to indicate that the risks to human populations and to deep-ocean communities and processes are possibly smaller than those associated with present methods of managing radioactive wastes, the present moratorium on their disposal in or under the ocean may need to be rewiewed. Then, and only then, might an experimental disposal programme be initiated to investigate finally the safety of deep ocean disposal.

3.3. Offshore Installations

3.3.1. Description of Waste Materials

Deep Sea disposal of large offshore structures or similar facilities has been considered by the oil and gas industry which will need to decommission an increasing number of offshore installations during the coming years. There are obvious economic advantages in disposing of large offshore installations in the deep ocean, particularly those whose designs pose particular difficulties for the alternative of bringing them inshore for dismantling. It is difficult to determine the total numbers of large structures for which deep-sea disposal may prove to be the best option. Considering only hydrocarbon installations, according to the oil and gas industries (International Offshore Oil and Natural Gas Exploration and Production Industry 1996), there are currently world-wide more than 6500 off-shore installations, located on the continental shelves of 53 countries. The vast majority (ca. 4000) occur in the Gulf of Mexico; 950 are in Asian waters, 700 in the Middle East, and in excess of 400 in European waters, most of them in the North Sea. These installations range in size and construction from small steel structures, weighing a few hundreds of tonnes being operated in shallow coastal areas, to massive steel and concrete structures weighing more than one million tonnes deployed in deep water. As exploration and extraction by the industries progress into deeper and deeper waters and into stormier regions, the number of these massive structures is likely to increase.

According to guidelines issued by the International Maritime Organization (IMO), installations deployed in less than 75m of water with superstructures weighing less than 4000 tonnes when decommissioned, must be removed from the sea in their entirety. Those emplaced in deeper water must be removed to depths of at least 55m. Nevertheless, a large number have already been toppled and/or left as artificial reefs, including about 90 platforms in the Gulf of Mexico; (representing about 10% of the platforms decommissioned so far).

More than half of the European installations are relatively small steel structures which are operating in water depths of less than 75m and will, on decommissioning, be completely disposed of on land. The majority of the remainder will either be removed entirely or, after the removal of their upper parts, will be toppled *in situ*. The deep-sea option, even if it becomes permissible, would be considered for only a few of the larger installations. Since all new oil and gas installations emplaced after 1 January 1998 will have to have their complete removal as an integral part of their design, the need for deep-sea disposal should eventually dwindle to zero.

The vast majority of the off-shore installations, whether operating in the North Sea or in other marine environments, are drilling platforms used to locate and extract the resources. While they are operational, these platforms cause fairly extensive damage to local environments as a result of their physical and chemical presence and also their operational discharges of drilling muds. However, these types of impact are not addressed in this report. Since these installations do not during working their life time accumulate significant quantities of contamination such as residues of hydrocarbons and radioactive scale, they should be relatively easy to clean up leaving comparatively simple inventories, albeit of large quantities of mainly structural metals and other materials.

In contrast, oil storage installations such as the *Brent Spar* in the North Sea are relatively rare, with only about four occuring in European waters. During their working lives many hundreds of thousands of tonnes of oil pass through the tanks of these storage platforms, so that the tank and pipe linings become progressively more contaminated with oily residues and low specific activity (LSA) radioactive scale, neither of which are easy to clean off. Consequently, storage systems are not particularly good analogues for the vast majority of oil and gas platforms. In view of the controversy surrounding the *Brent Spar*, the Natural Environment Research Council (NERC) set up an independent international group of scientists and engineers to consider and report upon the scientific evidence in relation to the environmental impact of various disposal options for offshore oil and gas structures (NERC 1996). Much of what follows is based on this report.

3.3.2. Description of Disposal Techniques

The deep-sea disposal of large structures would consist of three relatively simple operations: preparation of the structure for towing, towing to the disposal site and sinking. Although there may be considerable engineering problems in each of these operations, they are all achievable with available techniques.

Preparation for towing would involve two distinct phases. First, the structure would be stripped of all particularly noxious or valuable materials and equipment to leave a relatively 'clean' structure. Second, the structure would have to be made sufficiently buoyant, stable and mechanically sound for towing. The relative importance and difficulty of these phases would vary with the nature of the structure. For example, in the case of the *Brent Spar* the cleaning operation was extensive because of the amount of contamination accumulated during the *Spar's* working life. But it was already buoyant and stable so that only the anchoring system needed to be disconnected. In contrast, drilling platforms for deep-sea disposal would require relatively little cleaning, but would have to be disconnected from the seafloor, made buoyant and possibly toppled. The towing operation would be relatively straightforward, though clearly sensitive to adverse weather.

Once at the selected disposal site, the buoyancy package would be disrupted to allow the structure to sink. Again, the degree of difficulty in this procedure would depend upon the specific structure. Sinking of a drilling rig would be relatively simple, but the disposal of the *Brent Spar* or a similar structure would be a delicate operation in order to maintain equal pressure within and outside the storage tanks to avoid catastrophic toppling and break-up during the process.

3.3.3. Impacts of Disposal

The environmental impacts in the deep sea of large structures would be physical and chemical. Whether or not the structure breaks up during its descent, the physical effects would be predominantly on or close to the bottom; any physical effects in mid-water would be small, local and short-lived. However, if the structure contains significant contaminants, break-up during descent could have a considerable impact in the water-column as a result, for example, of the release of buoyant hydrocarbons.

Once at the seafloor, the impact would initially be largely physical, the severe disruption of the area impacted directly and the disturbance of large quantities of sediment that would be carried down-current and redeposited on areas more-or-less remote from the structure. Over the longer term, the physical impact would be largely the effect on the near-bottom currents of a large structure protruding from an otherwise generally flat seabed and the addition of hard substrate in an environment otherwise typified mainly by soft sediments.

The chemical impact would depend upon the nature of the structure. For those not containing significant contaminants, such as drilling rigs, the impact would be rather slow, as a result of the corrosion or disintegration of the structural components. Where contaminants are present, at least some of these are likely to be set free rapidly as a result of structural disintegration on impact with the seabed, while others would be released more slowly. The extent of any crucial or chronic effects of the contaminants would depend upon the amounts present, their toxicity, their rate of dissolution and the ambient dispersive regime.

3.3.4. Research Required to Evaluate Impacts

No significant direct research on the environmental impact of large structures such as oil rigs and storage platforms in the deep-sea has ever been conducted. NERC (1996) concluded, on theoretical grounds, that the impact of the deep-sea disposal of the *Brent Spar* would have been very small in the vast areas of the deep sea. However, they pointed out that the impact of such large structures could be analysed as the impact of different stages in the disposal operation and of different components of its structure, and that to each of these stages and components analogues in the oceans already exist such as the disposal of radioactive wastes (Subchapter 3.2), sewage sludge (Subchapter 3.4), and military wastes (Subchapter 3.1). Studies of these analogues were recommended. Ship wrecks and natural large-scale disturbances should allow further deductions to be made on environmental impacts which may originate from the disposal of large offshore installations.

Shipwrecks

Several aspects of the impact of large structures in the ocean are paralleled by shipwrecks, many of which are well documented in terms of their nature, timing and position. Between 1973 and 1995, 29.3 million tonnes of shipping, on average 1.3 million tonnes per year, were lost at sea (Institute of London Underwriters 1995), a significant fraction of it in the deep ocean. Losses during the two world wars were much greater and more concentrated in the deep oceans. In the four years of the First World War, over 7 million tonnes of British shipping were sunk, a large part of it in the deep North Atlantic (Hurd 1929). Similarly, during the Second World War more than 21 million tonnes of Allied merchant shipping were sunk in addition to substantial quantities of Axis shipping (Roskill 1961). Both wars also resulted in substantial losses of warships on both sides. Several tens of millions of tonnes of shipwrecks must therefore lie in the deep oceans, many of them originally including fairly noxious cargoes. Even closer analogues to large structures are provided by past intentional disposals of ships. A number of licences have been issued for the disposal of hulks used for target practice during military exercises in deep water to the west of the U.K.

With the exception of the celebrated cases of the *Titanic* (see Ballard 1987, Uchupi *et al.* 1988), the SS *Central America* (Herdendorf *et al.* 1995) the Russian nuclear submarine *Komsomolets* (Vinogradov *et al.* 1996) and data resulting from the salvage operations on the *Francois Vieljeux* (Dando *et al.* 1992), few studies have been made of shipwrecks in the deep sea. Even those studies which have been conducted did not consider the impact of the wrecks on the local environment. The Shepherd group (NERC 1996) therefore recommended that such studies should be conducted around wrecks of which the nature and "emplacement" date are known. These would employ conventional sampling and data gathering techniques at varying distances from the wrecks to examine primarily their physical impact but also, depending upon the original contents, possible chemical impacts.

Natural Phenomena

A large structure dumped onto an unstable sloping seabed might trigger a sediment slide. Similar natural disturbances of the deep ocean floor occur through submarine landslides, debris flows and turbidity currents, and, on a smaller scale, by benthic storms. The effects of these natural disturbances can be studied using conventional research techniques.

An estimated 250,000 tonnes of hydrocarbons are leaked into the ocean through natural seeps each year. Although no oil seeps have been described from the deep ocean, seeps of methane are common along many continental margins where this gas emerges from the seafloor after being produced from organic matter trapped in the sediments. Study of these seeps would provide useful information on the likely effects of hydrocarbon leakage from disposed structures. Similarly, many of the components of vent fields, such as the dissolved copper and zinc in the emergent fluids, or the presence of the resulting mass of chemically reducing material in the oxidising waters of the deep sea, are identical to, or closely similar to, the contaminants that would be present in oil field structures emplaced in the deep ocean.

The above mentioned naturally occuring disturbances and sources of pollution offer the advantage of a ready-made laboratory for studies of impacts to be expected from human activities. However, the environment and the communities occupying it are adapted to these natural conditions. Thus, considerable caution must be exercised when results are to be extrapolated to effects of man-made impacts.

3.4. Sewage Sludge

3.4.1. Description of Waste Materials

Sewage sludge is a product of the treatment of the domestic waste stream, and its characteristics are determined by the inputs and the extent of the treatments. The stream is initially the fluid wastes produced by the human population and so contains excreta and the "grey waters" containing detergents and other chemicals discharged domestically. However, the sewage stream is also often added to by storm waters, run-off from roads contaminated with hydrocarbons and, more seriously, industrial discharges in some of the older sewage systems. These more heavily contaminated sewage streams pose greater difficulties for processing and for the disposal of the resulting sludge because of low, but still environmentally challenging, concentrations of toxic substances including heavy metals and persistent organic materials. Table 3.4.1 lists data published for contaminated sewage sludge dumped off New York (Santoro and Fikslin 1987).

Table 3.4.1 Chemical composition in parts per million of sewage dumped off New York (Santoro and Fikslin 1987)

	Mean	minimum	maximum
Chemical Oxygen Demand	25570	635	126000
Total dissolved solids	31100	674	145000
Oil and grease	3090	11	25900
Petrol and hydrocarbons	1884	1	27000
mercury	0.23	0.002	4.1
cadmium	2.64	0.06	78.1
arsenic	0.59	0.009	11.4
lead	50.9	0.5	1009.2
copper	73.3	1.0	312.0
zinc	112.0	1.6	502.0
chromium	31.1	0.1	285.6
nickel	8.3	0.5	41.0
vanadium	2.8	0.2	80.0

The wide ranges between the maximum and minimum values illustrate the great variability in the composition of sewage sludges. The minimum values probably give an indication of the levels to be expected from uncontaminated sewage. The carbon content of the solid components of the sludge is about 5%. The average household in the UK produces 155 litres of watery wastes per person per day. Discharges and their contents are variable, reflecting the patterns of water usage by the local communities which are influenced by their affluence and the variability imposed by season and weather. For example, the use of washing machines and detergents is higher in affluent areas compared with deprived areas, and the flow will be influenced by weather patterns with high rainfall adding volume to the stream, whereas hot weather increases the quantities of water used for ablutions.

The Oslo Commission (1989) provided some figures for the amounts of sludge produced in those countries which are Contracting parties to the Convention (Table 3.4.2) and these give some feel for the size of the European problem. The differences in the amounts produced per capita also provide an approximate indication of the extent to which each country treats its sewage waste streams. So as more countries install secondary treatment, the average per capita annual production will rise to ca. 40kg of dry sludge, the quantity presently generated by Germany. Thus the 257 million people in the Convention community will be producing nearly 10 million tonnes of dry sludge per year, containing around 2 million tonnes of carbon (equivalent to 10^{-7} times the amount of carbon annually fixed by primary production in the ocean).

The objective of sewage treatment is to re-cycle the water in as chemically pure and environmentally safe a state as is practical and affordable, to prevent the spread of pathogens (including viruses) within the community and to minimise undesirable environmental impacts. The mode of treatment and level to which it is conducted is influenced by the standards set by central Governments (both national and European), and the regional setting which includes the state of the infrastructure of the processing plant. Domestic streams that are free from heavy industrial contamination result in sludges that are more readily disposed of through the production of soil conditioners. In some countries such as China untreated "night-soil" is added directly without treatment to agricultural land to maintain fertility, regardless of the public health hazards resulting from the ease with which pathogens and parasites get passed on in this way. Land-fill is also used for sludge, and others are incinerated leaving a residue of fly ash that is disposed of into land-fill. However, in some regions the discharge of partially or fully treated sewage into coastal waters still continues via long out-falls. Sewage sludge remains on the list of substances permitted to be disposed of into the ocean under the London Dumping Convention.

The sludges contain organic and inorganic particles including a very large component of silica. The organic material continues to be decomposed by microbial digestion under a high oxygen demand to produce methane and nitrogenous by-products. This organic portion also contains both bacterial and viral human pathogens and parasites. The major reductions in water-born diseases that occurred during the 19th century in Europe were directly attributable to the development of urban sewage systems. The fluids contain dissolved organic and inorganic substances rich in nitrogen and phosphorus which, if discharged without further treatment or adequate dilution, can lead to eutrophication in marine and freshwaters or unacceptable contamination of ground waters.

Table 3.4.2 Dry weight quantities of sewage sludge produced annually by the countries contracted to the Oslo Convention

	Year	Amount (tonnes)	Population (Millions)	amount per capita
Belgium	1984	29,000	9.9	0.003
Denmark	1984	150,000	5.1	0.029
Germany (West only)	1985	2,340,000	61.7	0.038
France	1984	840,000	53.6	0.016
Finland	1983	130,000	4.8	0.027
Iceland	n.d	n.d	n.d	n.d.
Ireland	1983	20,000	3.4	0.006
Netherlands	1983	310,000	13.9	0.022
Norway	1985	82,000	4.1	0.020
Portugal	n.d	n.d	n.d	n.d.
Spain	1983	45,000	37.0	0.001
Sweden	1984	190,000	8.3	0.023
U.K.	1983	1,500,000	55.9	0.027
Average				0.022

The main contents of a typical sludge consist of:

- Inert materials predominantly consisting of extremely fine particles of silica, most of which are <100μm in diameter; they tend to flocculate in sea-water.

- Reactive organic materials, which are predominantly cellulose fibres, higher fatty acids and oils. The organic nitrogen contents range from 6-10%. These create a chemical oxygen demand (COD) of 2000ml $O_2 \cdot g^{-1}$.

- Persistent organic materials, such as poly-chlorinated biphenyls (PCBs).

- Inorganic solutes, for example the liquid component contains about 1000ppm of ammonia.

- Bacteria dominated by methanogenic species, which continue to ferment the sludge, coliform bacteria and human pathogens.

- Industrial contaminants including heavy metals and hydrocarbons.

The physical characteristics of the sludge can be manipulated. Slurries containing <7% solids remain liquid with a specific gravity of 1.01 to 1.03 and can be pumped easily. If the solid content is increased to >7% the sludges become thixotropic and will form a gel if not agitated. The water content can be substantially reduced by centrifugation (e.g. prior to incineration). The method of disposal will determine the state in which the sludge is transported and discharged. The economics of transportation will demand that the sludges are de-watered to the maximum extent compatible with any further handling that is required.

3.4.2. Description of Disposal Techniques

Over the last two decades several European countries have disposed of sewage sludge in their coastal waters generally using hopper barges that empty their loads through doors in the bottom of the hull. The UK is the last European country to be involved in this practice which will be phased out in 1998. However, many countries still continue to discharge substantial quantities of treated, semi-treated and untreated sewage down outfalls into rivers, estuaries and inshore waters.

In 1982 the UK disposed of approximately $10 \cdot 10^6 m^3$ of sludge into its coastal waters, mostly at three sites: the Barrow Deep in the Thames Estuary, off Garroch Head in the Clyde and in Liverpool Bay. Quantitatively this volume is misleading since in order to keep the sludges liquid, they consisted of >93% water; the solid content amounted to $7 \cdot 10^5 m^3$.

Disposal options

Any waste disposal into the oceans will have detrimental effects, but if these impacts are small in extent, less persistent, and influence a smaller proportion of the global inventory of ecosystems, then they may be offset against impacts caused by alternative methods of treatment and/or disposal. The continuation of disposal of sewage sludge into continental shelf seas is not considered to be acceptable, particularly for those sludges which are heavily contaminated with industrial effluents. The majority of living resources exploited in the oceans are taken from shelf seas and over continental slopes. Although the growth rates of exploited stocks may be enhanced by the organic contents of the sludges, there is a risk of their becoming contaminated with pathogens and organic and inorganic contaminants from these sludges. In the North Sea, the incidence of fish diseases has been linked to discharges of contaminants including sludges (e.g. Möller 1992). However, if there is an association between the incidence of disease and sewage disposal, then the absence of diseases in fish stocks around the Garroch Head site needs to be explained (Pearson 1986). Nevertheless, other effects of the sludge disposal, such as reductions in water quality, restrictions on recreation, detrimental effects on shellfish farming and the effects of eutrophication degrading ecological and biological diversity and ecological processes, are sufficient argument for continuing efforts to reduce, and even, eliminate coastal and riverine discharges of sewage.

Similarly, the use of deep ocean techniques in the Mediterranean and Baltic Seas, where deep water circulation and exchanges are limited, is unlikely to be acceptable. However, the unusual conditions prevailing in the deep waters of the Black Sea might make deep disposal there environmentally acceptable. Thus, in a European context we are considering the possible use of the deep oceanic waters in the Northeast Atlantic. To limit possible detrimental effects water depths will need to be >4,000m and the morphology of the region's ocean basins (Fig. 3.4.1) would limit disposal to latitudes <51° N.

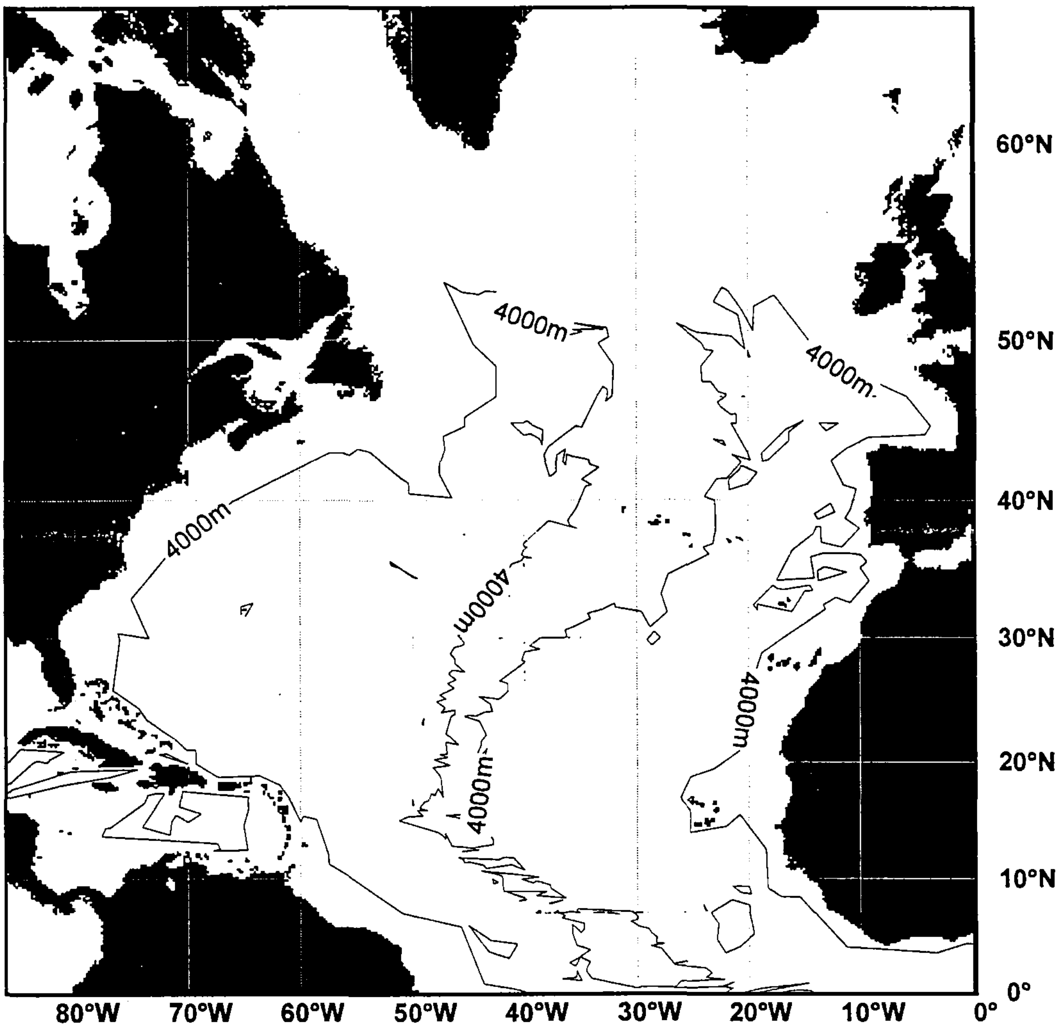

Figure 3.4.1 Map of the North Atlantic showing how waters deeper than 4000m (i.e. abyssal plains) are restricted to latitudes south of ca. 51° N.

There are three basic approaches to discharging sewage sludge and other wastes into the deep ocean:

- Discharge at the sea surface.
- Discharge at midwater depths of 1 - 2km, i.e. below the permanent thermocline.
- Discharge onto the sea-bed at depths >4,000m.

Discharge into near-surface waters or at subthermocline depths in midwater will result in greater dilution and dispersion. However, monitoring the dispersal of the contaminants will present serious technical problems as their concentrations become very low. Numerical modelling would need to be used to assess their fate, but uncertainties would persist as how best to monitor and analyse the impacts. There are likely to be impacts on the sun-lit upper layers which may contribute synergistically with other effects to change the basic biological processes of primary and the subsequent production levels. Hence, the experimental investigations discussed below are based on the assumption that disposal options most likely to be accepted will involve the isolation of the sludges from the water column either by discharging them via a pipeline directly onto the abyssal sea-bed (into an accumulative rather than a dispersive physical regime) or by dropping it in containerised packets such as „geotubes“ (De Bruin and Loos 1995).

Discharging sludge directly onto the sea-floor will minimise the impacts on the water column. Liquid slurry discharged within a few tens of metres of the abyssal sea-bed will form a coherent gravity flow until it reaches the sea-bed. Much of the material will rapidly sediment out close to the deposition point. The finer, lighter particles will be carried downstream in the benthic boundary layer with currents averaging a few $cm \cdot s^{-1}$ over abyssal plains. Impacts on the bottom communities will be maximal in the near-field region and decrease with distance away from the discharge point. The behaviour and spread of the discharge plumes will be analogous to the plumes of mining tailings discussed above.

Dropping synthetic "geotubes" containing large volumes of de-watered sludges can be expected to reduce the impacts substantially, so long as the tubes retain their integrity while being dropped, during descent and after hitting the bottom. If failures occur frequently, especially in midwater, the advantages gained in containing the sludges may be off-set. Technical trials would be needed before the mode of delivery can be selceted.

The use of the deep ocean for waste disposal will involve innovative techniques. Industry has already conducted limited development and testing of possible techniques for the mining of mangnese nodules (Chapter 2.1) and metalliferous muds (Chapter 2.3). No doubt, these techniques will undergo further development before mining on a commercial-scale starts, and final assessment of their impacts will have to await these developments. In contrast, apart from the basic technique of using hopper barges which empty their contents at the surface, proposed methods for deep ocean waste disposal are at an even more preliminary stage. Some of the problems encounteved will be analogous to those associated with the discharge of tailings after initial processing of the ores. So some of the experiments needed to examine aspects of the impact of mining activity will provide useful information for the assessment of any deep-ocean waste disposal options and *vice versa*. It is assumed herein that the future discharge of fine to coarse particulate material such as sewage sludges and dredge spoils (see Chapter 3.4) (even though not banned presently under the provisions of the London Dumping Convention) will only continue to be considered acceptable if additional steps are taken to isolate the materials from the water column.

Series of experiments, ranging from small to large-scale, will be needed using the disposal techniques deemed to be commercially feasible. It would be unrealistic to assume that a suitable technique might be evolved purely to conduct such experiments in the framework of an environmental impact assessment. Since the large-scale experiments can only be designed if they involve commercially viable methods, they will need to be undertaken in association with the development of industrial techniques.

Any consideration of the technical options will be constrained by the pragmatic needs to:

- use as far as possible existing methods for the loading, transport and disposal of the waste materials

- be capable of handling very large quantities of material at a scale of around a million metric tonnes per annum

- minimise risks to the human population

- minimise impact to coastal seas and the water column of the main ocean by keeping the waste isolated from the water until it is discharged at abyssal depths

- meet the criteria of acceptability suggested in Chapter 4.3.

Transportation would pose no novel problems. The shipping industry is well experienced in handling large bulk cargoes, but large shore facilities capable of supporting operations at an industrial scale will need to be developed near populations centres. If quantities of liquid slurries similar to those dumped until recently in UK coastal waters were disposed of into the deep ocean, it would require about 100 journeys by large bulk carriers, but if the sludge were to be de-watered for transportation the number of journeys would be dramatically reduced. Economic considerations will include:

- Very large Crude Carriers (VLCC) able to carry 250,000 to 300,000 tonnes may be adequate to serve most conurbations and can operate in and out of most large ports; Ultra Large Crude Carriers (ULCC) capable of carrying $5 \cdot 10^6$ tonnes could serve a few centres.

- Navigational conditions in local ports may constrain the vessel's size. However, off-shore loading techniques, well developed in the tanker and off-shore oil production industries, could be used to transfer wastes from small to large carriers for the transport to the disposal site.

- Voyage economies will be affected by the conurbations being served, the transit times between ports and disposal site.

- The discharge times will need to be minimised for economic and operational safety reasons. These may be shorter if wastes are containerised rather than discharged via a pipe-riser.

- Weather and sea conditions at the disposal site will at times limit operations and so there will be economic pressures to select sites where fewer days are likely to be lost, and to have alternative sites available if the weather and sea-states are constraining operations at the favoured site.

- The operations must minimise navigtional and safety hazards to other sea-users.

Several options for delivering particulate wastes to the deep ocean have been discussed in theory (e.g. Valent and Young 1995, Valent *et al.* 1997), but the further technical developments needed (including the design of specially built vessels and containment techniques) have been stalled by the swing of political and public opinion away from deep ocean options. Desk-top studies have addressed: Option 1. Shipborne pipe-risers; Option 2. Floating pipe-risers; Option 3. A tethered container; Option 4. Release of waste containerised in geotextile bags from a surface vessel; Option 5. Release of similarly containerised waste from a ROV glider; and Option 6. Release of similarly containerised waste from a free-fall discoid container.

Option 1: Shipborne Riser technique (Fig. 3.4.2)

The carrier vessel would deploy through a moonpool a vertically suspended discharge hose terminated with a clump weight to just above the sea-bed. Current shears within the water column will distort the pipe from the vertical, but theoretical analysis of the hose dynamics has demonstrated that by dynamic positioning, disposal operations could continue in gale force conditions (Beaufort 8), and if conditions deteriorated still further the hose could be retrieved safely.

The dump weight would keep the hose as straight as possible to carry both the exit nozzle and the instrument to monitor in real time the discharge and the nozzle's position. The optimum height of the exit nozzle will be a function of the relative densities of the sludge and the *in situ* sea water, the discharge rate, the buoyancy frequency and currents in the benthic boundary layer. Flume tests conducted in pressure tanks indicate that up to 1200 tonnes of slurry could be discharged per hour 50 - 150m above the sea-floor. Technical development of a hose system has been undertaken.

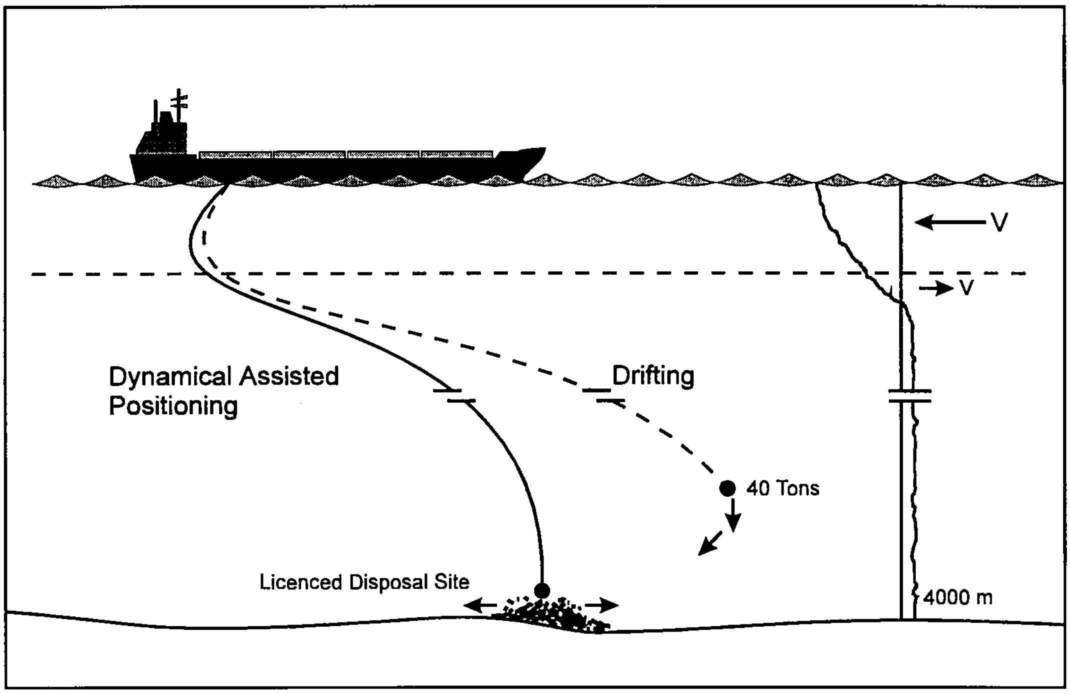

Figure 3.4.2 Sewage sludge and dredge spoil disposal.
Option 1: the shipborne riser concept (modified from B. Tallack, Northstar Marine and Environmental Consultancy).

Option 2: Floating Riser Concept (Fig. 3.4.3)

This method draws on techniques developed by the off-shore oil industry. A vertical unmanned spar buoy weighing approximately 10,000 tonnes and fitted with four polyethylene pipes discharging close to the sea-floor, would be semi-permanently moored over the disposal site. Shuttle tankers carrying the wastes would connect up to bulk transfer buoy and pump the wastes into the buoy. There the slurry would be diluted with surface sea-water in a mixing chamber to reduce its density before it is discharged, otherwise the velocity of the descending slurry could result in a catastrophic failure of the pipe (Valent and Young 1995).

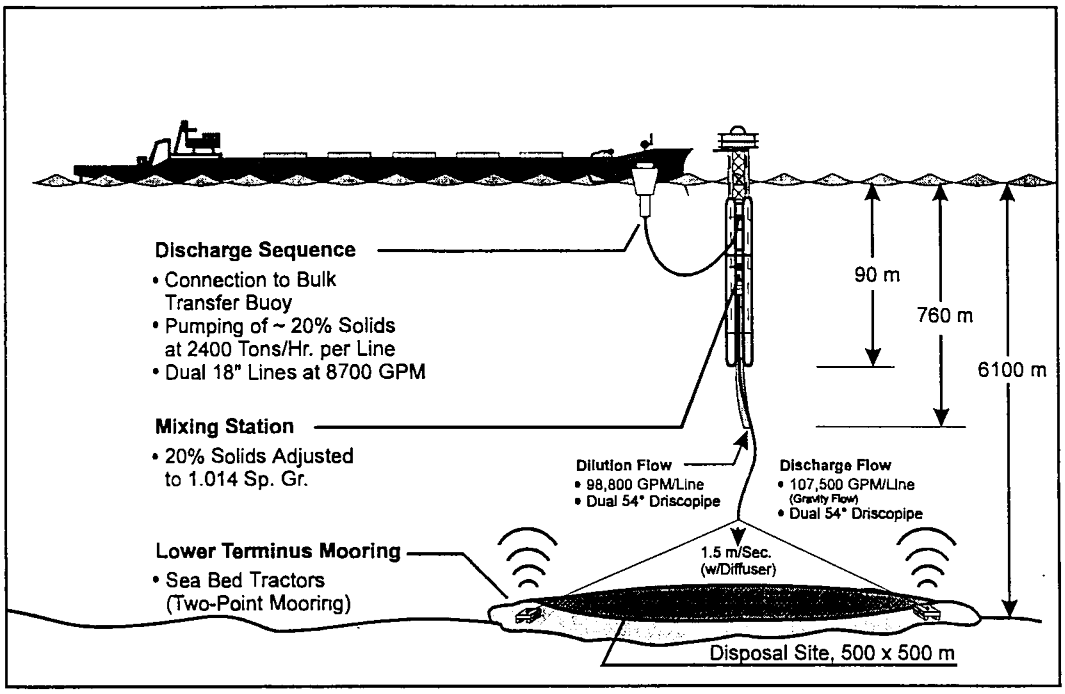

Figure 3.4.3 Sewage sludge and dredge spoil disposal.
Option 2: the floating riser concept (modified from Valent and Young 1995).

Option 3: Tethered container Concept

This method involves a free-falling tethered aluminium container loaded with $190m^3$ of waste from a vessel (Spencer 1991). At a preprogrammed depth doors at the top and bottom of the container snap open and the Venturi effect of the free-fall velocity vents its contents. The aramid-fibre cable halts the container's fall and is then used to haul it back to the vessel. The cycle time for descent and recovery is about one hour.

Option 4: Containerised from a surface vessel (Fig. 3.4.4)

In this method $350m^3$ waste is loaded into flexible geotextile bags (geotubes) lining individual cargo bays of a specially designed vessel. After filling, the bags are sealed ready for disposal. At the bottom of each bay is an individual exit hatch through the hull. Once at the disposal site, the bays are flooded before the hatches are opened in sequence, so that the bags are dropped without destabilising the vessel. The bags free-fall to the bottom at an estimated speed of about $5m \cdot s^{-1}$, ideally retaining their integrity down through the water and after hitting the bottom. If successful, this method would ensure that the wastes remain contained both during descent through the water column and on the sea-bed. A concept design described by Valent and

Young (1995) shows a vessel with 51 bays having a 25,000 tonne capacity. Modelling of the free-fall of the bags suggests that they would fall within a suitably sized dropping zone. One considerable advantage of this method is the very short time involved over the disposal site, saving time and reducing the risks of foul weather disrupting operations.

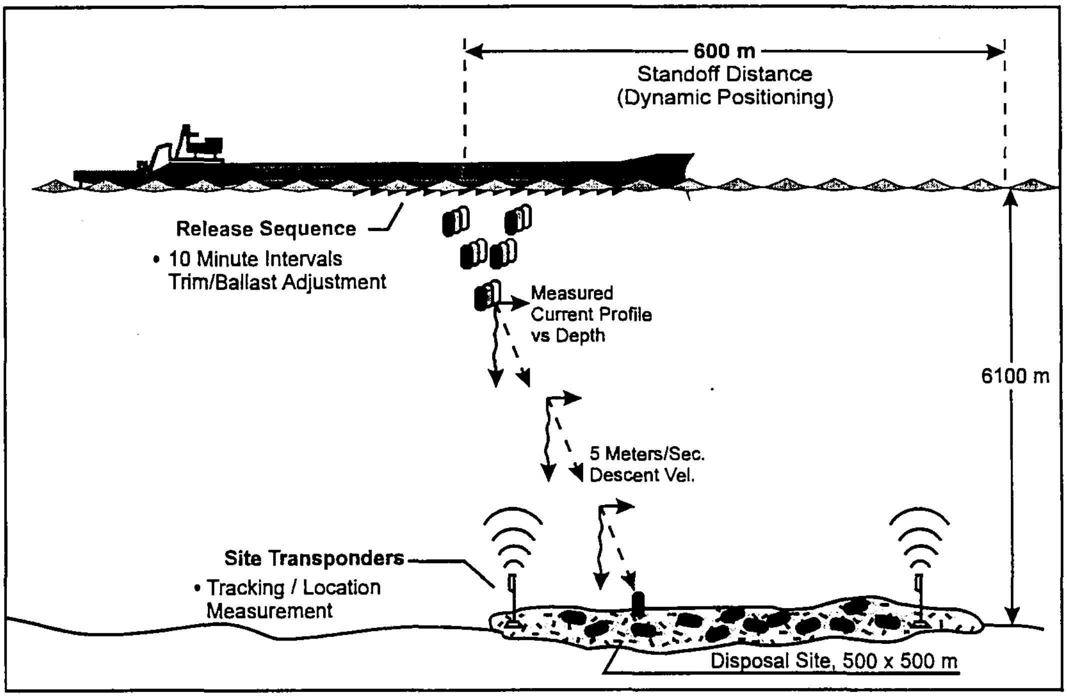

Figure 3.4.4 Sewage sludge and dredge spoil disposal.
Option 4: the surface emplacement concept (modified from Valent and Young 1995).

Option 5: Remote controlled glider (Fig. 3.4.5)

This concept is an elaboration of option 4 in which the carrier is replaced by a ROV glider, similarly fitted with trapdoored cargo bays in which geotubes with waste are loaded. The glider would be transported to the site in a mothership where it would be released. Being heavier than water, it would go into a dive at an angle of about 40° reaching a terminal speed of nearly 11m $\cdot$ s^{-1}. Close to the bottom it automatically opens all the traps and drops the geotubes. This makes the glider positively buoyant, so that it pulls out of its dive and starts its ascent. On reaching the surface it is recovered by the mothership and returned to port for reloading. The design concept sets the glider's capacity at just under 4,000 tonnes and a turn round time of about 15 minutes (Valent and Young 1995).

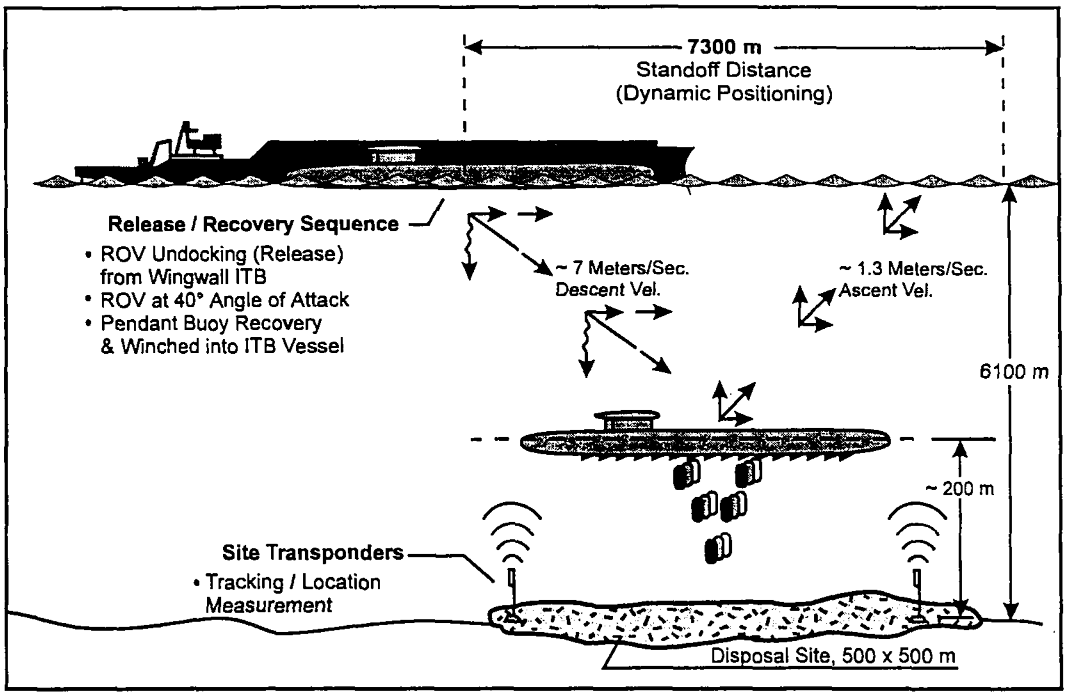

Figure 3.4.5 Sewage sludge and dredge spoil disposal.
Option 5: the ROV glider concept (modified from Valent and Young 1995).

Option 6: Direct Disk Concept (Fig. 3.4.6)

This is an alternative elaboration of Option 4, in which a series of free-fall discoid containers are dropped from a mothership over the disposal site. Each disk, loaded with geotubes in 169 bays, freefalls to within 100m of the sea-bed where the cell doors swing open and release the bags. Once they have been dropped, the disks become positive buoyant and rise back to the surface where they are recovered (Valent and Young 1995).

Option 1 and 2 are mostly based on existing techniques and involve the release of uncontained wastes directly on to the sea-bed. Option 3 will similarly release the waste uncontained and uses untried technology. The feasibility of the tether to decelerate and retrieve the container is the major unproven aspect of this concept. Options 4 - 6 have the possible advantage of containing the waste even after it has been released. Option 4 is probably the most feasible with existing techniques and while running the least risk of serious loss of hardware, involves the greatest risk of releasing the wastes in midwater. At this stage, the latter two options are viewed as being speculative, and require the greatest amount of effort in demonstrating concept feasibility. If the wastes can be maintained containerised their impacts on the deep ecosystems will be substantially reduced, although the acceptability of the introduction of synthetic materials (the bag-liners) into deep-ocean environments would need to be addressed.

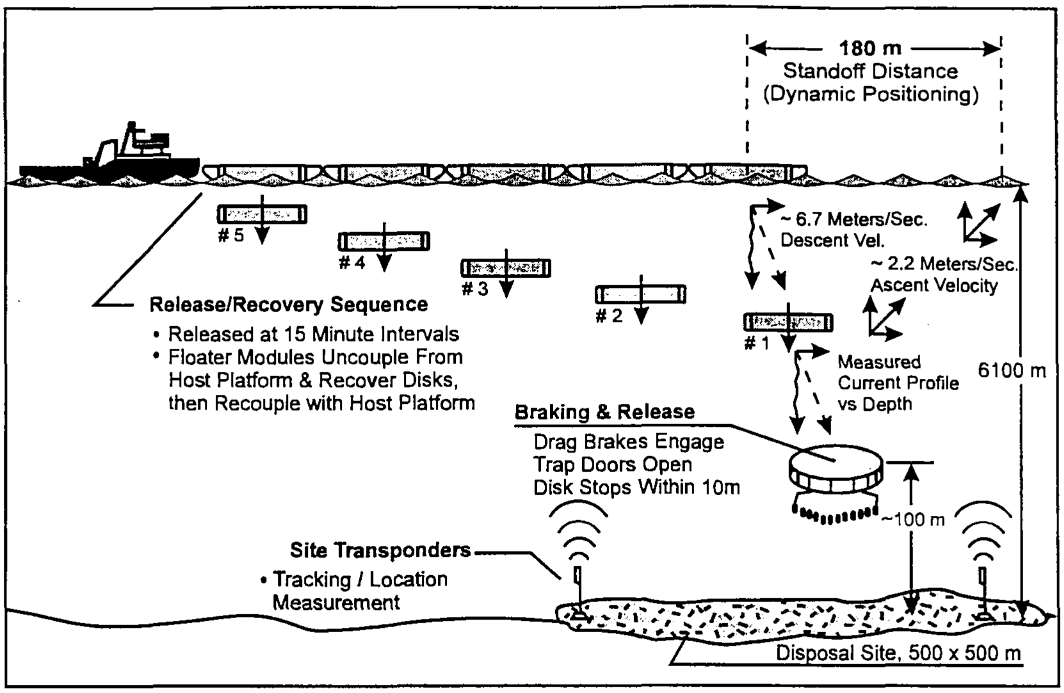

Figure 3.4.6 Sewage sludge and dredge spoil disposal.
Option 6: the direct descent disk concept (modified from Valent and Young 1995).

3.4.3 Impacts of Disposal

Since sewage sludge is largely composed of organic material, it is an indirect analogue for hydrocarbons in the ocean. Sewage sludge from New York and northern New Jersey was, until recently, dumped at Deep Water Dump Site 106 (DWD-106). This locality, where the sounding is about 2500m, is on the continental slope of the Atlantic, 106nm southeast of New York Harbour (Van Dover *et al.* 1992, Takizawa *et al.* 1993, Bothner *et al.* 1994, Lamoureux *et al.* 1996, Robertson 1996 and Sayles *et al.* 1996). From 1986 - 92 about 6 million wet tonnes of sludge were dumped there annually, a total of over 42 million tonnes. The site has been extensively monitored and studied using a variety of pollution indictors. While the dumping continued, there was a build up of significant quantities of organic material on the sea-floor, but some was dispersed into surrounding areas by the bottom currents and by the activities of the benthic biota. Once the dumping ceased, levels of contamination began to decline quite rapidly, but relatively high concentrations of sewage indictors are still found adsorbed onto particulates as far as 90km down current. The site continues to be monitored. The results demonstrate how far, in a dynamic physical regime, uncontained contaminants released at the surface can be dispersed and how long their impacts persist. Many, but not all of these, will be similar to the impacts of plumes of tailings from mining activities. However, extrapolating from effects observed at this dynamic site on the continental slope of the eastern seaboard of the USA in order to forecast what might happen in a much quieter abyssal regime may be misleading.

Disposal of sewage sludge into the oceans will result in a number of potentially deleterious impacts that must be carefully considered before such dumping can be approved as a widely employed and acceptable practise (e.g. Valent *et al.* 1997).

Those potential impacts are:

- the burial of benthic communities
- the clogging of the filtration apparatus of suspension feeders downstream of the dump-site
- increases in turbidity especially in the benthic-boundary layer
- toxic impacts of various sludge components
- reductions in the *in situ* oxygen concentrations as a result of increases in biological oxygen demand by sludge constituents
- changes in community structure around the disposal-site and associated changes in links and flows within the food webs.

a. Burial

This will essentially be a near-field effect limited to the epi- and in-fauna within the disposal site. The deposition of only a few millimetres of sludge may be significant for the infauna. Abyssal plain organisms are used to natural depositions of up to ca. 1cm of phytodetritus (Billett *et al.* 1983, Lampitt 1985, Thiel *et al.* 1989, Rice *et al.* 1994). However, since the sludge is denser than the natural phytodetritus, less may be tolerated. It should be feasible to keep burial impacts from a single load concentrated within an area of 25km^2 (i.e. a radius of just less than 3km).

b. Clogging and turbidity

These effects mostly will affect suspension feeding organisms, especially some of the mucusweb feeding pelagic animals. Suspension feeders in benthic and benthopelagic environments tend to occur either in regions where sedimentation rates are extremely high or extremely low, or where there are hard substrata and rapid current flows. Large and stalked suspension feeders have been observed to recover well from the passage of plumes generated during a large disturbance experiment (Bluhm 1993, Bluhm *et al.* 1995).

c. Toxicity

Deposition of organically rich fine material may well prove to be a food resource for deposit feeders. However, its quantity and quality may have adverse effects on commnity composition and hence diversity, especially if it has scavenged toxic chemicals. Laboratory tests on shallow-living species have shown that both the liquid and the solid fractions of sewage sludge are toxic to some taxa (Fava *et al.* 1985). Fish larvae appear to be particularly susceptible. For example, Costello and Gamble (1992) carried out toxicity tests on the eggs and larvae of herring (*Clupea harengus*) and cod (*Gadus morhua*) in areated experimental chambers. Sludge concentrations of >0.1% had significant and reproducible toxic effects, including premature hatching, increased mortality during hatching and throughout larval development, and suppression of feeding. Blaxter (1977) considered suppression of feeding to be a sensitive indicator of sub-lethal stress. Attempts to identify the toxic components indicated a number of factors which contribute to toxicity. Costello and Gamble (1992) observed that even though dissolved ammonia increased in experimental containers at sludge concentrations >0.2%, pH remained steady at 7.95 over the range of ammonia concentrations they investigated. Elevated concentrations of ammonia occur in discharge plumes; a maximum of 0.81mg per litre was recorded at the Garroch Head dumpsite in the Clyde in Scotland (Norton *et al.* 1981) and 1.14mg per litre in the Thames Estuary (Clarke 1989). Ammonia concentrations had to exceed 7mg per litre before there were significant increases in the mortality of herring larvae. While ammonia is an important factor in sludge toxicity, it is by no means the only toxic component of sewage sludge (Fava *et al.* 1985). Other potential toxicants include heavy metals, organochlorides, PCBs, detergents, oils, greases and phenols (Weis and Weis 1981). Copper concentrations of 30µg per†litre are toxic to herring embryos (Blaxter 1977) and such levels were reached at concentrations of 0.2% sludge in Costello and Gamble's experiments. Clearly the toxicity of sludge is increased by contamination with industrial effluents. Generally larval stages tend to be sensitive to suspensions of sewage sludge, so disposal at depths or at times when larval abundances are minimal will reduce the overall impact on the assemblages in the water column.

Toxicity experiments have also been carried out on the effects of sewage sludge on algae, polychaetes, molluscs, crustaceans and echinoderms (Read 1977, Chapman *et al.* 1986, Fava *et al.* 1985, Miller *et al* . 1987, Anderson and Hunt 1988, and Frithsen *et al.* 1989). These indicate that the impacts are very variable between the different taxa. This is further born out by surveys across the Garroch Head dumpsite reported by Pearson (1986). This dumpsite was at a location where the physical mixing was minimal and so it was an accumulative regime where the sludge piled up in the immediate vicinity of the central disposal position. Effects were detectable for a radius of 5km (Fig. 3.4.7). Around the fringes the infaunal benthic standing crop began to increase towards the centre. Numbers of both species and individual animals per unit area also increased towards the centre. Biomass and individuals per unit area continued to increase to a maximum at the centre of the dumpsite, whereas the numbers of species per unit area reached a maximum ca. 3km away from the centre, and then declined sharply to less than half the numbers found in "unperturbed" regions beyond the periphery of the dumpsite. Thus, the central region was dominated by vast numbers of a very few small species, mostly nematodes. The exchange of water over the dumpsite was enough to maintain adequate concentrations of dissolved oxygen, and the increased availability of organic material attracted large numbers of fish which were exploited commercially. Thus the sludge dumped at Garroch Head was toxic enough to exclude some species allowing more tolerant species to flourish.

There have been very few toxicity experiments carried out on deep-living species. The physico-chemical environment tends to be less variable in deep-water than near to the surface, so the species may be less tolerant of contaminant inputs. However, there are environments where natural levels of elements and compounds are far in excess of levels in coastal waters, for example in proximity to hydrothermal vents, cold saline and hydrocarbon seeps and in the vicinity of the warm (64°C) metalliferous muds of the deep Red Sea (Thiel *et al.* 1986, Karbe 1987). Concentrations of contaminants which would cause alarm in shallow water, may characterise the natural environment of some deep water species. Furthermore it is the larval stages of species that tend to be the most sensitive to toxins. Since deep-living species tend to have very different reproductive strategies to shallow-living species, tending to produce fewer yolk-rich eggs or having direct development, their larvae may not be so vulnerable to toxins. However, larval life, time for contaminant accumulation, and the vulnerable phase may be considerably prolonged. The phenomena need to be examined experimentally.

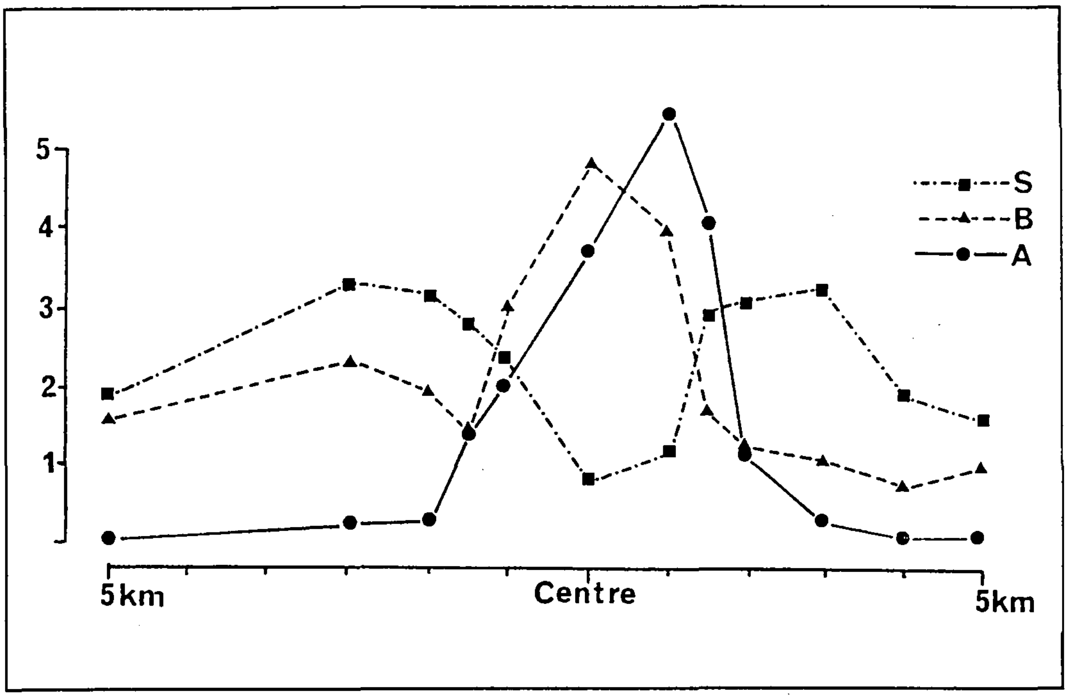

Figure 3.4.7 Concentrations of benthos at, and around the Garroch Head sewage sludge dumpsite, where up to $1.5 \cdot 10^6$ tonnes of sludge was dumped per year. A: Abundances of benthic infauna (numbers $\cdot$ $10m^{-2}$); B: Benthic biomass (g wet weight $\cdot$ m^{-2}); S: Numbers of benthic species ($\cdot$ m^{-2}); (from Pearson 1986).

d. Oxygen demand

Sewage sludges have a high oxygen demand caused by the biological and chemical oxidation of their organic contents (e.g. Goldberg 1995). When dispersed in aquatic environments the microbial digestion of the sludge can draw down the ambient oxygen levels, and unless oxygen is rapidly re-supplied, it may become exhausted. The digestion of the sludge is initially taken over by sulphur-bacteria and then by methanogenic bacteria. Under anaerobic conditions sulphate bacteria are able to thrive by oxidising organic compounds reducing sulphate ions, thereby releasing hydrogen sulphide which is extremely toxic to aerobes. In the absence of oxygen, once the sulphates are used up, the methanogenic bacteria take over the degradation. The rate of methane production decreases with temperature and is inhibited by pressure. Thus, microorganisms introduced into the deep sea together with sewage sludge will be killed or may escape death in the form of a resting stages. Local organisms may eventually take over the degradation processes. Whether psychrophiles and barophiles can cope with the organic matter from the sludge needs further investigation.

Oxygen occurs dissolved in most (but by no means all) sea-water at great depths in sufficient concentrations to support aerobic respiration. However, the oxygen will be exhausted faster in the Pacific deep sea where its concentration is only 2-3ml $\cdot$ l^{-1} in contrast to the Atlantic bottom waters containing 5 to more than 8ml $\cdot$ l^{-1}.

e. Community structure

Following sewage sludge disposal there will, undoubtedly be local changes in the community structure both in the benthos and in the benthopelagic zone. Enhancement of the organic inputs to these otherwise resource-limited systems would probably attract scavengers, and favour deposit-feeders and species with high reproductive potentials. There are significant differences in community structure between regions where there is a marked seasonality in organic sedimentation and those where the inputs are more consistent throughout the year (Rice *et al.* 1994). Where the inputs are seasonally pulsed, sediment-feeding megafauna such as holothurians become dominant contributors to the benthic standing crop. Similar effects are seen beneath the equatorial upwelling zone in the Central and Eastern Pacific (Smith *et al.* 1994, Smith *et al.* 1996). Community changes would be most extreme if anoxic conditions develop locally within the benthic boundary layer, as indeed occurs naturally at bathyal depths in the Eastern Tropical Pacific and in the South-east monsoon season in the North-west Arabian Sea at intermediate depths. However, the most likely effect on enriching a local patch of the seabed would be to attract mobile scavengers and predators to feed either directly on the organic sludge or to prey on scavengers. But since these suppositions are deduced from shallow water dump sites, they need to be verified for deep ocean conditions.

f. Food-webs links and flows

The attraction of mobile scavengers will also influence the food-web links and the flow of materials through the communities in the far-field. In modelling the possible pathways along which radioactive wastes disposed on/in the sea-bed might be returned to the upper ocean, Rowe *et al.* (1986) concluded that, apart from the thermohaline circulation which would take of the order of 200 years, the most significant potential flux would be via large scavenging fishes such as *Coryphenoides rupestris*. Some of these scavengers are cosmopolitan in the deep basins of the oceans. Some of the very large amphipod scavengers, such as *Eurythenes gryllus*, also appear to be cosmopolitan (Thurston 1979), though the extent to which there is significant transport of contaminants and pathogens along food-chains will depend on the degree to which there is any accumulation. Some heavy metals (e.g. mercury and lead) and persistent organic compounds (e.g. PCB's and organochlorides) do tend to accumulate in the body tissues of consumers resulting in biomagnification along food-chains (Phillips 1995). However, the majority of substances are de-toxified and/or excreted. There is a real concern that of the 100,000 chemicals in permanent use by industry only 100 - 150 are regularly monitored (Phillips and Rainbow 1990). Thus it may well be a sensible precaution to isolate wastes contaminated with such substances as much as possible. In general. the more remote the disposal site the less likely any significant quantities of toxins will be transferred along the food-chain. The deep ocean disposal would result in far lower transfers of bioaccumulating contaminants than is currently occurring as a result of discharges into fresh and coastal waters and into the atmosphere.

Advection will govern the transport of material from the release site, and this will be determined by the mean currents. Diffusion governs the rate of dilution after the immediate mixing which occurs as the plume is discharged. Diffusion in the open ocean is determined by current gradients and turbulence. Since the ocean is stratified, with density (and in most regions temperature) decreasing with depth, material dissolved and suspended in the water will spread along surfaces of constant density, or isopycnals, which are nearly horizontal. This can be seen in the way CFCs (chlorinated fluorocarbons) entering the ocean interior at convergences are spreading horizontally rather than vertically (Smethie 1993). Material released into sea-water will spread, initially slumping under buoyancy forces, and then will be carried by the mean currents.

g. Fertilisation of oligotrophic regions

Fertilisation and the resulting increase in productivity as a result of anthropogenic activities are often regarded positively. This view is partly based on the principles of the market economy, supported by many economists, promulgated by the media, and often uncritically assumed to be valid for natural populations. But energy increase in the "blue water ocean" does not necessarily have a beneficial effect. There may be some increase in primary production, but this may result in a new community structure with a decrease in diversity and a loss of species and genetic variability. Moreover, since natural systems cannot be managed like the economy, the higher production may not favour commercially valuable species.

In the case of the oligotrophic deep sea, the consequences of fertilisation cannot be predicted accurately. The common assumption that they would be beneficial should, therefore, not be transferred to this environment uncritically.

3.4.4. Research Required to Evaluate Impacts

This brief résumé of some of the detrimental environmental effects of sewage sludge disposal in shallow water has been used to infer what types of biological and chemical impacts might be expected to happen at deep water sites. However, such extrapolations carry many uncertainties, so an experimental approach will be needed both to validate the impacts and to quantify their extent. A set of integrated experiments ranging in time and space scales will be required to establish whether the impacts of any disposal will indeed remain within the predetermined limits of acceptability.

In common with evaluations to establish the impacts of any mining, such an experimental approach to investigate deep ocean dumping should have four phases. Initially a disposal site will be selected and a similar control site identified. These selections will probably be based on sites for which substantial amounts of data are already available. At least, this would reduce the overall effort.

Pre-experimental and design phase. Once selected, both the experimental and the control sites would need to be throughly surveyed - topographically, physically and biologically. An array of sampling positions will need to be established based on the residual current regime in the benthic boundary layer. Numerical modelling will be an integral part in this phase to develop predictions on plume generation and transport. The monitoring of a control site is needed to act as a benchmark against which any changes observed at the experimental site can be evaluated. It will be important to be able to differentiate between long-term changes that may result from the experimental intervention and any induced by natural variations. Selection of the disposal technique to be used will determine many aspects of the experimental design. For example, if the wastes are to be discharged from a pipe-riser, the behaviour of the plume will need to be thoroughly investigated. If free-fall containers are to be used then the behaviour and fate of full containers both in mid-water and on hitting the sea-bed will need to be known, that the arrays of monitoring instruments can be designed optimally.

Experimental Phase 1 would involve an initial experimental disposal of a single load of about $10^5 m^3$ of sludge within an area 1 km in radius. This is an amount that would be equivalent to the load of a single carrier. The effects of this limited discharge would need to be monitored over a year to assess the spread and duration of the impacts. The numerical models need to be tested and refined. Some knowledge of the rate at which deep benthic ecosystems recover from extensive perturbation already exists from the DISCOL experiment, but this was conducted in an oligotrophic region where recovery may be naturally slow; also the addition of copious quantities of organic matter may result in either quicker recoveries, ore more persistent effects. Assuming the impacts remain within the predetermined level of acceptability, the second phase would ensue.

Experimental Phase 2 would cover the disposal of further loads totalling 10^6m^3 over a year, within the same limited area, although pragmatically an extension of the disposal site to an area 2 - 3km in diameter might be contemplated. A further 12 months of evaluation would follow, and then assuming that once again the limits of acceptability had not been exceeded, the third and final experimental phase would be undertaken.

Experimental Phase 3 would would be the PDO (see Chapter 2.1) equivalent to a PMO for mining evaluation. However, in this case the primary emphasis would be on environmental effects and not necessarily involve economic evaluations. This phase would constitute a decade of regular disposal of 10^6m^3 of sludge per year. The total duration of the follow up monitoring needed to complete this phase cannot at this stage be determined, but planning on eight years of monitoring does not seem to be too unrealistic. Thus the total programme of evaluation would extend over two decades if the results are comprehensive to evaluate the environmental safety of deep ocean disposal.

The monitoring required at all phases will be similar in scale to that needed for the evaluation of mining and will consider both physico-chemical and biological impacts. The latter will include near-field and far-field changes induced by burial, clogging, turbidity, toxic inputs draw down of dissolved oxygen, and ecological perturbations caused by changes in food-web linkages and energy flows (see under Chapter 3.4.3).

3.5. Dredge Spoils

3.5.1. Description of Waste Materials

Dredging estuaries, harbours and navigational channels produces a wide range of muds, sands and gravels. Where the bottom muds are uncontaminated disposal presents no special problems - indeed clean dredged sands and aggregates are used extensively for land reclamation, building materials and road construction. However, the majority of spoils dredged from close to ports, oil refineries, power plants (nuclear and coal-burning), industrial complexes notably those including chemical manufacturing, and where heavily polluted rivers and outfalls discharge into coastal and estuarine waters are contaminated with pollutants ranging from heavy metals, hydrocarbon products to synthetic organic residues such as poly-chlorinated biphenyls (PCBs) (GESAMP 1990, OSPARCOM 1993). The sediments range in particle size from coarse-grained sediments (sands 62.5 - 2000μm and gravels >2000μm to fine-grained clays (<3.9μm). Bulk densities depend on the mineral and water contents, but average around $1.65 \cdot 10^3 kg \cdot m^{-3}$ with a water content of about 60%. Contaminant levels vary widely from place to place depending on the sources, but spoils dredged from industrialised ports and estuaries frequently contain levels of heavy metals, hydrocarbons and synthetic organic compounds that are too high for the spoils either for them to be discharged back into the sea or to be used for reclamation. However, the contaminant levels are so low that it would not be cost effective to clean them up.

A further problem is that in brackish estuarine conditions many heavy metals are precipitated and remain chemically bonded to the very fine silt and clay particles in anoxic conditions typical of the sediments (O'Reilly Wiese *et al.* 1995). The bonding of metal ions to these fine particles changes with the redox conditions. When the spoils are discharged, the clay particles remain in suspension far longer than the larger particles. In well oxygenated conditions the oxidative states of the metals change so that they tend to be released back into solution in the water and again become biologically available. Hence these spoils can neither be used for land-fill nor for reclamation. Nor is it desirable to dispose of them back into coastal waters where their heavy metal and synthetic organic contents can be taken up by biological processes and enter the human food chain. Even so, in the absence of any better alternative, it seems likely that their discharge back into coastal waters will continue to be permitted (McIntyre 1995). A persistent problem with disposal of all forms of waste materials into coastal seas is that contaminants and pathogens (e.g. from sewage sludge) are returned and maintained in inshore waters where they can have a deleterious impact on the water quality and the health of the local human population (e.g. Collins *et al.* 1980).

There is a further possible long-term problem. If sea-levels rise, sea-water will penetrate further up estuaries. Regions that presently are brackish will become progressively more marine, and this will favour the re-mobilisation of metal contaminants presently sequestered in the muds back into solution. Thus estuaries may become an even more important source of pollutants for coastal seas. Cleaning up the legacy of the industrial revolution, presently locked up within anoxic muds in European estuaries, may turn out to be a serious problem for future generations.

3.5.2. Description of Disposal Techniques

The potential disposal techniques for dredge spoils are similar to those discussed for sewage sludge (Chapter 3.4.2) and need not be re-iterated. Valent and Young (1995) compiled some transport criteria (Table 3.5.1) which will be useful both for site selection and for the design of suitable carriers for the spoils.

3.5.3. Impacts of Disposal

In general the expected impacts of the disposal of dredge spoils into the deep ocean will be very similar to those discussed for sewage sludge in Subchapter 3.4 above, but the relative severity of the different impacts will differ. The disposal option selected and the specific characteristics of the spoils will also influence the type and severity of the impacts. Disposal from the surface (Option 1) is currently used for disposal in nearshore waters, and may be used further offshore in the future to reduce contaminant impacts on inshore waters (McIntyre 1995). However, apart from removing the impact further offshore and away from regions where living resources are exploited, it provides few environmental advantages to offset the greater economic costs. The only additional advantage of discharging at midwater depths in the deep ocean (Option 2) is to reduce impacts on the euphotic zone. Discharging on to the sea-bed either uncontained (Options 3) or containerised (Option 4) offer the advantages of placing the spoils as far away from living resources as possible and effectively isolating and/or diluting inputs of contaminants from Mankind's ambit. Impacts on the sea-bed will be very different when the spoils are contained in geotubes which isolate the waste material from the environment in comparison to the release of particulate masses above the seafloor.

Table 3.5.1 Operational and systems performance criteria used for designing the deep ocean disposal system (extracted from Valent and Young 1995)

System capability:	a.	2.5 million metric tonnes per year per port.
	b.	Maximum transit distance of 1000nm (1852km).
	c.	Total absence of leakage into intervening water column.
	d.	Dissipation of static electricity.
	e.	Validation and verification of design features.
	f.	Range safety design - tracking system to allow minimum stand off of 1km between surfacing submersible and host platform.
Transiting speed:		12 knot (6.2m $\cdot$ s^{-1}) minimum.
Operational depth:		Maximum 6,700m.
Emplacement accuracy:		Within an area of 0.5 $\cdot$ 0.5km
Reliability:		Mean-time before failure: > 700h
Maintainability:		Mean-time to repair: < 80h
Environmental:	a.	Operational in sea state 5.
	b.	Survival in sea state 8.
	c.	Withstand surface currents of 1.5kts (0.78m $\cdot$ s^{-1}) and bottom currents of 0.75 kts (0.38m $\cdot$ s^{-1}).
	d.	Withstand pressures of up to 62MPa.
	e.	Temperature range 0° to 49°C
Waste stream capability:	a.	Contaminated dredge spoils of 32% solid by weight.
	b.	Sewage sludge up to 20% solids by weight.
	c.	Municipal incinerator fly ash up to 85% solid by weight,
Design requirement:		Meet national and international safety regulations for shipping

The main bulk of all spoils are inert, non-toxic mineral particles. Discharging uncontained material directly on to the sea-bed at abyssal depths will limit the impact almost exclusively to the benthic and benthopelagic faunas. The expected impacts will be much the same as for sewage sludge: burial, clogging, turbidity, toxicity, increased oxygen demand, changes in community structure and alterations to food web links and material flows.

Effects of burial within the near-field by spoil masses may be more severe, but because the grain size composition of dredge spoils tend to be larger than in sewage sludge, sedimentation rates will be faster, thus reducing the lateral spread and localising the impacts. According to Stoke's Law the sedimentation rate of the finest sand particles (50μm diameter and densities of 2.5) is 0.1cm $\cdot$ s^{-1}. Thus if these fine sand particles are discharged from a stationary pipe 100m above the bottom they will only take a day to reach the sea-bed. Mean current velocities in the north-east Atlantic at abyssal depths are around 2-3cm $\cdot$ s^{-1}, i.e. ca 2-3km $\cdot$ d^{-1} (Dickson *et al.* 1985). An area in the order of $10km^2$ would be effected to some extent with sands or coarser particles, whereas silt particles will be dispersed over very wide areas unless coagulants are added. Disposal of 10^5m^3 of uncontainerised spoils at a height of 100m above the sea-bed, would result in the sea-bed being covered to a mean depth of ca. 1cm within the area. The centre of the disposal site would be covered to a greater depth, especially if larger sized particles were abundant, or if the spoils were discharged closer to the bottom. Spoils that are containerised will bury a far smaller area providing the containers remain intact, but sediments in the immediate vicinity of the impact zone will be thrown into suspension. These effects will be analogous to those created by ploughing during the DISCOL experiment in order to mimic some of the mechanical impacts of sea-bed mining (Thiel 1991, Foell *et al.* 1992). Experimental trials will need to be carried out to assess what area will be affected if the containers burst on hitting the bottom. It should be feasible to restrict near-field effects of dropping a series of containers to an area of $<10km^2$. However, some may fail as the containers are being dropped from the transporter or in midwater. This would result in the contents being dispersed over a much wider area of the seafloor and having some effect on midwater communities. Nevertheless, these effects will be no worse than those resulting from the present discharges of spoils at the surface.

Burial of the natural communities with clay-sized spoils will create a novel environment which will be re-colonized by communities likely to resemble those which re-colonize areas of sea-floor covered with turbidite deposits. This will increase the structural diversity of the sea-bed of the general disposal site. Burial of very extensive areas of abyssal sea-floor does occur naturally following mass failures of continental slopes, usually at times of rapid sea-level change (Weaver *et al.* 1995). Since the particulate material of the dredge spoils will most likely be coarser than the sediment at the disposal site, some alteration in community structure, from the original set of species or from turbidite species composition, is to be expected. The predicted change should result in a decrease in species number and diversity since not all the species of the surrounding community will be able to cope with the dredge spoil conditions. However, pre-operational surveys of possible sites will need to ascertain whether the scaling and patchiness of the benthic communities is such that a significant percentage (say >0.1%) of any one type of community within that ocean basin is possibly restricted within the near-field.

During deposition and particle settlement there will be clogging of the gills and filtering structures of suspension feeders by the finer components of the spoils which will remain in suspension longer and so be advected some distance away from the disposal site. The dispersion (and dilution) will increase if spoils are released higher in the water column. Silt particles with diameters of 10, 5 and 1μm will take about 20 days, 3 months and 6 years respectively to sink 100m. These finer components are important because not only will they carry most of the organic carbon contained in the spoils absorbed on to their surfaces, but they will also carry most of the toxic load (heavy metals and synthetic organic carbon compounds). The addition of flocculating agents to the spoils would not only accelerate their deposition but also reduce their dispersion. They must be chemically inert or easily biodegradable. The overall impact of such fine suspended material will be analogous to that caused by the discharge plumes from mining activities, but the quantities involved, and hence the severity of any impacts, are likely to be far less.

Toxicity effects caused by heavy metals, hydrocarbon residues and synthetic organic compounds in the spoils will be similar to those discussed in Chapter 3.4.3.1 for sewage sludge, but will be relatively more important if contaminated spoils are being disposed of. However, the quantities of metals being mobilised from dredge spoils, and hence the overall impacts of toxicity, are likely to be less than those resulting from deep-sea mining activities. Hence it will be the impacts of the hydrocarbon residues and synthetic organics that would need most careful monitoring in any experimental programme, but the metal components must be investigated also. It may also be possible to moderate longer-term toxic impacts of highly contaminated spoils by subsequently capping the spoils with uncontaminated, non-toxic spoils. Alternatively, if containers of spoils do not break up on impact, their contents will remain isolated from the water.

Most dredge spoils will be richer in organic matter than the sea-floor sediments, but far less organically-rich than sewage sludge. Thus their impact on local oxygen demand will be minor and unlikely to induce substantial changes, especially in those regions where the bottom water is well oxygenated. However, community structure and material flows will be altered to some extent as a result of the additional organic supply, but the major effect will be as a result of the eradication of the sea-bed fauna in the near-field. The area affected by dredge spoil disposal will be very small compared with areas affected by naturally occuring geological events. The impacts of containerised material will be different from those generated by a stream of unconsolidated sediment flowing down to and spreading across the sea-bed. The containers, assumed to remain intact and to isolate the dredge spoils permanently, will collide with the seafloor at their arrival and will create a sediment plume which will drift downstream and eventually resettle. Most of the impacts will be restricted to the disposal area, while the resedimentation of the plumes from local sediment may not be a severe impact. The polyethylene fabrics of the geotubes may provide a hard substrate upon which epifauna may settle.

3.5.4. Research Required to Evaluate Impacts

The choice of a disposal option will determine the mode of the experimental investigation. Since theoretical prediction and numerical modelling cannot describe the true impact on the seafloor and its community, reliable environmental impact assessment can be derived only through dredge spoil disposal experiments. The size of such an experiment must be large enough for the results to be scaled up to full industrial level, and a series of integrated experiments, similar to those recommended for sewage sludge, will be needed.

Initially, feasibility studies to establish which are the most appropriate techniques (both in environmental and economic terms) for delivering the spoils to the sea-bed, are to be conducted. Discharging from a pipeline will result in greater dispersion of the spoils on the bottom. However, the near-field area will be minimised by having the pipeline discharging as close as possible to the sea-bed. For an initial experiment with limited amounts of dredge spoil, concentration in a restricted area is of importance for later extrapolation to larger experiments and to potential commercial-scale disposal. Potentially, containerisation in "geotubes" or other similar methods are likely to be more acceptable environmentally, only if the vast majority of containers remain intact during discharge and descent to the sea-bed, and after impact on the bottom.

Two comparable sites will need to be selected for the experiment and surveyed. One site will be used for the disposal experiment, the second will be a control site to monitor any long-term natural fluctuations in the deep ocean ecosystem. The initial surveys will be needed to characterise the sites and to establish the baseline against which all observed changes can be evaluated. The hydrographic regime at each site will need to be established, along with the bathymetric, sedimentological and ecological characteristics. With due regard to the hydrographic regime, a grid of sampling stations would be established so that the spread of the spoils and contaminants could be assessed, the dispersal of any sediment plume followed and the resultant changes in the bottom-living communities, from impact to recovery, monitored. Recovery from a moderate-scale disturbance in the deep ocean can be expected to take several years on the basis of the results from the DISCOL experiment (Bluhm *et al.* 1995).

This slowness of recovery (or low resilience) poses an environmental problem. Rather than a few large permanent sites, would it be better to use a series of smaller sites in rotation, with each site being used at a frequency allowing total or substantial recovery between disposals. Valent and Young (1995) argued in favour of the latter option, but if recovery rates are exceedingly slow it may be preferable to concentrate the impacts on a few permanent sites. In this way, disposal of dredge spoils will have far less impact than mining polymetallic nodules which must, by its very nature, be more peripatetic. Another approach might be to utilise areas already affected by mining activity, thus minimising the total area severely degraded by exploitation.

Survey techniques needed both for the initial baseline studies of the selected sites and the subsequent monitoring will require the use of deep towed sensors fitted with side-scan sonar, transmissometers, and chemical sensors. These will be used to map the near-field area and to track the dispersal of sediment plumes. Photographic (and/or) video surveys must be used to map the distributions of megafaunal species and provide the reference points for grab, corer and trawl surveys to examine the responses in the sea-bed and benthopelagic communities. A disposal site is likely to be surrounded by concentric "rings" (albeit distorted by the *in situ* current regime) of communities showing diminishing levels of impact away from the centre of the disposal-site, analogous to the effects of contamination in the vicinity of oil installations in shallow water (Olsgard and Gray 1995).

The experimental disposal would be based on a phased approach although phases 1 and 2 might run concurrently at separate sites:

Phase 1: Disposal of $5 \cdot 10^5 m^3$ is monitored for at least 12 months.

Phase 2: Disposal of a series of similar sized loads amounting to $10^6 m^3$ over a year and monitored for at least three years

Phase 3: Disposal of $10^6 m^3$ per annum for a decade at one selected site. Monitoring would be required both during and for at least five years after the cessation of the experiment to establish recovery rates.

There are two major factors that need to be evaluated: the total area affected by the experiment and the time taken for the ecosystem both in the far-field and the near-field to recover from an impact. Restablishment of pertubated communities even in shelf-sea areas has received relatively little attention. Recovery of shallow water coastal communities was clearly well underway 7.5 years after the cessation of the extensive dumping of colliery spoils at Horden, County Durham off the north-east coast of the UK (Johnson and Frid 1995). But at another site, where coal wastes (the main contaminant) had continued to accumulate as a result of the prevailing current, the communities still showed clear signs of perturbation more than 12 years after the dumping had stopped. In the deep-sea, recolonization of experimental, defaunated sediment samples deployed from a submersible at a depth of 1760m proceeded much more slowly than similar samples emplaced in shallow coastal waters (Grassle 1977, Levin and Smith 1984, Desbruyeres *et al.* 1985, Smith 1985, Smith *et al.* 1986, Grassle and Morse-Porteous 1987, Snelgrove *et al.* 1996). So the resilience (i.e. the rate of recovery) of deep-sea communities is much slower than of shallow communities, and this will make them more susceptible to synergistic effects; i.e. a series of perturbations will result in incremental effects if they occur at a frequency which does not allow the community to recover fully from previous perturbations. However, extrapolation from such very small-scale ($<1m^2$) experiments to what might happen at industrial scale disposal sites is probably misleading because re-colonization by lateral migration into the affected area - the important means of re-colonization and recovery in Grassle's experiments - will be too slow to be significant for perturbed areas many kilometres across. These sorts of impacts are now being studied in relation to manganese nodule mining (see Subchapter 2.1).

Thus the experimental approach advocated will cause some temporary damage to benthic, and possibly benthopelagic, organisms in the immediate vicinity of the disposal site. This situation will be similar when the experiment would employ containerised dredge spoils. Again, the experiment would be conducted in a number of phases, starting with a few dozen containers and subsequently adding containers after intervals of time (12 months) have passed. Each deposition would be followed by close-up investigation prior to additional experimental disposals. Total disposal quantities should be adjusted to those planned in the experiments proposed for uncontainerised dredge spoils.

The experimental depositions proposed, both in their single steps and their full extents, will aid in environmental impact assessments. They are designed to allow extrapolation of impacts to commercial-scale dredge spoil disposal, but they are large-scale experiments exerting inherent impacts of their own on the deep sea environment. The acceptability of such large-scale experiments in terms of environmental aspects is yet to be discussed (see Chapter 4).

3.6. Carbon dioxide

3.6.1. Description of Waste Material

It is well established that emissions of anthropogenic carbon dioxide (CO_2) to the atmosphere will almost certainly contribute to climate changes (Houghton *et al.* 1995), and that introductions of anthropogenic CO_2 from the atmosphere to the oceans during the past 200 years have already lead to noticeable changes in ocean carbon chemistry (Sarmiento *et al.* 1992, Quay *et al.* 1992, Chen 1993). The motivation for investigating mitigation options, which include the concept of disposing of carbon dioxide either in liquid or solid form on to the sea-bed at abyssal depths or its dissolution in sea-water, is provided by the steady increases in atmospheric CO_2 concentrations resulting from the present emissions of anthropogenic CO_2 of about 6Pg $C \cdot yr^{-1}$ (1Pg = 10^{15}g = 1Gt = 10^9 tonnes). This is equivalent to about 1% of the atmospheric reservoir of CO_2. These emissions are on track to double or triple within a century, unless real progress is made both in reducing emissions, strongly aided by international conventions, and in the development and application of techniques for long-term storage of carbon dioxide. Estimates suggest that, at the present rate of consumption with increases proportional to the growth in human population, all known reserves of fossil fuel will have been burnt within 150 to 200 years, and atmospheric carbon dioxide levels will have risen to between 925 and 1750ppm (Cole *et al.* 1995); i.e. possibly to as much as four-times the pre-industrial levels. At present the largest emissions come from the industrialised nations, which also have the highest emission per capita (Figs 3.6.1 and 3.6.2). However, some developing nations, which at present emit relatively low quantities, are likely to increase their outputs substantially. For example, China which has about 25% of the World's population, has an understandable policy to increase the general standard of living of it's population by increasing industrial outputs. Since China's only major fuel resource is coal, its emissions of carbon dioxide per capita are likely to increase sharply. However, increased industrialisation will be only one of the contributing factors. At present 15-17% of all energy generated is used in food production (Pimentel *et al.* 1994). With the World population set to double during the 21st century, energy demands associated with food production will increase and more of the land surface will have to be deforested for agriculture. Felling more of the forests will reduce terrestrial uptake of carbon dioxide and hence contribute to the rate at which its concentration in the atmosphere increases (e.g. Keller *et al.* 1996).

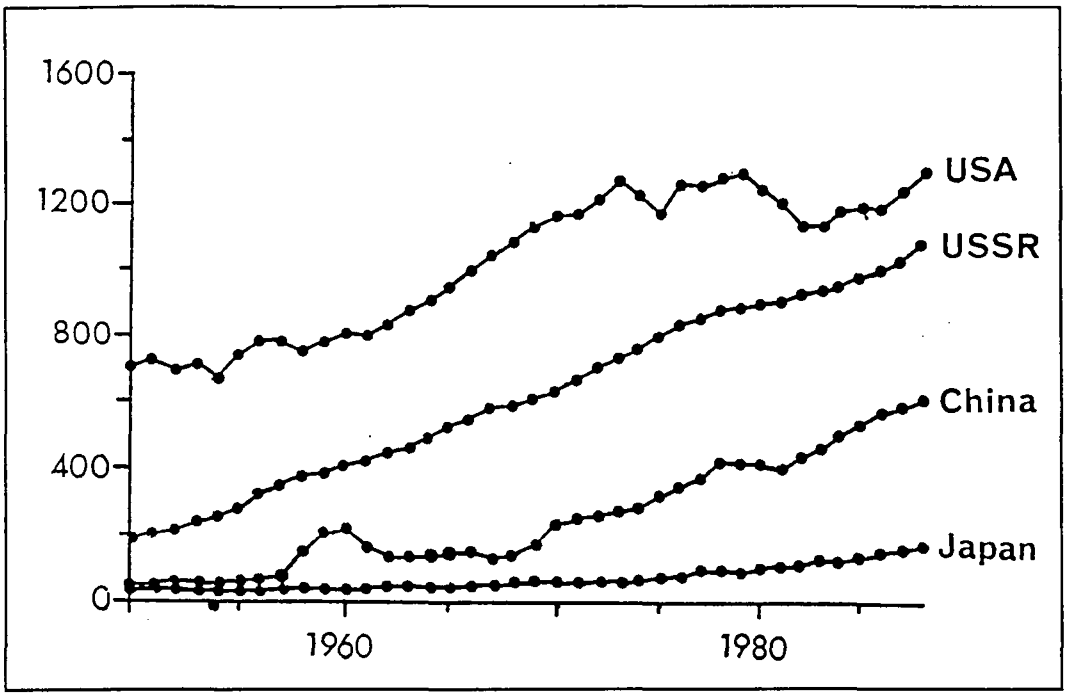

Figure 3.6.1 Total annual emissions of carbon dioxide (millions of tonnes) by four major countries (from Marland *et al.* 1994).

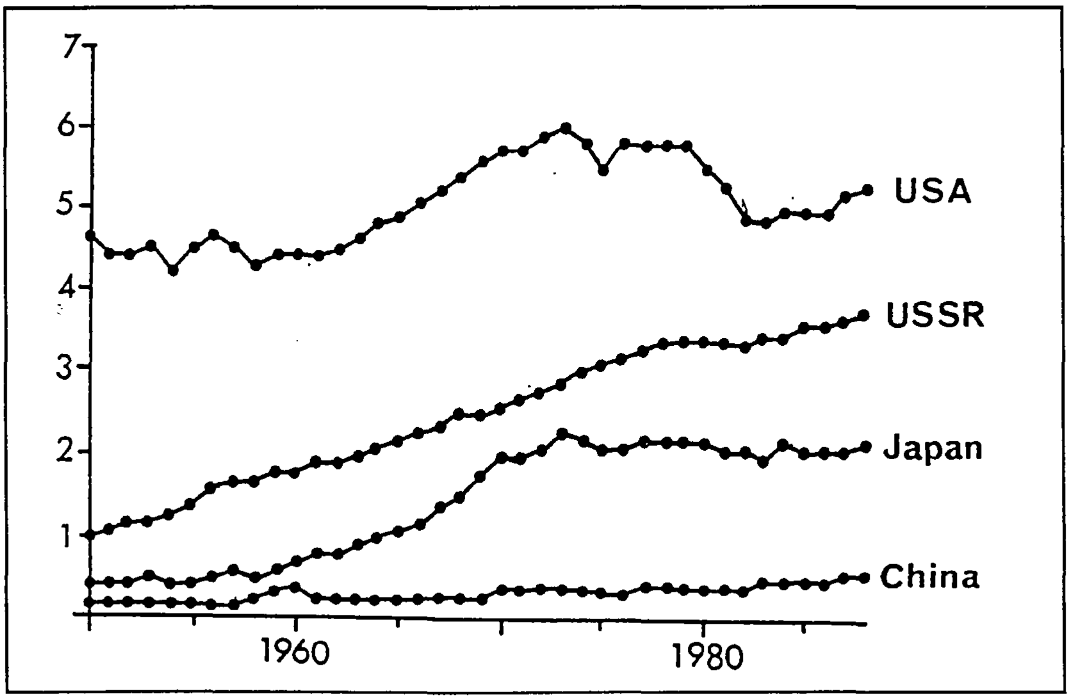

Figure 3.6.2 Annual per capita emissions from the four major countries as in Figure 3.6.1 (from Marland *et al.* 1994).

The volumes of carbon dioxide requiring disposal are quite staggering. A 500 megawatt (MW_e) power plant burning pulverised coal will generate a mass flow of about 115kg of CO_2 per second, which amounts to 3.6 million tonnes per annum. This contains about 1 million tonnes of carbon and is equivalent to 0.016% of the World's present atmospheric loading from anthropogenic sources). At present about 57% of the total annual emission remains in the atmosphere, while the rest either enters the oceans via air-sea exchange or is absorbed by terrestrial reservoirs (Cole *et al.* 1995). Over a millennium time scale, there will be a transient atmospheric maximum which will be reached in 150 - 200 years, and will then decline as partial pressures of carbon dioxide in the atmosphere and oceans equilibrate in about 1000 - 1500 years. The question that needs answering is whether the costs (economic, social and environmental) that would be incurred in reducing the size of the transient peak in atmospheric concentrations are likely to be less than the "business as usual" option of taking no amelioration actions whatsoever. The possible global environmental and socio-economic implications of the greenhouse effect are so devastating that, even if some doubt still prevails about the role of carbon dioxide in climate change, the precautionary principle requires serious and urgent attention be paid to researching options for the reduction of the potential impacts.

Amongst the options that are being advocated is the use of deep ocean storage as not only being a practical, but also one of the best options for such amelioration (Ormerod and Angel 1996). A practical alternative for Northwest Europe is discharging carbon dioxide into geological strata, and beneath the North Sea suitable deposits have been located with the potential to accommodate about 800Gt CO_2 (Holloway *et al.* 1996) which is equivalent to >25% of the atmosphere's present carbon dioxide content. In 1990 the discharge of CO_2 into the atmosphere by power generating plants in the European Union and Norway amounted to 950 million tonnes (i.e. just less than 1Gt and about a third of the total emissions from all sources). The Norwegian hydrocarbon industry is beginning to pioneer a disposal approach using geological formations. Natural gas from the Sleipner West field contains up to 9.5% of carbon dioxide, and this CO_2 has to be stripped out before the methane can be introduced into the gas supply. As a result about a million tonnes of CO_2 will be injected, via a specially drilled well, into a deep aquifer 250m thick and 800m beneath the sea-bed in the Utsira formation, thus avoiding a $50 per tonne carbon tax which was introduced by the Norwegian Government in January 1991 (Kaarstad 1994). While this approach may be ideal wherever there are suitable geological deposits, many regions throughout the World remain where alternative approaches will be needed. It is for some of these regions that deep ocean disposal may be the best practical option. However, we note that developments at the London Dumping Convention (LDC) may soon lead to carbon dioxide being classified as a product of industrial processes and, therefore, banned from being disposed of in the ocean. Once more is understood about impacts of soaring carbon dioxide levels in the atmosphere, this decision may need to be re-negotiated. Moreover, the authors of this report consider that such developments in the LDC must not preclude conducting the long-term research that will be required to evaluate the deep ocean option, since foreclosing it at this stage would seem most unwise.

The oceanic inorganic pool of carbon dioxide is about 60-fold larger than the atmospheric reservoir. The geological engineering solution to discharge carbon dioxide was first proposed by Marchetti (1977) who proposed that CO_2 might be discharged into the Mediterranean outflow which cascades to depths of around 1km as it enters the North Atlantic. Capacity calculations based on carbonate ion concentrations indicate that the deep oceans can absorb at least 1200Gt of carbon dioxide without significantly changing the chemical balances of sea-water (Cole *et al.* 1995); i.e. twice the CO_2 contained in the pre-industrial atmosphere. This input will result in general reductions of sea-water pH by a few tenths and increase the area of sea floor that is in contact with water undersaturated with calcite. Dissolution and reaction of sediment calcite will, if it proceeds fast enough, serve to increase further the capacity of the ocean, possibly enough to accommodate the total 2000 to 6000Gt that might ultimately need to be accommodated in the deep ocean.

At present about a third of the total CO_2 emissions come from large point sources (power plants and industries, notably cement manufacturing). These point sources, especially those that are located close to coastlines where the continental shelf is narrow, would be the first targets for CO_2 removal and ocean sequestration. It is estimated that, by the year 2025, about 70% of the World's population will be living within 100km of the coastline (Hinrichsen 1989). As a result, energy generation capacity is likely to become more concentrated close to the ocean margins. A network of pipelines could be used to pipe the gas, collected from several point sources, either to port facilities for loading on large bulk carriers or for direct discharge at depths >1.5km in the deep ocean. Discharge depths, whether from surface vessels or pipeline, would need to be below the seasonal and permanent thermoclines. Otherwise upwelling processes will lead to carbon dioxide enriched water being rapidly mixed up into the upper ocean where it will vent the excess CO_2 into the atmosphere, thus losing any advantage from ocean disposal.

Carbon dioxide introduced into the deep water layers of the oceans will ultimately be transported back to the ocean surface where a neo-equilibrium will be established with the atmosphere. This will occur after about 500 years have passed, by which time other methods for CO_2 utilization or disposal must be available.

3.6.2. Description of Disposal Techniques

The various disposal options that have been proposed (Ohsumi 1996) are based on the known thermophysical properties of CO_2 (Fig. 3.6.3). At normal atmospheric temperatures and pressures CO_2 is a gas. However, at depths >500m where temperatures are almost always <11°C (though not in the Mediterranean and Red Seas) it condenses to its liquid phase. If the depth of discharge or storage is increased, so that the hydrostatic pressure increases, clathrates (a kind of crystal structure containing both carbon dioxide and water) begin to form (Uchida *et al.* 1995) at the interface between the liquid carbon dioxide and the surrounding water. Natural occurrences of clathrates have been observed *in situ* along the slope of the Izena Cauldron in the backarc basin of the Okinawa Trough (Honda *et al.* 1995). At still higher pressures, equivalent to depths of 3700m, liquid CO_2, being more compressible than water, becomes denser than the *in situ* sea-water. Then at pressures equivalent to depths >6000m and at the temperatures of 2-3°C typical of deep ocean environments, the liquid CO_2 freezes to form dry ice.

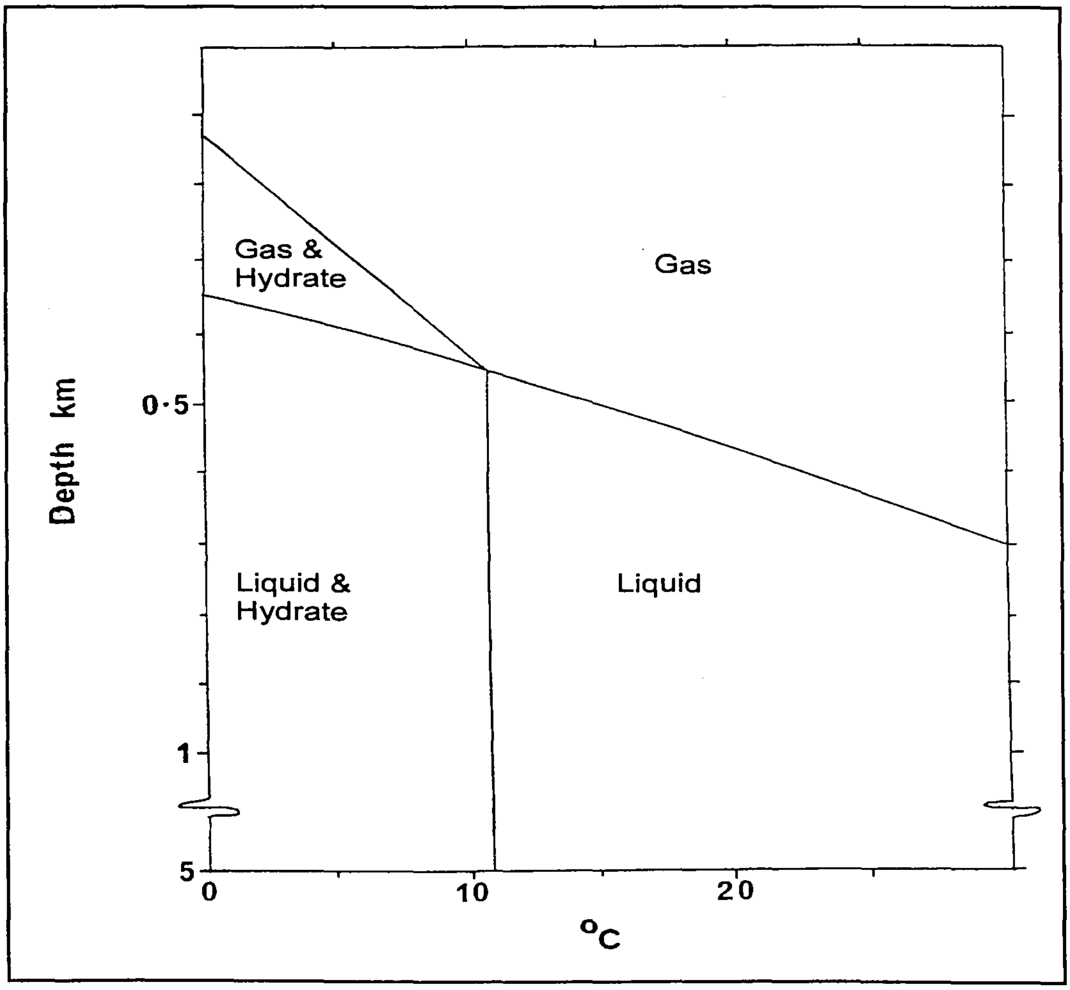

Figure 3.6.3 Phase diagram for carbon dioxide in water, with pressure expressed in terms of ocean depth (modified from Weaset and Selby 1967).

Figure 3.6.4 illustrates some of the proposed options for ocean disposal techniques. These can be divided into two main categories: those which aim

- for dissolution and dispersion within the deep water column

- to isolate the carbon dioxide on the sea-floor either as dry ice or as a lake of liquid carbon dioxide.

The first of these concepts can be considered to be accelerating the on-going uptake of excess CO_2 from the atmosphere, and if dissolution from the solid or liquid phases proceeds rapidly (i.e. the isolation is only partially effective) then the second alternative will function in a similar manner. Thus this approach to long-term storage of carbon dioxide will essentially speed up the eventual attainment of the long-term equilibrium between the atmospheric, oceanic and sediment reservoirs. If it becomes necessary to isolate much of the anthropogenic CO_2 from the normal carbon cycle, an option originally devised for the possible disposal of high level radioactive waste deep within oceanic sediments could be resurrected. The idea was to free-fall stream-lined penetrometers which would hit the sea-bed at a speed fast enough for them to penetrate tens of metres into the sediment. The sides of the holes they produce will collapse filling in the holes and sealing the waste beneath a layer of sediment. However, the cost for this method in terms of energy using today's technology, would be prohibitive.

Partial isolation can be achieved by discharging liquid CO_2 into topographical depressions in the sea-floor at depths >3700m, creating a "density lake". Dissolution from such a lake would be retarded by the formation of a coating of clathrate over its surface; also it would soon become overlain by a dense layer of sea-water saturated with dissolved CO_2. Original optimism that these layers would keep the carbon dioxide in place for hundreds of years, if not millennia, has been confounded by recent laboratory experiments which suggest that the dissolution would be relatively rapid (Hirai *et al.* 1997, Wong and Hirai 1997). Only where the depression is steep-walled would the CO_2 be contained for more than a few decades because, within a rather short time, dissolution across the upper surface of the lake would balance the rate of disposal. Thus, this approach of isolation in topographical depressions should be more appropriately termed "delayed dissolution". Venting back into the atmosphere will largely be avoided for long periods of time because of the great depths at which this technique of disposal would have to be operated. There would also be substantial medium-term differences in the environmental impact of the CO_2 saturated water being either widely dispersed throughout the deep ocean or largely localised within depressions.

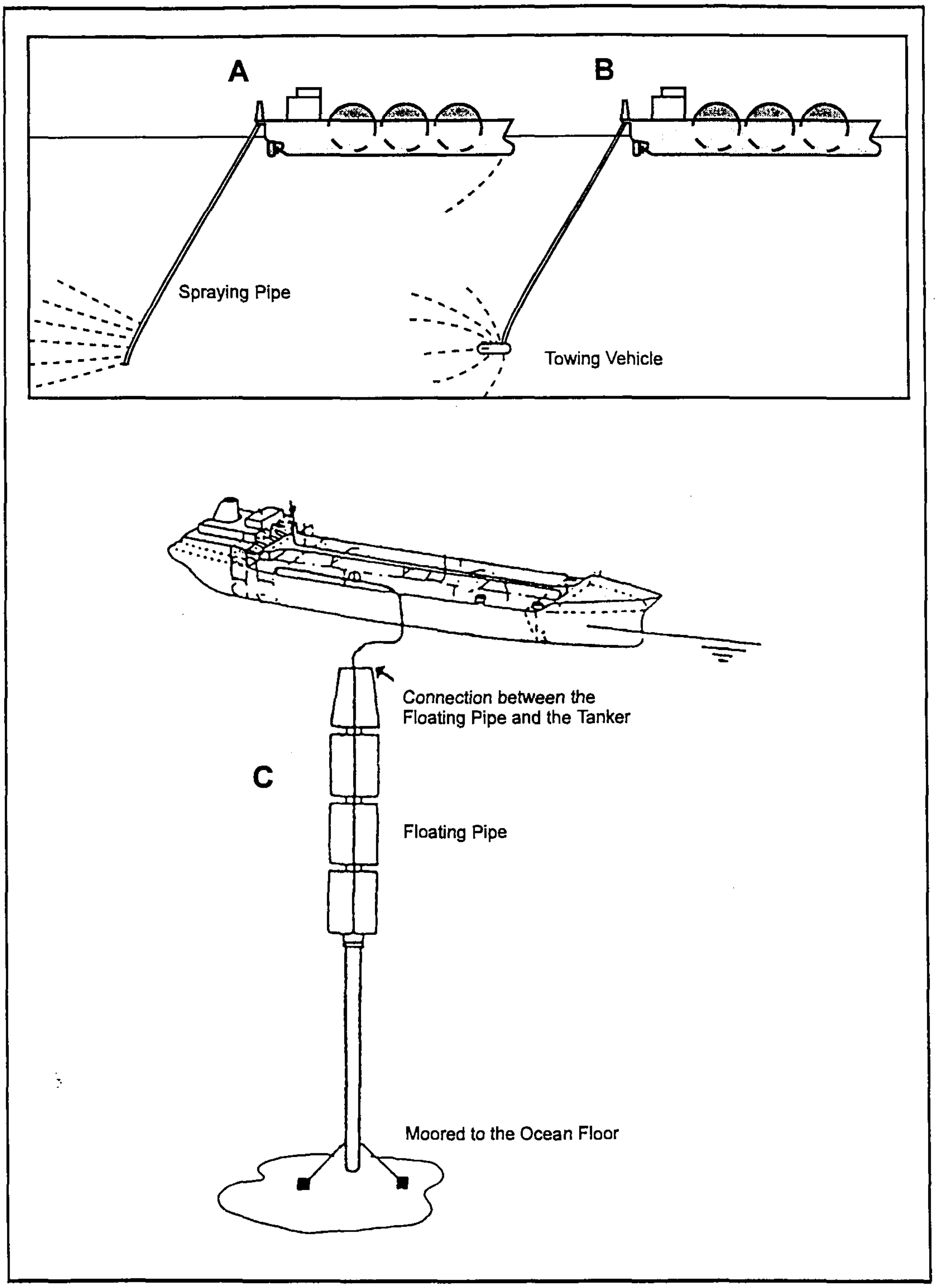

Figure 3.6.4 Possible carbon dioxide disposal options (modified from Nakashiki et al. 1995).

Disposal on the sea-floor at depths <3000m is expected to result in significant dissolution of CO_2. The options theoretically considered include dropping packages of dry ice and the production of dumps of dense, semi-stable hydrates. The area of sea-floor directly effected would be limited but spread of CO_2-saturated water (low pH) across the sea-floor would probably be extensive. While this would increase the dissolution of carbonate sediments and hence increase the total quantities that could be discharged, the impact on the benthic communities might be too high to be warranted.

Another approach is to maximise the rate of dissolution and dilution of the CO_2 as it is discharged at depths of 1 to 3km. Rising plumes of bubbles and/or droplets will entrain water which will dissolve the carbon dioxide and be removed into the ambient water mass (Fig. 3.6.5). If either the discharge depth is too shallow or the discharge technique is inadequate to ensure efficient dissolution some of the carbon dioxide may escape directly to the atmosphere. In the absence of hydrates, the carbon dioxide will dissolve rapidly and efficiently, but if a shell of clathrate forms around each rising droplet, dissolution may be impaired. In addition, if the droplets tend to coalesce then the vertical distance travelled before dissolution is completed will increase. Various designs of static bottom-mounted or anchored dissolution chambers have been proposed (Adams *et al.* 1995, Kajishima *et al.* 1995) which may be used to generate plumes of CO_2-enriched water either cascading along the sea-floor (Haugan and Drange 1992) or for open water dissolution (Haugan *et al.* 1995). Alternatively, disposal from a moving ship might be considered (Nakashiki *et al.* 1995) which would ensure greater efficiency in the dispersion of dissolved CO_2 being discharged from a trailing pipe, or by dropping blocks of dry ice.

3.6.3. Impacts of Disposal

The form of impact will depend on the mode of disposal adopted. The minimum impact will probably be caused by dropping dry ice penetrometers, which will leave a wake of dissolved CO_2 in the water column and create a hole in the sediments. However, the high energy penalty of converting enough carbon dioxide into dry ice to contribute effectively to the reduction of anthropogenic carbon dioxide inputs to the atmosphere probably rules out this approach. Creation of liquid lakes on the sea-floor will certainly cause the sterilisation of the area directly effected. The dispersion of CO_2-saturated water away from the upper surface of such a lake will also severely effect the neighbouring benthic and pelagic environments. High concentrations of CO_2 are likely to be highly toxic both because of the water having a lower pH and through direct physiological effects, e.g. by interfering with respiratory transport. Similarly options involving the dissolution and dispersion of CO_2 will have near-field effects within which virtually all living organisms will be killed, and far field effects where viability is reduced. Sensitivities should be species specific. Any estimation of the dimensions of the near- and far-fields given specific sizes of discharge remains fraught with uncertainty because of a chronic lack of information about basic physiological responses of relevant species.

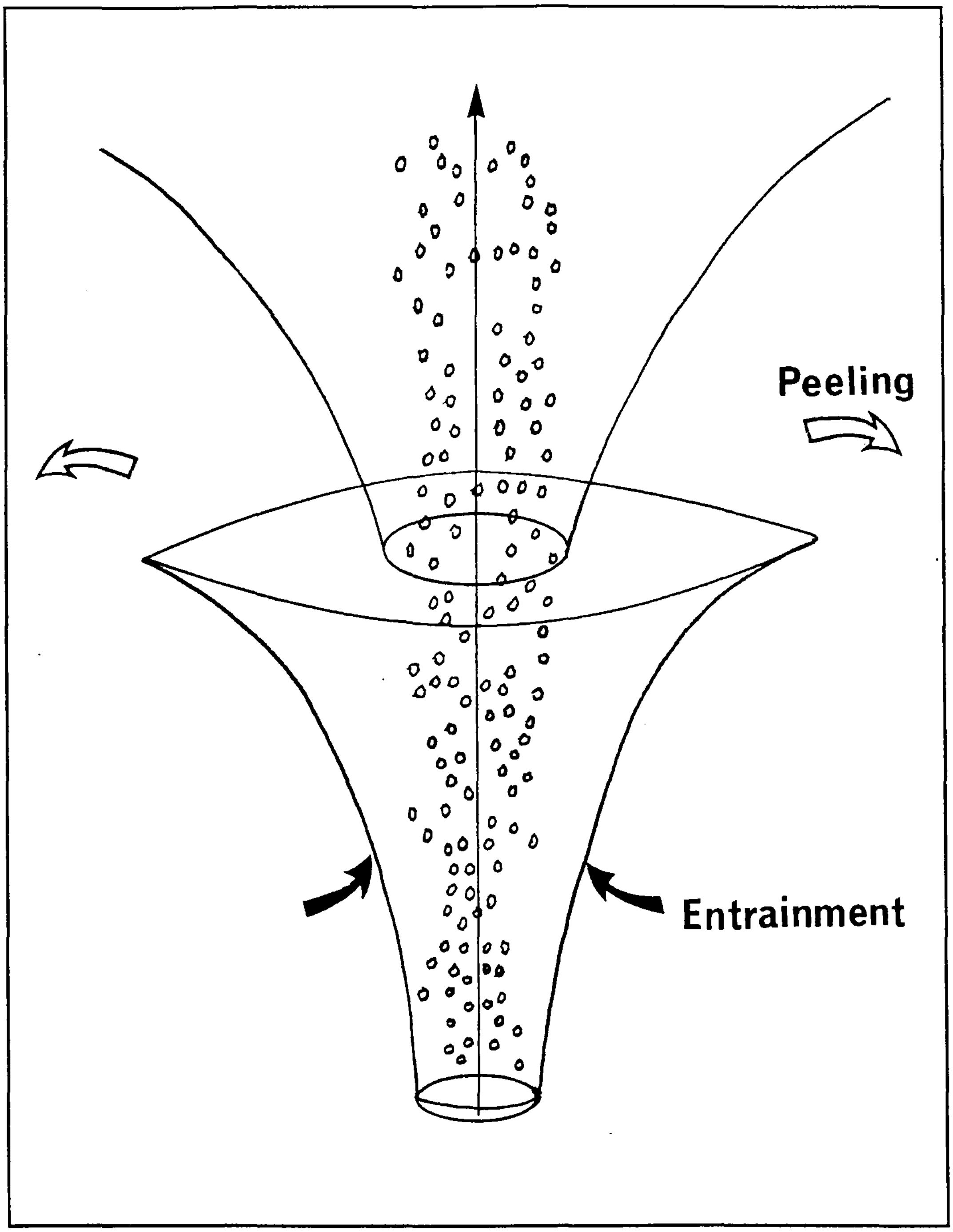

Figure 3.6.5 Conceptualisation of the dynamics of a droplet plume released from a stationary source at a depth of <3000m, showing how the buoyant plume will entrain water as it rises, and dense carbon dioxide-enriched water will peel off from the rising plume.

When CO_2 is added to sea-water the pH is reduced (i.e. acidified) with potentially detrimental effects on marine life (Magnesen and Wahl 1993). Since pre-industrial times there has been a reduction in pH of about 0.1 in ocean waters, and during the next century a further decrease of at least 0.2 to 0.3 is to be expected (Haugan and Drange 1996). These changes will occur uniformly over the entire ocean surface, so even if amelioration action is taken, marine life in the euphotic zone will be subjected to reduced pH.

In the oceans, values of pH typically range from 7.7 to 8.2 (Brewer *et al.* 1995). The highest natural values occur at the surface at high latitudes in summer, often associated with periods of high production when there is a draw down of CO_2 in the near-surface layers because of photosynthesis, and this results in an increase in the water's alkalinity. The lowest natural pH values occur at intermediate depths in "old" water (i.e. water that left the surface of the ocean several centuries ago) that has become enriched with inorganic carbon as a result of remineralisation of organic matter. Surface waters show natural variations in alkalinity with a seasonal amplitude of about 0.1 pH unit. In tropical seas pCO_2 is generally high, partly because the surface waters are warmer, but also because the global thermohaline circulation results in slow but widespread upwelling. Hence tropical oceans tend to be de-gassing CO_2 and to have somewhat lower pHs. Where there is more active coastal and open ocean upwelling, alkalinity is lowered (Lampitt *et al.* 1995). This effect is enhanced if blooms of coccolithophorids are stimulated, since the formation of calcium carbonate results in increases in pCO_2 and hence lowers pH (Robertson *et al.* 1994). Below the euphotic zone at depths of 100 - 200m seasonal variations in pH tend to be very small.

Releasing liquid CO_2 from a fixed point at a depth of about 1km, where currents are typically about $10cm \cdot s^{-1}$, would reduce the local pH by 2 - 3 units within the "dissolution" zone which would extend a few hundred metres vertically and a few tens of metres horizontally (Haugan *et al.* 1995, Haugan 1997). Since reductions of pH to <6.5 appear to be lethal to all marine organisms tested so far (Magnesen and Wahl 1993, Pörtner and Reipschläger 1996) those unable to avoid the plume would be killed. Mobile species would stand a good chance of escaping from the plume so long as their behavioural responses did not entrap them (e.g. a species which has an escape response to swim up or down once it enters such a plume may not escape from it before being overwhelmed). Beyond the dissolution zone there would be a diffusion regime wherein turbulent mixing associated with advection along isopycnals disperses the CO_2 and pH levels would increase.

The near-field effects will be defined by the levels of alkalinity at which long-term organism survival is not threatened. Theoretical studies have used a pH of 7.0 as the boundary of the near-field in modelling (Stegen *et al.* 1995, Auerbach *et al.* 1996). There is a need for far more extensive physiological studies to evaluate this choice, but the model outputs provide an indication of the extent of the impact. They suggest that if the CO_2 from a single $500MW_e$ plant is discharged as a droplet plume (i.e. dropping liquid CO_2 unconfined at a depth of 1 to 1.5km), its near-field effects will spread through 1.42km^3 of water which will extend 19.8km "down-stream" from the discharge point. The combined discharge from 10 such plants would create near-field effects in 100km^3 of water extending 138km from the source. Since a single 500MW$_e$ plant will generate annually about 3.6 million tonnes of CO_2, to sequester 1Gt (about a third of the total emissions from all sources) of carbon in the deep ocean in this manner would require the CO_2 output of 1000 such plants to be discharged at depth. Near-field effects may then extend through 10^{6}km^3. While this is an immense volume it is still less than 0.1% of the total volume of the oceans.

Release at the sea-floor on a continental slope will result in the formation of a dense plume which will flow downslope as a gravity current, carrying the CO_2 to much greater depths. This will have severe downstream impacts on the benthic communities. Particular care would need to be taken to ensure that this would not threaten any species with extinction or interfere with living resource exploitation. Moreover, since the plume will remain in close contact with the carbonate deposits on the slope it can be expected to erode a channel, thus limiting its lateral spread. On the negative side the erosion of such a channel could ultimately threaten the stability of the slope. Intermittently throughout the geological record there is evidence of massive underwater "landslides" having occurred, usually at times when sea-levels have been changing rapidly during the change from glacial to interglacial conditions (and *vice versa*), often involving hundreds of km^3 sediments. These must have had far-reaching impact across the ocean basins not only on the sea-bed but also in shelf-sea areas by creating huge tsunami-like waves. On the positive side more CO_2 would be carried into very deep water and more would be neutralised by the dissolution of carbonate, and hence a greater amount of CO_2 could be sequestered in deep ocean water masses.

The major environmental effects are expected to be associated with lowering of the pH in the water column, its effects on the fauna and the erosion of carbonate deposits on the sea-floor. There may also be sub-lethal physiological impacts associated with exchanges and transport of respiratory gases, and there have been some preliminary experimental indications that potassium metabolism may be particularly sensitive to enhancement of carbon dioxide concentrations (Lockwood, pers. comm.). Such effects are likely to be expressed in nature by shifts in competitive balances between species leading to changes in community structure and dynamics, and hence to changes in ecological processes.

However, it must constantly be born in mind that if emissions to the atmosphere continue at present levels (or higher) then reductions in the pH of the upper waters of the global ocean are likely to begin to interfere with upper ocean processes and may even begin to approach the physiological limits of tolerance for key marine biota. Increasing CO_2 concentrations in the atmosphere has substantial effects on terrestrial plant ecosystems, and similar effects can be expected in oceanic communities. These effects may be beneficial if they accelerate the "biological-pump", the mechanism whereby biological processes remove organic carbon to the deep ocean, particularly if the suggestion that primary production can be carbon, rather than nutrient, limited is confirmed for large areas of the ocean (Riebesell *et al.* 1993). On the other hand, increases in carbon dioxide which lead to decreases of pH in the euphotic zone will suppress primary production by some and ultimately all species. Since global patterns in the seasonality of primary production (Longhurst 1995) match with the classical biogeographical boundaries in pelagic, and possibly also benthic, communities in the deep ocean (Angel in press), shifts in the productivity of the upper ocean may have very far reaching effects on the total ocean. Further evaluations of options available for the management of CO_2 discharge, including deep ocean disposal, will need improved assessments of what might be the large-scale implications of taking no amelioration action. If reductions in CO_2 generation cannot be achieved, in a holistic global scenario it may be better to impose greater stresses on deep ocean ecosystems than to allow too much deterioration of upper ocean and terrestrial ecosystems.

The range of natural variability of pH at depths >1km is much smaller than in the euphotic zone. So it is to be expected that deep ecosystems will show lower tolerance to perturbations in CO_2 concentrations than shallow ecosystems. On the other hand, it is in the euphotic zone where most primary production occurs, especially the part that is finally important to commercial fisheries.

One process whereby the deep water is linked to the upper water column is through vertical migrations. Diel vertical migrations by macroplankton and micronekton are mostly restricted to the upper 1km, but in the centre of subtropical gyres some species of lantern fish regularly migrate to depths of 1.5km and some decapod crustaceans to 1.2km (Angel 1989). Many species undertake ontogenetic migrations in which the normal pattern is for the early stages to be spent in the upper water column and the mature stages at greater depths. In *Euphausia* spp. the eggs are heavier than sea-water and sink deep into the water before hatching. The larvae, which are initially non-feeding, swim back towards the surface arriving as they reach the first feeding stage. There are also seasonal migrations, especially at high latitudes, whereby the over-wintering stocks of some species reside in a state of diapause (a kind of hibernation) in deep water, and return to shallower depths in spring. Any sterilisation of deep water layers through the introduction of CO_2 might disrupt these migrations and have substantial effects on the ecology of the local region including the surface layers.

3.6.4. Research Required to Evaluate Impacts

Experiments at different scales are needed in order to assess whether the major environmental effects and concerns, the lowering of the pH, its effect on the fauna and the erosion of carbonate deposits on the seafloor, have any substance. Small scale physiological and ecological experiments in laboratory conditions or in mesocosms and *in situ* benthic chambers (for benthos) will provide useful and important results on the reaction of selected key species and organism assemblages to increased CO_2 levels. There is an urgent need to define the bounds for lethal and sublethal effects on organism viability, growth and reproductive capacity for a range of relevant species. These laboratory experiments may not allow the extrapolation of effects to community levels, so that whole system experiments in which the responses of a total ecosystem can be studied, may be required. Experiments in the open ocean in which quantities of carbon dioxide are injected into deep water at depths of ca. 1km would be faced with substantial logistical problems in tracking the plume and monitoring the changes. Restricting or containing a CO_2 plume within a natural system would be the essential approach, while still being able to conduct an experiment that has space and time scales which can credibly be extrapolated to full scales of CO_2 deep ocean disposal.

Plans are presently being discussed for an experiment in which carbon dioxide is to be injected into the deep water of a fjord in Norwegian or other nation's territorial waters. Since some fjords are sufficiently deep to have communities that are quite similar in composition and ecological function to the open ocean, they can be used as "model" oceanic ecosystems. The logistics of conducting such an experiment could realistically be achieved. This should enable the results to be extrapolated with some measure of confidence from one system to the other. Injection of relatively small quantities (of order 100 - 1000 tonnes) of carbon dioxide could be used to lower pH in the deep water sufficiently to mimic the conditions likely to prevail in a disposal plume. The morphology and hydrology of many fjords, having a shallow sill at the entrance and relatively limited exchange of water below sill depths, would allow experimental designs to be developed that would follow the induced changes and monitor the rate of recovery of the system to its pre-experimental condition. The results of such an experiment would also provide insights into what sorts of environmental responses may occur in the oceans if no limitation of atmospheric emissions of CO_2 is achieved.

The selected fjord would need to be extensively surveyed to establish its pre-experimental baseline characteristics before any experimental inflow and the build-up of a CO_2-enriched deep water mass. The experiment would best be carried out during summer when there is some thermohaline stratification, which would restrict impacts mostly to the deeper layers and any enhancement of CO_2 concentrations in the upper waters would rapidly be compensated by venting to the atmosphere. Any reduction of the deep water communities is likely to be temporary. The primary reactions of the key species in the plankton and the benthos systems need to be observed, and changes of community structures must be registered. It is to be expected that the CO_2 level in the deep water will successively diminish, and that its concentration as well as species distribution and abundance would need to be monitored at least for a period of two years. Ventilation of the fjord with normal sea-water (and freshwater inputs) will ensure that the CO_2-enriched water is flushed out after some time. An analogy might be drawn with the impact of the intermittent formation of anoxic conditions in the Baltic as a result of the limited deep water exchanges and eutrophication. There benthic communities exterminated by the anoxic conditions begin to recover through re-invasion by pelagic larvae as soon as the de-oxygenated water is flushed out and replaced. Restoration of the pelagic fjord system impacted by the experiment will occur through the advection back into the fjord of larvae and adults of plankton and nekton species. The only pelagic species that might be delayed in their return will be any which at the time of the experiment are at the deep phase of a seasonal or ontogenetic migration and so may not re-enter across the shallow sill until they migrate back up through the water column. Thus pelagic communities can be expected to be back to normal within a year.

The benthic system will regenerate through the transport of larvae into the fjord. Only some of the deepest-living benthic species which either do not have pelagic larvae, or have larvae with very brief pelagic phases, or do not regularly move shallow enough to re-enter over the sill, may not re-colonise rapidly. But even if the deepest-living communities may remain relatively de-pauperate in their species composition for a few years, in time they will be restored. Another analogous situation is the aftermath of major oil-spills where re-invasion of benthic and littoral species re-establishes similar (if not identical) communities within a few years.

Generally, such large-scale CO_2 disposal fjord experiment requires a preparatory phase and a long-term observational and monitoring phase. This second part of the experiment will demand high sampling and observation efforts following the experimental impact, but decreasing activities during subsequent years.

4. The Acceptability of Large-scale Ecological Research in the Deep Sea

4.1. The Changing Scale of Deep-sea Research

Traditionally, research in the deep sea concerned itself with descriptions of the natural ecosystem and an understanding of how it works. Physical and chemical oceanographers characterised the sea water masses by taking samples and analysing their properties. Samples usually amounted to no more than a few litres, though sample volumes of several hundreds of litres became necessary to measure low concentrations of contaminants. Biologists employed similar water bottles, but also sampled the water column and the bottom with a variety of nets equipped with various mesh sizes. Large plankton nets used in this way may filter volumes of several tens of thousands of cubic metres, while dredges and bottom trawls sample areas of the sea floor up to about 25,000m^2. Finally, benthic biological, geological and geochemical studies have generally used corers and grabs, each sampling areas of no more than a quarter of a square metre. Usually, sampling stations employing these techniques have in the past been rather widely spaced, although the high local variability of physico-chemical properties and faunal abundance lead to repeated sampling at the same locality. These methods, more or less unchanged, have been employed for more than 120 years, from pre-*Challenger* days until very recently. Technical developments improved the operation and monitoring of the equipment, but basically the same types of tools were employed and the impacts associated with oceanographic research remained in the same order of magnitude throughout this time span. Impacts associated with these activities are considered to be small-scale. The results led to a better understanding of the ecosystem and its subunits.

In considering the potential impacts on the deep sea of resource extraction or waste deposition, the same small scale techniques were used initially. However, it soon became apparent that extrapolation from the resulting data to full-scale commercial mining or waste disposal could not be validly achieved on the basis of such traditional small-scale research. For example, within the DOMES and the MESEDA projects, environmental studies were conducted in association with the PPMTs (Thiel *et al.* 1991), but the results could not be used to predict potential effects of commercial-scale mining operations.

A new approach, the artificial disturbance experiment, was therefore proposed in which the areal coverage and impact were planned to be much greater than those investigated in traditional research. Within the DISCOL disturbance and recolonization experiment, for example, 11km^2 of the deep-sea floor have been impacted, while a number of different ooze suspension and resedimentation experiments (e.g. the US-BIE: Benthic Impact Experiment; JET: Japan Deep Sea Impact Experiment; IOM-BIE: IOM Benthic Impact Experiment) have directly impacted about 0.5 to 1km^2 (Trueblood 1993, Barnett and Yamauchi 1995, Kotlinski 1995). It is within this context of changing scales of deep-sea research that the impacts of oceanographic research on the oceanic ecosystem must be evaluated.

4.2. The Deep Sea as a Pristine Environment?

Those who object in principle to any human impact on the deep sea often argue that this is a pristine environment so far unaffected by man's activities. Any deep-sea benthic biologist can testify that this is not so. It is almost impossible to drag a trawl or dredge across the ocean floor without retrieving evidence of man's activities, ranging from about one hundred years' accumulation of clinker cleared out of the boilers of coal burning steamships to the beer and soda cans, wine bottles, plastic cocktail sticks, fishing line and rope and similar assortment of materials of a more recent era (e.g. Thiel 1972, Galil *et al.* 1995). Despite their ubiquity, these artefacts are relatively harmless. Much more insidious is the fact that man-made chemicals such as DDT and PCBs are detectable almost everywhere in the sea, albeit in low concentrations. Anthropogenic persistent compounds, e.g. chlorofluorocarbons (CFCs), characterise specific water masses and are used as oceanographic tracers (Schlitzer 1991, Rhein *et al.* 1995), and radioactive fallout may arrive in the deep sea soon after its concentration by phytoplankton in ocean surface water layers (Erlenkeuser and Balzer 1988). The waters of the open ocean have already absorbed about half of the anthropogenic CO_2 emitted above natural levels into the atmosphere since the start of the Industrial Revolution. Furthermore, the deep sea floor is littered with shipwrecks. Over 7 million tonnes of British merchant vessels were lost during the First World War (Hurd 1929), while over 21 million tonnes of Allied merchant shipping were sunk during the Second World War (Roskill 1961). In addition, there were substantial losses of Axis merchant shipping and of warships on both sides during both wars. This form of impact continues, of course, even in peacetime though a higher proportion occurs in shallow inshore waters. Between 1973 and 1995, some 29.3 million tonnes of shipping was lost, an average of 1.3 million tonnes each year (Institute of London Underwriters, 1995). Many of these vessels contained potentially harmful cargoes, ranging from hydrocarbons to munitions and toxic chemicals.

While the fact that the deep ocean has already been affected by man's activities is no argument for further impact, it is significant that no widespread deleterious effect has ever been attributed to any of these events. However, none of these activities has been sufficiently investigated.

But apart from these anthropogenic impacts, the deep ocean is not the tranquil and monotonous environment it was thought to be only a few decades ago. Large regions of the sea floor are from time to time subjected to massive physical disturbance from turbidity flows, while smaller areas are frequently affected by benthic storms. Still smaller portions, such as those around hydrocarbon seeps and hydrothermal vents, experience chemical emissions that in any other context would be considered to be extreme pollution. Given these characteristics, is it possible to assess the likely environmental impacts of research at the scales which we have identified in the body of this report? Would these impacts be acceptable?

4.3. Criteria for Acceptability

All human activity has environmental impact, but for many of our activities the scale of the impact in terms of both time and space is either too small or too short to cause concern. However, as the scale of the activity increases, so does the scale of the impact (Fig. 1.1). Eventually this starts to cause some concern as the bounds of acceptability are approached. In the absence of clear scientific criteria, the assessment of acceptability is highly subjective, and so clarifying this through more objective assessment is a highly desirable goal. In 1972, the United Nations Conference on the Human Environment set out 26 general principles from which useful guide-lines can be derived. Four are of particular relevance in this context:

- Development must be undertaken in a manner that avoids prejudicing environmental amenities for future generations.

- There must be avoidance of serious, irreversible damage to the environment.

- There must be avoidance of measures that transfer damage from one biome to another (i.e. terrestrial environments to marine or atmospheric environments).

- There must be concerted international action for environmental protection and preservation.

We feel that the principle consideration must be the maintenance of global ecosystems. No development should be allowed that disrupts global environmental processes, either individually or through synergistic interaction with other activities. The conservation of natural systems and the preservation of ecological processes are essential to the maintenance of human health and well-being. Criteria are often presented with a strong anthropocentric bias. While socio-economic evaluations should be taken into account, they should play a secondary rather than a primary role. Emphasis should be placed on the sustainability of the global system and its subunits and less placed on the ceaseless quest for improved living standards. The well-being of the world population, after all, is dependent upon the existence of a healthy global ecosystem. We further believe that assessment of the impacts of new developments should also be evaluated within the context of the impacts of customary activities, so that evaluations can balance the environmental gains and losses of adopting the new development. Experimental procedures will often play a key role in these evaluations.

What criteria can then be used for assessing the acceptability of large-scale experimental programmes? First and foremost, there must be a *prima facie* case that the eventual industrial activity will, on balance, prove to be beneficial to society. Even if this condition is met, large-scale experiments will only be acceptable if:

- The objectives of the experiment can reasonably be expected to lead to the demonstration of the environmental safety (acceptability) of the full-scale industrial activity.

- The impacts must be monitorable, otherwise it will not be possible to evaluate convincingly the acceptability of the development at full scale.

- There must be no persistent adverse effects on biodiversity at regional scales, although some localised impacts may be unavoidable. This means that no species should be driven to extinction, the introduction of exotic species must be avoided, and the area (or volume) inhabited by any community should not be reduced by some arbitrarily chosen figure, say >0.1%.

- The experiment must not impair any ecological processes, particularly those influencing climate and the dynamics of material fluxes, especially of those substances considered to be toxic.

- No living resource, whether potential or currently being exploited, must be adversely effected.

- The experiment must not significantly disrupt other uses of the oceans (for transport, recreation, security, extraction of energy or the exploitation of other non-living resources).

- There must be no critical or mass transport pathways via which significant quantities of potentially deleterious substances and organisms (pathogens, heavy metals, radioactive substances and toxic substances including synthetic organics which tend to bioaccumulate) can impair the functioning of keystone communities and/or processes, or may even be transported back to man.

- The risks to human safety, health and well-being must not be significantly increased by adopting the deep-ocean option rather than alternative options presently considered to be acceptable.

Due care must be taken with site selection for experiments. This can not be based solely on scientific considerations, since operational limitations and legal constraints will be unavoidable. For example, at present a large-scale experiment to evaluate the disposal of waste materials would probably require modification of international law. Selection should be based on the best scientific evidence available, but will inevitably involve some intuitive elements when baseline data are inadequate. Indeed it may be preferable to avoid interfering with those few sites where adequate baseline data already exist because these may be the only sites where early detection of climatic change in the deep ocean will be possible. However, such sites may be used as experimental controls.

Valent and Young (1995) presented a site selection protocol based on a concept of geographic overlays of exclusionary and variable, weighted attributes as a means of identifying sites where the conditions best match the predefined criteria. The system has the advantage that along the range of variability of a certain factor the weighting can be varied, so that, for example, where sedimentation rates are low, variations in these rates can be given a lower weighting, but where they are high the weighting can be increased. Thus, as knowledge of the deep ocean improves, the selection procedure can be quickly updated. It also means that the sensitivity of the selection procedure can be tuned to take account of synergies between factors. Exclusions adopted by Valent and Young (1995) in evaluating possible sites for waste disposal experiments included:

- Areas with bathymetric depths <3000m.
- Areas within the EEZ's of other nations.
- Areas more than 1800km from possible loading ports (an exclusion based on minimising operational costs).
- Areas of special sea-bed features (seamounts, open ocean banks, trenches, hydrothermal vents).

They also considered a number of environmental characteristics that would reduce the likelihood of a site being selected which included:

- Areas of rugged topography and steep slopes.
- Areas of high tectonic and seismic activity.
- Areas where ocean eddies are a regular feature.
- Areas where there are frequent benthic storms and/or strong near-bottom currents.
- Areas close to cable routes.
- Areas with busy shipping lanes.
- Areas of high biological productivity.
- Areas with a high incidence of storms and severe wave conditions.

For many of these factors, charts and climatological maps already exist or could be compiled. For others, such as benthic biomass, empirical mathematical models relating factors such as benthic biomass and benthic respiration to depth and surface productivity may be used (e.g. Rowe and Pariente 1992). These emperical methods my have to be refined further by global climatological studies based on the seasonality and total mean primary productivity, factors which through variations in export production influence biological activity and also identify the mean locations of major oceanographic fronts. Site selection must be followed by site characterisation to provide a proper baseline against which any impacts can be assessed. Carrying out these procedures in themselves will be basic research and will not carry significant environmental risk.

4.4. Proposed Research

Because of the scales of the expected commercial mining operations reviewed in Chapter 2, much of the environmental research required to make appropriate risk assessments will also have to be conducted at the large-scale experimental level. Although monitoring the expected PMO activities will be conducted primarily with conventional methods, new techniques and concepts will be required to address some of the outstanding questions related to industrial penetration of the oceans.

The situation with waste disposal is somewhat different. In the case of munitions, radioactive wastes and sewage sludge, relatively large-scaled dumping has already occurred at various localities in the deep sea. In many cases, information on the timing of the disposal, the nature and packaging of the materials and the quantities disposed of are available (although not necessarily always a matter of public record). These disposal actions (accidental or intentional) provide a unique and ready-made laboratory for detailed scientific investigation. Some of the existing disposal sites should be fully utilized and appropriate studies conducted to assess environmental impacts on a *post facto* basis. The information obtained would be invaluable in making risk assessments if the oceans again come under consideration for such disposal activities or, indeed, for the disposal of any large structures.

Disposal of other waste materials such as dredge spoils, large offshore structures and carbon dioxide will again require large-scale experimental approaches to make meaningful assessments of environmental impacts. Much as the nacent ocean mining industry is planning to conduct PMOs which would be accompanied by environmental monitoring programs, waste disposal should also be initially conducted under a pilot program which we herein term a pilot disposal operation (PDO). Such operations should also be monitored and investigated before the oceans are opened to wide-spread and high volume disposal activities. Experimentation at large-scales may also be necessary to answer specific questions that can not be addressed through simple monitoring using conventional techniques.

4.5. Acceptability of the Proposed Research

The scales of traditional research activities fall well short of the limits of acceptability discussed above. So all the research proposed in the section on mining and waste disposal that involve laboratory and mesocosm experiments and monitoring of previous activities (i.e. dumps of munitions and radioactive waste, and shipwrecks) are deemed to be acceptable. The question of acceptability should then address the larger-scale experiments. These can be categorised into purely scientific experiments and those that will be conducted in conjunction with commercial testing and development. In mineral exploitation the concept of a Pilot Mining Operation (PMO) is now well established in the literature. We suggest that an analogous concept of a Pilot Disposal Operation (PDO) be developed for waste disposal, in which large-scale scientific experiments necessary to evaluate the environmental impacts of a full-scale industrial programme would have to be conducted in conjunction with industrial evaluations of the technical and commercial feasibility.

Since the aim of these experiments is to explore the extent of the eventual impacts, the criteria of acceptability for the experiments suggested above are precautionary. Thus the requirement that <0.1% of any particular habitat should be effected during the experiment may need to be relaxed if the assessments conclude that commercial exploitation is viable and environmentally acceptable. The full extent of the impacts of the commercial activity will need further consideration, but may be much larger than the limit suggested. For example, at present in the North Sea, vast areas of the sea-bed are mechanically disturbed on an annual basis by trawling. In addition, extensive areas are affected by other forms of exploitation such as aggregate extraction, discharges of drilling fluids from hydrocarbon installations, disposal of dredge spoils, and cable and pipeline routes. It should be noted, however, that these activities are being conducted in relatively shoaler waters and not in the deep sea where ecosystems may respond quite differently and have slower recovery periods.

A key unknown in the deep ocean is the rate at which the communities recover after disturbance. This is certainly much slower than in shallow seas. If recovery takes 100 years, the use (disturbance) of 0.1% per annum, will accumulate to up to a maximum of 10% of the area within this period showing effects of the disturbance, yet by these criteria would still be acceptable. Industrial activity may well change the characteristics of the seabed permanently. For example, manganese nodule mining will remove many of the hard substrata converting the seabed into a predominantly soft-sediment environment. This may not prove to be a major concern since large areas within the nodule field will remain unexploited, so even if the incidence of hard substrata has been reduced they will not be eliminated.

PMOs for manganese nodule mining will probably involve extraction of nodules from 1 - 5km^2 per day. Thus 30 days of pilot mining operations will affect from 30 to 150km^2. This is still far less than 0.1% of the total area of a typical mining concession (about 150,000km^2) which are themselves a subset of the oceanwide nodule fields. PDOs for the disposal of sewage sludge and dredge spoils should similarly be expected to demonstrate an ability to comply with the acceptability criteria through previous small-scale experiments and technical trials. Acceptability of a PDO for carbon dioxide is more problematic because there are many more unknown factors at present. A fjord experiment has been suggested as a practical means of containing the discharges within its basin so that their impacts can be tracked in time and space. The overall local impact on the individual fjord ecosystem is likely to exceed greatly the areal limits of acceptability. However, ecosystem recovery in fjords can be expected to be rapid via advection inflows replacing adults and larvae of species that have been exterminated locally during the experiment. Also the ecosystem of the fjord selected would be replicated in many other localities so that the experimental approach suggested should be acceptable.

Taking the environmental criteria into account, and judging from present knowledge and acceptability assessments, the large-scale experiments needed to evaluate full industrial development of the mining and waste disposal potential of the deep ocean can be conducted without unacceptable environmental risks. The main question asked in this desk study, whether large-scale experiments such as DISCOL and the BIEs, or the PMO/PDO activities, are acceptable or not can thus be answered positively. They are acceptable under the criteria selected. Since these criteria were chosen within precautionary limits, the experimental and pilot operation scale impacts can be conducted in good conscience.

It is important to note, however, that other groups might apply quite different criteria, either more or less stringent than those employed in the report. Their conclusions might therefore differ signigicantly from ours. Clearly, this is an area which would benefit from further discussions at a variety of levels.

However, no matter what criteria are eventually agreed, the authors of this report believe that large-scale experiments will be essential before commercial scale deep-sea mining or deep-sea disposal of sewage sludge, dredge spoil or carbon dioxide can be contemplated. Finally, such experiments must be conducted sufficiently early to ensure that adequate data are available to guide, and possibly restrain, a potentially rapid proliferation of commercial development in the deep ocean.

5. Literature

Abdullah, M.I., Z. Shiyu and K. Mosgren 1995. Arsenic and selenium species in the oxic and anoxic waters of the Oslofjord, Norway. *Marine Pollution Bulletin*, **31**: 116-126.

Abu Gideiri, Y.B. 1984a. Implications of sea mining for the Red Sea environment. *Hydrobiologia*, **110**: 15-19.

Abu Gideiri, Y.B. 1984b. Impacts of mining on central Red Sea environment. *Deep-Sea Research*, **31A**(6-8): 823-828.

ACOPS 1988. *ACOPS Yearbook 1986/7*. Advisory Committee of Pollution of the Sea, London 1988.

Adams, E.E., D. Golumb, X.Y. Zhang and H.J. Herzog 1995. Confined release of CO_2 into shallow seawater. pp 153-164 In: N. Handa and T. Ohsumi (eds) *Direct Ocean Disposal of Carbon Dioxide*, Terra Scientific Publishing Co, Tokyo.

Amann, H. 1983. The Atlantis II Deep Project in the Red Sea as a source of technology for the development of marine polymetallic sulphides. *Proc., OCEANS '83*, San Francisco, Aug. 29 - Sept. 1, 1983, Vol. **2**: 802-810.

Amann, H. 1985. Development of ocean mining in the Red Sea. *Marine Mining*, **5**(2): 103-116.

Amann, H. 1989. The Red Sea pilot project: lessons for future ocean mining. *Marine Mining*, **8**(1): 1-22.

Amann, H. 1990. Outlook for technical advances and environmental requirements in hard mineral mining on the deep-sea floor. *Materials and Society*, **14**(3/4): 387-396.

Amann, H., H. Bäcker and E. Blissenbach 1973. Metalliferous muds of the marine environment. *Proc. Offshore Technol. Conf.*, Houston, Texas, 29 April - 2 May 1973, Vol. **1** (OTC Paper 1759): 345-358.

Ames, L.L. 1959. The genesis of carbonate apatite. *Economic Geology*, **54**: 829-841.

Amos, A.F. and O.A. Roels 1977. Environmental aspects of manganese nodule mining. *Marine Policy*, **1**(2): 156-163.

Amos, A.F., C. Garside, K.C. Haines and O.A. Roels 1972. Effects of surface-discharged deep sea mining effluent. *MTS Journal*, **6**(4): 40-45.

Amos, A.F., C. Garside, R.D. Gerard, S. Levitus, T.C. Malone, A.Z. Paul and O.A. Roels 1973. Study of the impact of manganese nodule mining on the seabed and water column. pp. 221-264 In: *Inter-University Program of Research on Ferromanganese Deposits of the Ocean Floor, Phase I Report*. Seabed Assessment Program, IDOE/NSF, Washington, DC. 358 pp.

Anderson B.S. and J.W. Hunt 1988. Bioassay methods for evaluating the toxicity of heavy metals, biocides and sewage sludge using microscopic stages of giant kelp *Macrocystsis pyrifera* (Agardh): a preliminary report. *Marine Environmental Research*, **26**: 113-134.

Angel, M.V. 1989. Does mesopelagic biology affect the vertical flux? pp. 155-173 In: W.H. Berger, V.S.Smetacek and G. Wefer (eds) *Productivity of the Oceans: Past and present.* John Wiley.

Angel, M.V. in press. Importance of biogeography in global change and biodiversity studies. *Proceedings of ICOPB 2 Symposium*, July 1995.

Angel, M.V., P. Hargreaves, P. Kirkpatrick and P. Domanski 1982. Low variability in planktonic and micronektonic populations at 1000m depth in the vicinity of 42°N 17°W; Evidence against diel vertical migratory behaviour in the majority of species. *Biological Oceanography*, **1**: 287-319.

Anonymous 1995. Dumped munitions resurface. *Marine Pollution Bulletin*, **30**: 768.

Auerbach, D., J. Caulfield, H. Herzog and E. Adams 1996. pp 41-55 In: B. Ormerod and M. Angel (eds) *Ocean Storage of CO_2: Workshop 2. Environmental Impact.* IEA Greenhouse Gas R&D Programme, Cheltenham.

Bäcker, H. 1980. Erzschlämme. *Geologisches Jahrbuch*, **D38**: 77-108.

Baker, K.B. and W.C. Burnett 1988. Distribution, texture and composition of modern phosphate pellets in Peru shelf muds. *Marine Geology*, **80**: 195-213.

Ballard, R.D. 1984. Exploring the East Pacific Rise. *Oceanus*, **27**(3): 7-14.

Ballard, R.D. 1987. The discovery of the Titanic, Hodder and Stoughton, London, 232pp.

Ballard, R.D. and J. Francheteau 1982. The relationship between active sulfide deposition and the axial processes of the Mid-Ocean Ridge. *MTS Journal*, **16**(3): 8-22.

Barnett, B. and H. Yamauchi 1995. Deep sea sediment resuspension system used for the Japan Deep Sea Impact Experiment. *Proc. ISOPE Ocean Mining Symposium*, Tsukuba, Japan, Nov. 21-22, 1995: 175-179.

Bath, A.R. 1989. Deep sea mining technology: recent developments and future projects. *Proc. Offshore Technol. Conf.*, Houston, Texas, Vol. **2** (OTC Paper 5998): 333-340.

Batiza, R. 1985. Qualitative assessment of ploymetallic sulfide mineral deposits on seamounts. *Marine Mining,* **5**(2): 181-190.

Baturin, G.N. 1982. Phosphorites on the sea floor. *Elsevier*, **343**.

Baturin, G.N., K.I. Merkulova and P.I. Chalov 1972. Radiometric evidence for recent formation of phosphatic nodules in marine shelf sediments. *Marine Geology*, **13**: M34-M41.

Baturin, G.N. and P.L. Bezrukov 1979. Phosphorites - the unsolved problems. In: Y.K. Bentor, Marine phosphorites - geochemistry, occurrence, genesis. Tulsa, *SEPM Spec. Publ.*, **29**: 3-18.

Beauchamp, R.G. 1984. Geological perspectives of metalliferous sulfides: offshore exploration in the Gorda Ridge area. *Proc. Offshore Technol. Conf.*, Vol. **3** (OTC Paper 4778): 17-24.

Bechtel Corporation 1996. *Deep-sea nodule mining Prefeasibility Study (Executive Summary).* Bechtel Corp. Mining and Metals, San Francisco, CA (July 1996). 24 pp.

Bentor, Y.K. 1980. Phosphorites - the unsolved problems. In: Y.K. Benthor, Marine phosphorites - geochemistry, occurrrence, genesis. Tulsa, *SEPM Spec. Publ.*, **29**: 3-18.

Bignell, R.D. 1978. Genesis of the Red Sea: metalliferous sediments. *Marine Mining*, **1**(3): 209-235.

Billett, D.S.M., R.S. Lampitt, A.L. Rice and R.F.C. Mantoura 1983. Seasonal sedimentation of phytoplankton to the deep-sea benthos. *Nature, London*, **302**, 520-522.

Birch, G.F. 1979a. Phosphatic rocks on the western margin of South Africa. *Journal of Sedimentary Petrology*, **49**: 93-110.

Birch, G.F. 1979b. Phosphorite pellets and rock from the western continental margin and adjacent coastal terrace of South Africa. *Marine Geology*, **33**: 91-116.

Birch, G.F. 1980. A model of penecontemporaneous phosphatization by diagenetic and authigenic mechanism from the western margin of southern Africa. pp. 79-100 In: Y.K. Benthor, Marine phosphorites - geochemistry, occurrence, genesis, Tulsa, *SEPM Spec. Publ.*, **29**.

Birch, G.F., J. Thomson, J.M. McArthur and W.C. Burnett 1983. Pleistocene phosphorites off the west coast of South Africa. *Nature*, **302**: 601-603.

Blaxter, J.H.S. 1977. The effect of copper on the eggs and larvae of plaice and herring. *Journal of the Marine Biological Association of the United Kingdom*, **57**: 849-856.

Blissenbach, E. and Z. Nawab 1982. Metalliferous sediments of the seabed: the Atlantis-II-Deep deposits of the Red Sea. pp.77-104 In E. Mann Borgese and N. Ginsburg (eds) *Ocean Yearbook 3*, Univ. Chicago Press, Chicago, 1982.

Bluhm, H. 1993. Effects of deepsea mining for manganese nodules on the abyssal megabenthic community. *Proc. 25th Ann. Offshore Technol. Conf.*, Houston, Texas, 3-6 May 1993, Vol. **1** (OTC Paper 7134): 521-529.

Bluhm, H., G. Schriever and H. Thiel 1995. Megabenthic recolonization in an experimentally disturbed abyssal manganese nodule area. *Marine Geosciences and Geotechnology*, **13**, 393-416.

Bothner, M.H., H. Takada, I.T. Knight, R.T. Hill, B. Butman, J.W. Farrington, R.R. Colwell and J.F. Grassle 1994. Sewage contamination in sediments beneath a deep-ocean dump site off New York. *Marine Environmental Research,* **38**: 43-59.

Bremner, J.M. 1980. Concretionary phosphorite from S.W. Africa. *Journal of the Geological Society, London,* **137**: 773-786.

Bremner, J.M. and J. Rogers 1990. Phosphorites deposits on the Namibian continental shelf. pp.143-152 In: W.C. Burnett and S.R. Riggs, *Phosphate deposits of the world: Genesis of Neogene to Recent phosphorites,* Vol. **3**, Cambridge University Press, Cambridge.

Brewer, R.G. 1975. Minor elements in sea water. pp. 415-491 In: J.P. Riley and G. Skirrow, *Chemical Oceanography*, Vol. **1**, London, Academic Press.

Brewer, P.G., D.M. Glover, C. Goyet and D.K. Shafer 1995. The pH of the North Atlantic Ocean: Improvements to the global model for sound absorption in seawater. *J. Geophys. Res.*, **100**: 8761-8776.

Broecker, W.S. 1991. The great ocean conveyor. *Oceanography*, **4**: 79-89.

Browitt, L.W.A. 1995. Beaufort's Dyke briefing note. *Brit. Geol. Surv.*, Edinburgh

Buchannon, R.C. 1995. *A model of the economic feasibility of utilizing Blake Plateau manganese nodules for desulfurization and metals recovery.* Unpublished PhD dissertation, Univ. of Mississippi. 91 pp.

Burnett, W.C. 1977. Geochemistry and origin of phosphorite deposits from off Peru and Chile. *Geological Society of America Bulletin*, **88**: 813-823.

Burnett, W.C. 1990. Phosphorite growth and sediment dynamic in the Modern Peru shelf upwelling system. pp. 62-72 In: W.C. Burnett and S.R. Riggs (eds), *Phosphate deposits of the world: Genesis of Neogene to Recent phosphorites*, Vol. **3**, Cambridge University Press, Cambridge.

Burnett, W.C., M.J. Beers and K.K. Roe 1982. Growth rates of phosphate nodules from the continental margin off Peru. *Science*, **215**: 1616-1618.

Burnett, W.C., K.K. Roe and D.Z. Piper 1983. Upwelling and phosphorite formation in the ocean. pp. 377-397 In: J. Thiede and E. Suess, *Coastal upwelling, its sediment record, Part A*. New York, Plenum Press.

Burnett, W.C., D.J. Cullen and G.M. McMurtry 1987. Open-ocean phosphorites - in a class by themselves? pp. 119-131 In: P.G. Teleki, M.R. Dobson, J.R. Moore and U. von Stackelberg (eds), *Marine Minerals: Resource Assessment strategies*. Reidel, Dordrecht.

Burnett, W.C., K.B. Baker, P.A. Chin, W. McCabe and R. Ditchburn 1988. Uraniumseries and AMS 14^{C} studies of modern phosphatic pellets from Peru shelf muds. *Marine Geology*, **80**: 215-230.

Burnett, W.C. and P.N. Froelich 1988. The origin of marine phosphorite. The results of the R.V. Robert D. Conrad cruise 23-06 to the Peru shelf - Preface. *Marine Geology,* **80**: I-VI.

Cavanaugh, C.M. 1994. Microbial symbiosis: Patterns of diversity in the marine environment. *Amer. Zool.* **34**: 79-89

Chapman, D.V., C.D. Byrne, R.J. Law and M.J.Waldock 1986. A comparison of methods for measuring the dispersion in the water column of sewage sludge dumped at sea. pp. 131-133 In: *Environmental Contamination*, CEP Consultants, Edinburgh.

Chen, C.-T.A. 1993. The oceanic anthropogenic CO_2 sink. *Chemosphere*, **27**: 1041-1064.

Chryssostomidis, C. 1991. An abyssal ocean option for future waste management monitoring: Engineering systems and research needs. *Report of a Workshop held at the Massachusetts Institute of Technology*, June 12-14, 1991. Woods Hole Oceanographic Institution, Woods Hole, MA.

Claque, D., J. Bischoff and D. Howell 1984. Nonfuel mineral resources of the Pacific Exclusive Economic Zone. *Proc., OCEANS '84,* Vol. **1**: 438-443.

Clark, A., C. Johnson and P. Chin 1984. Assessment of cobalt-rich manganese crusts in the Hawaiian, Johnston and Palmyra Islands Exclusive Economic Zones. *Natural Resources Forum,* **8**(2): 163-174.

Clark, A.L. and C.J. Johnson 1986. Cobalt-rich manganese crust potential of the U.S. Trust and affiliated territories. *Proc. Offshore Technol. Conf.*, Houston, Texas, 5 - 8 May 1986, Vol. **3** (OTC Paper 5233): 111-118.

Clark, A.L., J.A. Lum, C. Li, W. Icay, C. Morgan and Y. Igarashi 1995. Economic and developmental potential of manganese nodules within the Cook Islands Exclusive Economic Zone (EEZ). East-West Center, Univ. Hawaii, Honolulu. 34 pp.

Clarke, G.K. 1989. The fate of trace metals at the Garroch Head sewage sludge site in the Firth of Clyde, Scotland. PhD thesis, Napier Polytechnic of Edinburgh.

Cohen, J.E. 1994. Marine and continental food webs: three paradoxes? *Philosophical Transactions of the Royal Society of London*, **B 343**: 57-69.

Cole, K.H., G.R.Stegen and D. Spencer 1995. The capacity of the deep oceans to absorb carbon dioxide. pp.143-152. In: N. Handa and T. Ohsumi (eds) *Direct Ocean Disposal of Carbon Dioxide*, Terra Scientific Publishing Co, Tokyo.

Collins, M.B., F.T. Banner, P.A. Tyler, S.J. Wakefield and A.E. James (eds) 1980. *Industrialised Embayments and Their Environmental Problems: A case study of Swansea Bay.* Pergamon Press, Oxford. 616 pp.

Colombo, P., R.M. Neilson and M.W. Kendig 1983. Analysis and evaluation of a radioactive waste package retrieved from the Atlantic Ocean. In: *Wastes in the Ocean*, Vol. **3**. Wiley-Interscience, New York.

Commeau, R.F., A. Clark, C. Johnson, F.T. Manheim, P.J. Aruscavage and C.M. Lane 1984. Ferromanganese crust resources in the Pacific and Atlantic Oceans. *Proc., OCEANS '84*, Vol. **1**: 421-430.

Cook, P.J. 1988. Eustatic sea-level changes, environments, tectonics and resources. Bruunmemorial lecture, 1987. *Intergovernm. Oceanogr. Comen. (UNESCO), Techn. Ser.*, **34**: 35-57.

Cook, P.J. and M.W. McElhinny 1979. A reevaluation of the spatial and temporal distribution of sedimentary phosphate deposits in the light of plate tectonics. *Economic Geology*, **74**: 315-330.

Cook, P.J. and G.W. O'Brion 1990. Neogene to Holocene phosphorites of Australia. pp. 98-115 In: W.C. Burnett and S.R. Riggs (eds), *Phosphate deposits of the world: Genesis of Neogene to Recent phosphorites*, Vol. **3**, Cambridge University Press, Cambridge.

Corliss, J.B., J. Dymond, L.I. Gordon, J.M. Edmond, R.P. von Herzen, R.D. Ballard, K. Green, D. Williams, A. Bainbridge, K. Crane and T.H. van Andel 1979. Submarine thermal springs on the Galapagos Rift. *Science*, **203**(4385): 1073-1083.

Costello, M.J. and J.C. Gamble 1992. Effects of sewage sludge on marine fish embryos and larvae. *Marine Environmental Research*, **33**: 49-74.

Cronan, D.S. 1976. Basal metalliferous sediments from the eastern Pacific. *Geological Society of America Bulletin*, **87**(6): 928-934.

Cronan, D.S. 1977. Metalliferous sediments and manganese nodules on the ocean floor. *Chemistry and Industry* (July 1977): 576-578.

Cronan, D.S., R. Hodkinson and P. Stoffers 1992. Submarine hotspot hydrothermal mineralization in the Pitcairn Island EEZ, South Pacific. Abstracts, *23rd Underwater Mining Institute*, Washington, DC, Sept. 27 - 29, 1992.

Cruickshank, M.J. 1990. Mining technology for Gorda Ridge sulfides. pp. 211-221 In: G.R. McMurray (ed.) *Gorda Ridge - a seafloor spreading center in the United States Exclusive Economic Zone*. Springer Verlag, NY.

Cullen, D.J. 1980. Distribution, composition and age of submarine phosphorites on Chathman Rise, east of New Zealand. In: Y.K. Bentor (ed.), *Marine Phosphorites - geochemistry, occurrence, genesis*, Tulsa, SEPM special. Publication, **29**: 139-148.

D'Anglejan, B.F. 1967. Origin ofmarine phosphorites off Baja California, Mexico. *Marine Geology*, **5**: 15-44.

Dando, P.R., A.J. Southward, E.C. Southward, D.R. Dixon, A. Crawford and M. Crawford 1992. Shipwrecked tube worms. *Nature*, **356**: 667.

De Bruin, P. and C. Loos 1995. The use of Geotubes as an essential part of an 8.8m high North Sea dike and embankment, Leybrucht, Germany. *Geosynthetics World*, **5**: 7-10.

Degens, E.T. and D.A. Ross (eds) 1969. *Hot brines and recent heavy metal deposits in the Red Sea (A geochemical and geophysical account).* Springer Verlag, New York, NY. 600 p.

Department of the Environment (DoE) 1994. *Review of radioactive waste management policy preliminary conclusions. A consultation document.* London: Department of the Environment. 68 pp.

Desbruyéres, D., J.W. Deming, A. Dinet and A. Khripounoff. 1985. Reactions de l'ecosysteme benthique profond aux perturbations: Nouvaux resultats experimentaux. pp. 193-209 In: *Peuplements profonds du Golfe de Gascogne.* L. Laubier and C. Monniot (eds), Institute Francais de Recherche pour l'Exploitation de la Mer.

Dickson, R.R., W.J. Gould, T.J. Miller and C. Maillard 1985. Estimates of the mean circulation in the deep (>2000m) layer of the eastern North Atlantic. *Progress in Oceanography,* **14**: 103-127.

Drodt, M., A.X. Trautwein, I. König, E. Suess and C.B. Koch. 1997. Mössbauer spectroscopic studies on the iron forms of deep-sea sediments. *Phys. Chem. Minerals,* **24**: 281-293

Duckworth, R.C., A.E. Fallick and D. Rickard 1994. Mineralogy and sulfur isotopic composition of the Middle Valley massive sulfide deposit, northern Juan de Fuca Ridge. pp. 373-380 In: M.J. Mottl, E.E. Davis, A.T. Fisher and J.F. Slack (eds) *Proc. Ocean Drilling Program, Scientific Results,* Vol. **139**.

Duedall, I.W., H.B. O'Connors, S.A. Oakley and H.M. Stanford 1977. Short-term water column perturbation caused by wastewater sludge dumping in the New York Bight apex. *Journal of the Water Pollution Control Federation,* **10**: 2074-2080.

Dyer, R.S. 1976. Environmental surveys of two deep-sea radioactive waste disposal sites using submersibles. *Management of radioactive wastes from the nuclear fuel cycle, Vol II,* International Atomic Energy Agency, Vienna, 1976: 317-328.

Edmond, J.M. 1984. The geochemistry of ridge crest hot springs. *Oceanus,* **27**(3): 15-19.

Edmond, J.M., C. Measures, B. Mangum, B. Grant, F.R. Sclater, R. Collier, A. Hudson, L.I. Gordon and J.B. Corliss 1979. On the formation of metal-rich deposits at ridge crests. *Earth and planetary Science Letters,* **46**: 19-30.

Edwards, R. 1995. Danger from the deep. *New Scientist,* **148** (2004): 16-17.

Emery, K.O. 1960. *The sea off southern California, a modern habitat of petroleum.* Wiley, New York, 366 pp.

ENEA 1968. Radioactive waste disposal operation into the Atlantic 1967. Paris: OECD, European Nuclear Energy Agency, 74pp.

Erlenkeuser, H. and W. Balzer. 1988. Rapid appearance of Chernobyl radiocesium in the deep Norwegian Sea sediments. *Oceanolog. Acta,* **11**: 101-106.

Fava, J.A., J.J. Gift, A.F. Maciorowski, W.L. McCulloch and H.J.H. Reisinger 1985. Comparative toxicity of whole and liquid phase sewage sludges to marine organisms. pp. 229-252 In: R.D. Cardwell, R. Purdy and R.C. Bahner (eds) *Aquatic Toxicology and Hazard Assessment: Eleventh Symposium.* ASTM STP 854. American Society for Testing and Materials, Philadelphia.

Feldt, W., G. Kanisch and M. Vobach.. 1989. Deep-sea biota of the Northeast Atlantic and their radioactivity. pp. 178-204. In: *Interim oceanographic description of the Northeast Atlantic site for the disposal of low-level radioactive waste*, volume **3**. Paris: OECD Nuclear Energy Agency, 374 pp.

Feldt, W., G. Kanisch, M. Kanisch and M. Vobach. 1985. Radioecological studies of sites in the Northeast Atlantic used for dumping of low-level radioactive wastes - results of the research cruises of FRV "Walter Herwig" 1980-1984. *Arch. Fischwiss.*, **35**: 91-195.

Fellerer, R. 1980. Manganknollen. *Geologisches Jahrbuch*, **D38**: 35-76.

Fiala, G., C.R. Woese, T.A. Langworthy and K.O. Stetter. 1990. *Flexistipes sinusarabici*, a novel genus and species of eubacteria occuring in the Atlantis II Deep brines of the Red Sea. Arch. Microbiol., **154**: 120-126.

Fisher, C.R. 1995. Toward an appreciation of hydrothermal-vent animals: Their environment, physiological ecology, and tissue stable isotope values. pp. 297-316. In: S.E. Humphris, R. Zierenberg, L.S. Mullineaux and R.E. Thomson (eds): *Seafloor Hydrothermal Systems: Physical, Chemical, Biological, and Geochemical Interactions.* AGU Monographs Series.

Foell, E.J., H. Thiel and G. Schriever 1990. DISCOL - a long-term, large-scale, disturbance-recolonization experiment in the abyssal eastern tropical South Pacific Ocean. *Proc. Offshore Technol. Conf.*, Houston, Texas, 7-10 May 1990, Vol. **2** (OTC Paper 6328): 497-503.

Foell, E.J., H.F. Hennigar, Jr. and D.L. Pawson 1991. Assessment of abyssal benthic megafauna on a ferromanganese nodule deposit using videotaped television survey data. *Transactions of the Society for Mining, Metallurgy and Exploration*, **288**: 1854-1858.

Foell, E.J., H. Thiel and G. Schriever 1992. DISCOL: a long-term, large-scale, disturbance-recolonization experiment in the abyssal eastern tropical South Pacific. *Mining Engineering* (January 1992): 90-94.

Francheteau, J., H.D. Needham, P. Choukroune, T. Juteau, M. Seguret, R.D. Ballard, P.J. Fox, W. Normark, A. Carranza, D. Cordoba, J. Guerrero, C. Rangin, H. Bougault, P. Cambon and R. Hekinian 1979. Massive sulphide ore deposits discovered on the East Pacific Rise. *Nature*, **277**(5697): 523-528.

Franklin, F.L. 1983. Laboratory tests as a basis for the control of sewage sludge dumping at sea. *Mar. Poll. Bull.*, **14**: 217-233.

Friedrich, G. and A. Schmitz-Wiechowski 1980. Mineralogy and chemistry of a ferromanganese crust from a deep-sea hill, Central Pacific, "Valdivia" Cruise 13/2. *Mar. Geol.*, **37**: 71-90.

Frithsen, J.B., D. Nacci, C. Oviatt, C.J. Strobel, and R. Walsh 1989. Using single-species and whole ecosystem tests to characterise the toxicity of a sewage treatment plant effluent. pp. 231-250 In: G.W. Suter II and M.A. Lewis (eds) *Aquatic Toxicology and Environmental Fate*, **11**, ASTM STP 1007, American Society for Testing and Materials, Philadelphia.

Froelich, P.N., M.A. Arthur, W.C. Burnett, M. Deakin, V. Hensley, R. Jahnke, L. Kaul, K.-H. Kim, K. Poe, A. Soutar and C. Vathakanon 1988. Early diagenesis of organic matter in Peru continental margin sediments: Phosphorite precipitation. *Marine Geology*, **80**: 309-343.

Fukushima, T. 1995. Overview - Japan Deep-sea Impact Experiment: JET. *Proc. 1st ISOPE - Ocean Mining Symposium*, Tsukuba, Japan: 47-53.

Gage, J.D. and P.A. Tyler 1991. *Deep-sea biology. A natural history of organisms at the deep-sea floor.* Cambridge Univ. Press, Cambridge, 490 pp.

Galil, B.S., A. Golik and M. Türkay 1995. Litter at the bottom of the sea: A sea bed survey in the eastern mediterranean. *Mar. Poll. Bull.* **30**, 22-24.

Garnett, R.H.T. 1995. Offshore diamond mining in South Africa. *Proc. 27th Ann. Offshore Techn. Conf.*, Houston, Texas, OTC Paper 7643: 71-86.

Garnett, R.H.T. 1996. Marine dredging for gold, offshore Alaska. *Proc. 28th Ann. Offshore Techn. Conf.*, Houston, Texas, OTC Paper 8008: 455-463.

Garnett, R.H.T. and D.V. Ellis 1995. Tailings disposal at a marine placer mining operation by West Gold, Alaska. *Mar. Georesourc. and Geotechn..*, **13**: 41-57.

Geminder, R. and E.J. Lecourt, Jr. 1972. Deep ocean mining system tested.*World Dredging and Marine Construction*, **8**(8): 35-38.

GESAMP 1983. *An oceanographic model for the dispersion of wastes disposed of in the deep sea.* IMO/FAO/UNESCO/WMO/WHO/IAEA/UN/UNEP Joint Group of Experts on the Scientific Aspects of Marine Pollution Reports and Studies, **19**, 182 pp. [IAEA Vienna].

GESAMP 1990. *The state of the marine environment.* UNEP Regional Seas Reports and Studies, **115.**

Glenn, C.R. and M.A. Arthur 1988. Petrology and major element geochemistry of Peru margin phosphorites and associated diagenetic materials: authigenesis in modern organic-rich sediments. *Marine Geology*, **80**: 231-267.

Goldberg, E.D. 1995. Emerging problems in the coastal zone for the twenty-first century. *Marine Pollution Bulletin*, **31**: 152-158.

Gorsline, D.S. and D.B. Milligan 1963. Phosphatic deposits along the margin of the Pourtalés Terrace, Florida. *Deep-Sea Research*, **10**: 259-262.

Grassle, J.F. 1977. Slow recolonization of deep-sea sediment. *Nature, London.* **265**: 618-619.

Grassle, J.F. and N.J. Maciolek 1992. Deep-sea richness: regional and local diversity estimates from quantitative bottom samples. *The American Naturalist*, **139**: 313-341.

Grassle, J.F. and L.S. Morse-Porteous 1987. Macrofaunal colonization of disturbed deep-sea environments and the structure of deep-sea benthic communities. *Deep-Sea Res.* **43**: 1911-1950.

Greenslate, J.L., M. Fisk and D. Felix 1979. Systematics in manganese concretion composition in the northeastern equatorial Pacific.pp. 39-54 In: La Genese des Nodules de Manganese. *Colloques Internationaux du CNRS, No. 289.* Paris, 410 pp.

Greenwald, R.J. and H.F. Hennigar, Jr. 1988. Designation of an ocean mining Stable Reference Area. *Proc., OCEANS '88* , Vol. **3**: 752-761.

Guillaumont, G., A. Cottenie, J. Harrison, J.D. Schmitt-Tegge, R.T.E. Schüttenhelm, and J.K. Wright 1989. Evaluation of the Third Community Programme "Radioactive Waste Management and Storage" (1985-1989). *Commission of the European Communities Research Evaluation Report* , **40** [EUR 12264 EN], 95 pp.

Guney, M., Z. Nawab and M.A. Marhoun 1984. Alantis-II-Deep metal reserves and their evaluation. *Proc. Offshore Technol. Conf.*, Houston, Texas, 7 - 9 May 1984, Vol. **3** OTC Paper 4780: 33-44.

Halbach, P. and F.T. Manheim 1984. Potential of cobalt and other metals in ferromanganese crusts on seamounts of the Central Pacific Basin. *Marine Mining*, **4**(4): 319-335.

Halbach, P., C.D. Sattler, F. Teichmann and M. Wahsner 1987. Characterization of Co-rich ferromanganese crust fields within the Johnston Island EEZ (Central Pacific). *Proc., OCEANS '87*, Vol. **3**: 1015-1020.

Halbach, P., G. Friedrich and U. von Stackelberg (eds). 1988. *The manganese nodule belt of the Pacific Ocean: geological environment, nodule formation, and mining aspects*. F. Enke Verlag, Stuttgart, 254 pp.

Halbach, P., K. Nakamura, M. Wahsner, J. Lange, H. Sakai, L. Kaselitz, R.D. Hansen, M. Yamano, J. Post, B. Prause, R. Seifert, W. Michaelis, F. Teichmann, M. Kinoshita, A. Marten, J. Ishibashi, S. Czerwinski and N. Blum 1989. Probable modern analogue of Kuroko-type massive sulphide deposits in the Okinawa Trough back-arc basin. *Nature*, **338**: 496-499.

Halbach, P.E., C.D. Sattler, F. Teichmann and M. Wahsner 1989. Cobalt-rich and platinum-bearing manganese crust deposits on seamounts: nature, formation, and metal potential. *Marine Mining*, **8**(1): 23-39.

Halbach, P.E., N. Blum and W. Plüger 1994. The SONNE orefield - the first massive sulfides from the Indian Ocean floor. Abstracts, *25th Underwater Mining Institute*, Monterey Bay, CA, Dec. 1 - 4, 1994.

Halkyard, J.E. 1985. Technology for mining cobalt rich manganese crusts from seamounts. *Proc., OCEANS '85*, Vol. **1**: 352-374.

Hall, R.R. 1993. Immiscible-liquid extraction of cobalt-rich ferromanganese crust. Abstracts, *24th Underwater Mining Institute*, Estes Park, CO, Nov. 7 - 10, 1993.

Hannington, M.D., J.M. Franklin and P.M. Herzig 1994. Base and precious metal resources in seafloor polymetallic sulfides. Abstracts, *25th Underwater Mining Institute*, Monterey Bay, CA, Dec. 1 - 4, 1994.

Haugan, P.M. 1997. Direct ocean storage of CO_2: options and obstacles. *Proc. Seventh Intern. Offshore Polar Engineer. Conf.*, Honululu. **1**: 499-504

Haugan P.M. and H. Drange 1992. Sequestration of CO_2 in the deep ocean by shallow injection. *Nature,* London, **357**: 318-320.

Haugan P.M. and H. Drange 1996. Effects of CO_2 on the ocean environment. *Energy Conversion Management,* **37**: 1019-1022.

Haugan P.M., F. Thorkildsen and G. Alendal 1995. Dissolution of CO_2 in the ocean. *Energy Conversion Management,* **36**: 461-466.

Haymon, R.M. and M. Kastner 1981. Hot spring deposits on the East Pacific Rise at 21° N: preliminary description of mineralogy and genesis. *Earth and planetary Science Letters,* **53**(3): 363-381.

Heggie, D.T., G.W. Skying, G.W. O'Brion, C. Reimers, A. Herczeg, D.J.W. Moriarty, W.C. Burnett and A.R. Milnes 1990. Organic carbon cycling and modern phosphorite formation on the East Australian continental margin: an overview. In: Notholt, A.G.J. and I. Jarvis (eds): *Phosphorite research and development.* London, Geol. Soc. Publ., **52**: 87-117.

Hein, J.R., F.T. Manheim and W.C. Schwab 1986. Cobalt-rich ferromanganese crusts from the Central Pacific. *Proc. Offshore Technol. Conf.*,Vol. **3** (OTC Paper 5234):119-126.

Hekinian, R., M. Fevrier, F. Avedik, P. Cambon, J.L. Charlou, H.D. Needham, J. Raillard, J. Boulegue, L. Merlivat, A. Moinet, S. Manganini and J. Lange 1983. East Pacific Rise near 13°N: geology of new hydrothermal fields. *Science*, **219**(4590):1321-1324.

Hekinian, R., M. Fevrier, J.L. Bischoff, P. Picot and W.C. Shanks 1980. Sulfide deposits from the East Pacific Rise near 21°N. *Science*, **207**(4438): 1433-1444.

HELCOM CHEMU 1994. *Chemical munitions dumped in the Baltic Sea.* Report to the 16th Meeting of the Helsinki Commission. 43 pp.

Hemming, C.R. and T.J. Freeman 1988. *Feasibility of deep ocean disposal of heat generating waste. Vol. 1. Overview of feasibility.* Department of the Environment Report, No. DOE/RW/88.099, 18 pp. [Department of the Environment Commissioned research on radioactive wste management 1988/89].

Hencke, D. 1995. MOD releases details of 71,000 bombs dumped at sea. *Guardian* , 28 March 1995.

Herdendorf, C.E., T.G. Thompson and R.D. Evans 1995. Science on a deep-ocean shipwreck. *Ohio Journal of Science*, **95**: 4-212.

Hess, H.D. 1978. *Preliminary resource assessment - phosphorites of Southern California Borderland.* Washington, US Geological Survey open-file report, **51**.

Hessler, R.R. and W.M. Smithey 1983. The distribution and community structure of megafauna at the Galapagos Rift hydrothermal vents. pp 735-770. In: P.A. Rona, K. Bostrom, L. Laubier and K.L. Smith (eds). *Hydrothermal Processes at Seafloor Spreading Centers*, NATO Conference Series IV. New York: Plenum Press.

Hessler, R.R., W.M. Smithey, M.A. Boudrias, C.H. Keller, R.A. Lutz and J.J. Childress 1988. Temporal change in megafauna at the Rose Garden hydrothermal vent (Galapagos Rift; eastern tropical Pacific). *Deep-Sea Research*, **35** (10/11A): 1681-1709.

Hinrichsen, D. 1989. Living on the edge. *Earthwatch*, **35**: 2.

Hirai, S., K. Okazaki, T. Tabe, K. Hijikata and Y. Mori in press. Dissolution rate of liquid CO_2 in pressurized water flow and effect of Clathrate film. *Energy International Journal.*

Hirt, W.C. 1992. Column flotation of cobalt-rich ferromanganese crust. Abstracts, *23rd Underwater Mining Institute*, Washington, DC, Sept. 27 - 29, 1992.

HMSO 1986. Assessment of Best Practicable Environmental Options (BPEOs) for Management of Low- and Intermediate-Level Solid Radioactive Wastes. London: HMSO, 80 pp. [Radioactive Waste (Professional) Division of the Department of the Environment].

Hobbs, C.H., III 1991. Marine mineral resources of the U.S. middle and south Atlantic coasts. *Marine Mining*, **10**(3): 215-230.

Holliday, F.G.T. [Chairman] 1984. Report of the Independent Review of Disposal of Radioactive Waste in the Northeast Atlantic. London: HMSO, 60 pp.

Holloway, S., J.P. Heederik, L.G.H. van der Meer, I. Czernichowski-Lauriol, R. Harrison, E. Lindeberg, I.R. Summerfield, C.Rochelle, T. Scharzkopf, O. Kaarstad and B. Berger 1996. *The Underground Disposal of Carbon Dioxide: Summary Report.* Non-Nuclear Energy R&D Programme JOULE II, CEC; British Geological Survey, Keyworth,UK. 24 pp.

Honda, M., J. Hasimoto, J. Naka and H. Hotta 1995. CO_2 hydrate formation and inversion of density between liquid CO_2 and H_2O in deep sea: Experimental study using submersible *Shinkai 6500*. pp. 35-44 In: N. Handa and T. Ohsumi (eds) *Direct Ocean Disposal of Carbon Dioxide*, Terra Scientific Publishing Co, Tokyo.

Houghton, J.T., L.G. Meira Filho, J. Bruce, H. Lee, B.A.Callander, E. Haites, N. Harris and K.Maskell (eds) 1995. *Climate Change 1994. Radiative Forcing of Climate Change and an Evaluation of the IPCC IS92 Emission Scenario.* Cambridge University Press.

Hurd, A. 1929. *The Merchant Navy*, Volume **3**, John Murray, London.

IAEA 1978. Convention on the Prevention of Marine Pollution by Dumping Wastes and Other Matter. The Definition required by Annex 1 paragraph 6 to the Convention, and the Recommendaton required by Annex II, section D. INFCIRC/205/Add.1/Rev.1. International Atomic Energy Agency, Vienna.

IMO 1994. Disposal at sea of obsolete ammunition - prior reporting procedure. Notification by the Government of South Africa. London Convention 1972 LC.2/Circ.347 5 December 1994.

Institute of London Underwriters 1995. *Annual Report for 1995*.

International Offshore Oil and Natural Gas Exploration and Production Industry 1996. *Decommissioning offshore oil and gas installations: finding the right balance,* 16 pp.

Jagt, H.J. 1988. Bed protection, Old Meuse, by means of geocontainers (R) *Report VXT/HJ 88.168, Nicolon b.v. Almelo, The Netherlands*. Report for the Publics World Department of the Netherlands, Dordrecht, 19 pp.

Jahnke, R.A., S.R. Emerson, K.K. Roe and W.C. Burnett 1983. The present day formation of apatite in Mexican continental margin sediments. *Geochim. Cosmochim. Acta*, **47**: 259-266.

Jankowski, J.A., A. Malcherek and W. Zielke 1996. Numerical modelling of suspended sediment due to deep-sea mining. *J. Geophys. Res.*, **101** (C5): 3545-3560.

Jankowski, J.A. and W. Zielke 1997. Data support for modelling of deep-sea mining impacts. *Proc. Seventh Intern. Offshore Polar Engineer. Conf.*, Honululu. **1**: 451-460.

Jenkins, W.A. and R.J. Gibbs 1988. Settling and dispersion of ocean dumped sewage sludge. *Marine Pollution Bulletin*, **19**: 120-124.

Jenkyns, H.C. 1977. Fossil nodules. pp. 87-108 In: G.P. Glasby (ed). *Marine Manganese Deposits*. Elsevier, Amsterdam.

Johnson, L.J. and C.L.J. Frid 1995. The recovery of benthic communities along the Durham coast after cessation of colliery spoil dumping. *Marine Pollution Bulletin*, **30**: 215-220.

Johnson, R.G., M. Kahn and C. Robbins 1984. United States Practices and Policies for Ocean Disposal of Radioactive Wastes, 1946-1984. *EPA 530/1-84-017*. US Environmental Protection Agency, Washington, D.C.

Jones, M.L. (ed.) 1985. Hydrothermal vents of the eastern Pacific: an overview. *Bulletin of the Biological Society of Washington*, **6**: 1-547.

Kaarstad, O. 1994. Norway pioneers large scale CO_2 disposal in 1996. *Greenhouse Issues*, **No 15,** August 1994.

Kajishima, T., T. Saito, R. Nagaosa and H. Hatano 1995. A gas-lift system for CO_2 release into shallow water. *Energy Conversion Management*, **36**: 467-470.

Kaneko, S.T., K. Ogawa and T. Fukushima 1995. Preliminary results of meiofauna and bacteria abundance in an environmental impact experiment. *Proc. 1st ISOPE Ocean Mining Symp.*, Tsukuba, Japan: 181-186.

Kaneko, S.T., Y. Maejima and H. Teishima 1997. The abundance and vertical distribution of abyssal benthic fauna in the Japan deep-sea impact experiment. *Proc. Seventh. Intern. Offshore Polar Engineer. Conf.*, Honululu. **1**: 475-480.

Karbe, L. 1987. Hot brines and the deep sea environment. pp. 70-89 In: A.J. Edwards and S.M. Head (eds) *Red Sea (Key Environments Series)*. Pergamon Press, Oxford.

Karbe, L., H. Thiel, H. Weikert and A.J.B. Mill (eds) 1981. Mining of metalliferous sediments from the Atlantis-II-Deep, Red Sea: pre-mining environmental conditions and evaluation of the risk to the environment. Unpublished EIS Report to the Saudi-Sudanese Red Sea Joint Commission, Jeddah.

Kaufman, R. 1985. Conceptional approaches for mining marine polymetallic sulfide deposits. *MTS Journal,* **19**: 50-56.

Kaufman, R. and J.P. Latimer 1971. The design and operation of a prototype deep-ocean mining ship. *Proc. Soc. Naval Arch. and Mar. Eng.*, Spring Meeting, Honolulu, Hawaii, Vol. **3**: 1-24.

Keller, M., D.A. Clark, D.B. Clark, A.M. Weitz and E. Veldkamp 1996. If a tree falls in the forest. *Science* **273**: 201.

Klausewitz, W. 1989. Evolutionary history and zoogeography of the Red Sea ichthyofauna. *Fauna Saudi Arabia*, **10**: 310-337.

Klausewitz, W. 1994. Comparative studies on the vertical distribution of bathybenthic deep-sea fishes of the Red Sea. *Proc. fourth Indo-Pacific Fish Conf.*, 462-468.

König, I., M. Drodt, E. Suess and A..X. Trautwein 1997. Iron reduction through the tangreen color transition in the deep-sea sediments. *Geochim. Cosmochim. Acta.*Vol. **61** (8): 1679-1683

Koschinsky, A. and P. Halbach 1995. Sequential leaching of marine ferromanganese precipitates - genetic implications. *Geochim. Cosmochim. Acta,* **59**: 5113-5132.

Koschinsky, A., H. Gerber and A. Szemeitat 1997. Experiments on the influence of technical activities in the deep-sea on heavy metal cycles. *Proc. Seventh Intern. Offshore Polar Engineer. Conf.*, Honululu. **1**: 445-450.

Koski, R.A., W.R. Normark and J.L. Morton 1985. Massive sulfide deposits on the southern Juan de Fuca Ridge: results of investigations in the USGS study area, 1980-83. *Marine Mining,* **5**(2): 147-164.

Kotlinski, R. 1995. InterOceanMetal Joint Organization: achievements and challenges. *Proc. ISOPE Ocean Mining Symposium*, Tsukuba, Japan, Nov. 21-22, 1995: 5-7.

Kudrass, H.-R. 1984. The distribution and reserves of phosphorite on the central Chatham Rise (Sonne-17 cruise 1981). pp. 179-194 In: U. von Rad and H.R. Kudrass (eds), *Phosphorite deposits on the Chatham Rise of New Zealand: results of a prospection cruise with R/V Sonne*. Hannover, BGR, **D 65**.

Kudrass, H.-R. and U. von Rad 1984a. Underwater television and photography observations, side-scan sonar and acoustic reflectivity measurements of phosphorite-rich areas on the Chatham Rise (New Zealand). pp. 69-80 In: U. von Rad and H.R. Kudrass (eds), *Phosphorite deposits on the Chatham Rise of New Zealand: results of a prospection cruise with R/V Sonne*. Hannover, BGR, **D 65**.

Kudrass, H.-R. and U. von Rad 1984b. Geology and some mining aspects of the Chatham Rise phosphorite: a synthsis of Sonne-17 results. pp. 233-252 In: U. von Rad and H.R. Kudrass (eds), *Phosphorite deposits on the Chatham Rise of New Zealand: results of a prospection cruise with R/V Sonne*. Hannover, BGR, **D 65**.

Länder Government Working Group 1993. *Chemical Munitions in the Southern and Western Baltic Sea; Compilation, assessment and Recommendations*. Bundesamt für Seeschiffahrt und Hydrographie. Hamburg, 59 pp.

Lamboy, M. and J. Lucas 1979. Les phosphorites de la marge Nord de l'Espagne. Étude géologique et pétrographique. *Oceanologica Acta*, **2**: 325-337.

Lamoureux, E.M., B.J. Brownawell and M.H. Bothner 1996. Linear alkylbenzenes as tracers of sewage sludge derived inputs of organic matter, PCBs and PAHs to sediments at the 106-mile deep water disposal site. *J. Mar. Environ. Eng*., **2** (3-4): 325-342.

Lampitt, R.S. 1985 Evidence for the seasonal deposition of detritus to the deep-sea floor (Porcupine Bight, N.E. Atlantic) and its subsequent resuspension. *Deep-Sea Research*, **32A**: 885-897.

Lampitt, R.S., E. Fahrbach, B. Schneider, R.R. Schneider, S.V. Smith, J.R. Toggweiler, A.J. Watson and B. Zeitschel 1995. Group Report: Does upwelling have a significant influence on the global carbon cycle? pp. 383-404 In: C.P. Summerhayes, K.C. Emeis, M.V. Angel, R.L. Smith and B. Zeitschel (eds) *Upwelling in the Oceans: Modern Processes and Ancient Records*, John Wiley and Sons Ltd.

Latimer, J.P. and R. Kaufman 1985. Preliminary considerations for the design of cobalt crust mining systems in the U.S. EEZ. *Proc., OCEANS '85*, Vol. **1**: 378-399.

Le Suave, R., C. Pichocki, G. Pautot, M. Hoffert, Y. Morel, M. Voisset, S. Monti, J. Amosse and A. Kosakevitch 1989. Geological and mineralogical study of Co-rich ferromanganese crusts from a submerged atoll in the Tuamotu Archipelago (French Polynesia). *Marine Geology*, **87**: 227-247.

Lecourt, E.J., Jr. and D.W. Williams 1971. Deep ocean mining - new application for oil field and marine equipment. *Proc. Offshore Technol. Conf.*, Houston, Texas, 19 - 21 April 1971, OTC Paper 1412, Vol. **1**: 859-868.

Lenoble, J.P. 1989. The French research programme for deep sea bed mining of the polymetallic nodules deposits. IFREMER/AFERNOD (July 1989). 9 pp.

Lenoble, J.P. 1990. The French research programme for deep sea mining of the polymetallic nodule deposits. IFREMER/PDG/CNOD/90.096 (November 1990). 14 pp.

Levin, L.A. and C.R. Smith 1984. Response of background fauna to disturbance and enrichment in the deep sea: a sediment tray experiment. *Deep-Sea Res.*, **31**: 1277-1285.

Longhurst, A.R. 1995. Seasonal cycles of pelagic production and consumption. *Prog. Oceanogr.* **31**: 51-99.

Lonsdale. P. 1984. Hot vents and hydrocarbon seeps in the Sea of Cortez. *Oceanus*, **27**(3):21-24.

Lonsdale, P., R. Batiza and T. Simkin 1982. Metallogenesis at seamounts on the East Pacific Rise. *MTS Journal*, **16**(3): 54-61.

Lonsdale, P.F., J.L. Bischoff, V.M. Burns, M. Kastner and R.E. Sweeney 1980. A high-temperature hydrothermal deposit on the seabed at a Gulf of California spreading center. *Earth and planetary Science Letters*, **49**: 8-20.

Lucas, J., L. Prévôt and M. Lamboy 1978. Les phosphorites de la marge N de l'Espagne, Chimie, Minéralogie, Genése. *Oceanographica Acta*, **I**: 55-72.

MacDonald, K.C., K. Becker, F.N. Spiess and R.D. Ballard 1980. Hydrothermal heat flux of the "black smoker" vents on the East Pacific Rise. *Earth and planetary Science Letters*, **48**: 1-7.

Magnesen, T. and T. Wahl 1993. *Biological impact of deep sea disposal of carbon dioxide.* Technical Report **77A**: Nansen Environmental and Remote Sensing Center.

Malahoff, A. 1981. Comparison between Galapagos and Gorda spreading centers. *Proc. Offshore Technol. Conf.*, Vol. **4** (OTC Paper 4129): 115-121.

Malahoff, A. 1982a. A comparison of the massive submarine polymetallic sulfides of the Galapagos Rift with some continental deposits. *MTS Journal*, **16**(3): 39-45.

Malahoff, A. 1982b. Massive enriched polymetallic sulfides of the ocean floor - a new commercial source for strategic minerals? *Proc. Offshore Technol. Conf.*, Vol. **2** (OTC Paper 4293): 725-730.

Malahoff, A., C.E. McLain and M.A. Ransone 1982. Polymetallic sulfides - a new resource on the ocean floor. *Proc., OCEANS '82*, Washington, DC, 20 - 22 Sept. 1982: 1222-1224.

Manheim, F.T., R.M. Pratt and P.F. McFarlin 1980. Composition and origin of phosphorite deposits of the Blake Plateau. pp. 117-137 In: Y.K. Bentor (ed.), *Marine phosphorites - geochemistry, occurrence, genesis*. Tulsa, SEPM Special Publication, **29**.

Manheim, F.T., T.H. Ling and C.M. Lane 1983. An extensive data base for cobalt-rich ferromanganese crusts from the world oceans. *Proc., OCEANS '83*, Vol. **2**: 828-831.

Marchal, P. 1984. Préleveur libre autonome. Véhicule expérimental pour le ramassage des nodules et la reconnaissance des fonds sous-marins. *Proc. 2nd International Seminar on the Offshore Mineral Resources*, GERMINAL, Brest, France: 597-625.

Marchetti, C. 1977. On geoengineering and the CO_2 problem. *Climate Change* **1**: 59-68.

Marchig, V., H. Gundlach and shipboard scientific party 1987. Ore formation at rapidly diverging plate margins. Results of Cruise GEOMETEP 4. *Bundesanstalt Geowissenschaften Rohstoffe (Hannover) Circular*, **4**: 3-22.

Marland, G., R.J. Andres and T.A. Boden 1994. Global, regional and national CO_2 emissions. pp. 505-584 In: T.A. Boden, D.P. Kaiser, R.J. Sepanski and F.W. Stoss (eds). *Trend 93: A Compendium of Data on Global Change*. Oak Ridge, TN; Oak Ridge National Laboratory.

Marshall, J.F. 1979. The development of the continental shelf of northern New South Wales. *J. Aust. Geol. Geophys.*, **4**: 281-288.

Marvasti, A. and S. Riggs 1989. The U.S. phosphate industry: pressures for evolutionary change. *MTS Journal*, **23**(1): 27-36.

Masuda, Y. 1985. Continuous line bucket (CLB) mining systems. pp. 61-71 In: P.B. Humphrey (ed.) *Marine Mining: A New Beginning*. State of Hawaii, Dept. Planning and Economic Development, Honolulu. 319 pp.

Masuda, Y. 1987. Cobalt-rich crust mining by continuous line bucket. *Proc., OCEANS '87*, Vol. **3**: 1021-1026.

Masuda, Y. 1991. Crust mining plans of the Japan Resources Association. *Marine Mining*, **10** (1): 95-101.

Masuda, Y., M.J. Cruickshank and J.L. Mero 1971. Continuous bucket-line dredging at 12,000 feet. *Proc. Offshore Technol. Conf.*, Houston, Texas, 19 - 21 April 1971, Vol. **1** (OTC Paper 1410): 837-858.

Masuda, Y. and M.J. Cruickshank 1995. Study of the CLB mining system for nodule and crust recovery. *Proc. ISOPE Ocean Mining Symposium*, Tsukuba, Japan, Nov. 21-22, 1995: 91-97.

McAllister, D.E. 1993. How much land is there on Earth? For people? For nature? *Global Biodiversity*, **3**: 6-7.

McArthur, J.M. 1990. Fluorine deficient francolite. *Mineral. Mag.*, **54**: 508-510.

McArthur, J.M., J. Thomson, I. Jarvis, A.E. Fallick and G.F. Birch 1988. Eocene to pleistocene phosphogenesis off western South Africa. *Marine Geology*, **85**: 41-63.

McArthur, J.M., A.R. Sahami, M. Thirlwall, P.J. Hamilton and A.O. Osborn 1990. Dating phosphogenesis with strontium isotopes. *Geochimica Cosmochimica Acta*, **54**: 1343-1351.

McIntyre, A.D. 1995. Human impact on the oceans: The 1990s and beyond. *Marine Pollution Bulletin*, **31**: 147-151.

McKelvey, V.E. 1985. The U.S. phosphate industry: revised prospects and potential. *MTS Journal*, **19**(4): 65-67.

McKelvey, V.E., N.A. Wright and R.W. Bowen 1983. Analysis of world distribution of metal-rich subsea manganese nodules. *US Geol. Survey Circ.*, **886**: 1-55.

Mero, J.L. 1965. *The mineral resources of the sea. Elsevier Oceanography Series*, **1**. Elsevier, New York, 312 pp.

Mero, J.L. 1978. Ocean mining - a historical perspective. *Marine Mining,* **1**(3): 243-255.

Merrick, T.W. 1990. *U.S. population assistance: A continued priority for the 1990's?* Population Reference Bureau, April 1990, 20 pp.

Miller, A.R., C.D. Densmore, E.T. Degens, J.C. Hathaway, F.T. Manheim, P.F. McFarlin, R. Pocklington and A. Jokela 1966. Hot brines and recent iron deposits in deeps of the Red Sea. *Geochimica Cosmochimica Acta*, **30**: 341-359.

Miller, D.C., M. Marcy, W. Berry, C. Deacutis, S. Lussier, A. Kuhn, M. Heber, S.C.Schimmel and E. Jackim 1987. The acute toxicity of sewage sludge to marine fish, mysids and copepods. pp. 103-113 In: D.A. Wolfe and T.P. O'Connor (eds) *Ocean Processes in Marine Pollution* **5**, *Urban Wastes in Coastal Marine Environments*. Robert E. Krieger Publishing Company, Malabar, Fl.

Möller, H. 1992. Findings of investigations into fish diseases in the coastal waters of the North Sea and Baltic Sea. *Deutsche Hydrographische Zeitschrift*, **44**(5/6): 321-327.

Moorby, S.A., D.S. Cronan and G.P. Glasby 1984. Geochemistry of hydrothermal Mn-oxide deposits from the S.W. Pacific island arc. *Geochimica cosmochimica Acta,* **48**(3): 433-441.

Moreau, J.P. 1984. La construction navale et les projets miniers en mer. *Proc. 2nd International Seminar on the Offshore Mineral Resources,* GERMINAL, Brest, France: 427-440.

Morgan, C.L. and B.W. Selk 1984. Ore assays of massive sulfides from three spreading centers. *Proc., Offshore Technol. Conf.,* Vol. **3** (OTC Paper 4777): 9-16.

Mullins, H.T. and A.C. Neumann 1979. Geology of the Miami Terrace and its paleooceanographic implictions. *Marine Geology*, **30**: 205-232.

Murray, C.N. and 35 co-authors 1991. The feasibility of disposal of high level radioactive wastes into the seabed. *Radioactive Waste Management and the Nuclear Fuel Cycle*, **15** (2-3): 77-98.

Murray, J. and A. Renard 1891a. Deep-sea deposits. *Rep. scient. Results Voyage Challenger, Geol.*, **1**: 1-583.

Murray, J. and A.F. Renard 1891b. Voyage of H.M.S. Challenger during the years 1873-1876. *Rep. on Deep Sea Deposits, Chapt. VI*: 378-391.

Mustafa, Z. and H. Amann 1978. Ocean mining and protection of the marine environment in the Red Sea. *Proc. Offshore Technol. Conf.*, Houston, Texas, 8 - 11 May 1978, Vol. **2** (OTC Paper 3188): 1199-1214.

Mustafa, Z. and H. Amann 1980. The Red Sea pre-pilot mining test 1979. *Proc. Offshore Technol. Conf.*, Houston, Texas, 5 - 8 May 1980, Vol. **4** (OTC Paper 3874): 197-210.

Mustafa, Z., Z. Nawab, R. Horn and F. Le Lann 1984. Economic interest of hydrothermal deposits - the Atlantis II Project. pp. 509-538 In: *Proc., 2nd Intl Seminar on the Offshore Mineral Resources (GERMINAL)*, Brest, France, 19 - 23 March 1984.

Nakashiki, N., T. Ohsumi and N. Katano 1995. Technical view on CO_2 transportation onto the deep ocean floor and dispersion at intermediate depths. pp 183-193 In: N. Handa and T. Ohsumi (eds) *Direct Ocean Disposal of Carbon Dioxide*, Terra Scientific Publishing Co, Tokyo.

Narita, Y., T. Takakura and Y. Fujii 1990. Total metallurgical processing tests on cobalt-rich ferromanganese crust. *Proc. Fourth Pacific Congr. Mar. Sci. Technol. (PACON)*: 421-428.

National Research Council 1984. *Deep seabed stable reference areas.* Ocean Policy Committee/NRC, National Academy Press, Washington, DC. 74 pp.

Nawab, Z. 1984. Red Sea mining: a new era. *Deep-Sea Research*, **31**A(6-8): 813-822.

NEA 1980. Review of the continued suitability of the dumping site for radioactive waste in the North-East Atlantic. Paris: OECD Nuclear Energy Agency, 100 pp.

NEA 1981. Research and environmental surveillance programme related to sea disposal of radioactive waste. Paris: OECD Nuclear Energy Agency, 38 pp.

NEA 1982. Disposal of Radioactive Waste: an overview of the principles involved. Paris: OECD, 22 pp.

NEA 1983a. Interim oceanographic description of the North-east Atlantic site for the disposal of low-level radioactive waste, Volume **1**. Paris: OECD Nuclear Energy Agency, 132 pp.

NEA 1983b. Research and environmental surveillance programme related to the sea disposal of radioactive waste. Paris: OECD Nuclear Energy Agency, 55 pp.

NEA 1984. Seabed disposal of high-level radioactive waste. A status report on the NEA coordinated research programme. Paris: OECD Nuclear Energy Agency, 247 pp.

NEA 1985. Review of the continued suitability of the dumping site for radioactive waste in the North-east Atlantic. Paris: OECD Nuclear Energy Agency, 448 pp.

NEA 1986a. Research and environmental surveillance programme related to the sea disposal of radioactive waste. Paris: OECD Nuclear Energy Agency, 63 pp.

NEA 1986b. Iterim oceanographic description of the North-east Atlantic site for the disposal of low-level radioactive waste, Volume **2**. Paris: OECD Nuclear Energy Agency.

NEA 1988a-h. Feasibility of disposal of high-level radioactive waste into the seabed. Vol. **1**. Overview of research and conclusions; Vol. **2**. Radiological assessment; Vol. **3**. Geoscience characterization studies; Vol. **4**. Engineering; Vol. **5**. Dispersal of radionuclides in the oceans: models, data sets and regional descriptions; Vol. **6**. Deep-sea biology, biological processes and radiobiology; Vol. **7**. Review of laboratory investigations of radionuclide migration through deep-sea sediments, Vol. **8**. Review of processes near a buried waste canister. Paris: OECD, Nuclear Energy Agency, 66 pp; 325 pp; 318 pp; 236 pp; 462 pp; 219 pp; 273 pp; 136 pp.

NEA 1989. Interim oceanographic description of the North-east Atlantic site for the disposal of low-level radioactive waste, Volume **3**. Paris: OECD Nuclear Energy Agency, 374 pp.

NEA 1990. Co-ordinated Research and Environmental Surveillance Programme related to sea disposal of radioactive waste - activity report 1986-1990. Paris: OECD Nuclear Energy Agency, 152 pp.

NEA 1995a. The Environmental and Ethical Basis of Geologic Disposal. [A collective opinion of the NEA Radioactive Waste Management Committee]. Paris: OECD Nuclear Energy Agency, 30 pp.

NEA 1995b. Update on waste management policies and programmes. *OECD NEA Nuclear waste Bulletin*, No. **10**: 90 pp.

NERC 1995. Radioactive Waste Management. 4 pp. pers. comm. *Natural Environment Research Council*, November 1995.

NERC 1996. Scientific Group on Decommissioning Off-shore Structures: First report. *Report by the Natural Environmental Research Council for the Department of Trade and Industry (UK)*. 76 pp.

Nielsen, T.G., T. Kiorboe and P.K. Bjornsen 1990. Effects of a *Chrysochromulina polylepis* sub-surface bloom on the planktonic community. *Marine Ecology - Progress Series*, **62**: 21-35.

Normark, W.R., J.E. Lupton, J.W. Murray, J.R. Delaney, H.P. Johnson, R.A. Kloski, D.A. Clague and J.L. Morton 1982. Polymetallic sulfide deposits and water-column of active hydrothermal vents on the southern Juan de Fuca Ridge. *MTS Journal*, **16**(3): 46-53.

Norton, M.G., R.A. Eagle, R.S. Nunny, M.S. Rolfe, P.A. Hardiman and B.L. Hampson 1981. The field assessment of effects of dumping wastes at sea: 8. Sewage sludges dumping in the outer Thames estuary. *Fisheries Research Technical Reports. MAFF Directorate of Fishery Research*, Lowestoft, **62**: 1-62.

O'Brion, G.W. and H.H. Veeh 1980. Holocene phosphorite on the east Australian continental margin. *Nature*, **288**: 690-692.

O'Brion, G.W., J.R. Harris, A.R. Milnes and H.H. Veeh 1981. Bacerial origin of east Australian continental margin phosphorites. *Nature*, **292**: 442-444.

O'Brion, G.W. and H.H. Veeh 1983. Are phosphorites reiable indictors of upwelling? pp. 399-419 In: J. Thiede and E. Suess (eds), *Coastal upwelling, its sediment record, Part A*. New York, Plenum Press.

O'Brion, G.W., H.H. Veh, D.J. Cullen and A.R. Milnes 1986. Uranium-series isotopic studies of marine phosphorites and associated sediments from the east Australian continental margin. *Earth and Planetary Science Letters*, **80**: 19-35.

O'Reilly Wiese, S.B. J.M. Bubb and J.N. Lester 1995. The significance of sediment metal concentrations in two eroding Essex salt marshes. *Marine Pollution Bulletin*, **30**: 190-199.

Ocean Drilling Program Leg 106 Scientific Party 1986. Drilling the Snake Pit hydrothermal sulfide deposit on the mid-Atlantic Ridge, lat 23°22'N. *Geology*, **14**: 1004-1007.

Ockels, R. 1991. Innovative hydraulic engineering with geosynthetics. *Geosynthetic World*. October 1991, 26-27.

Ohsumi, T. 1996. CO_2 storage options in the deep sea. *MTS Journal*, **29**: 58-66.

Olsgard, F. and J.S. Gray 1995. A comprehensive analysis of the effects of offshore oil and gas exploration and production on the benthic communities of the Norwegian continental shelf. *Marine Ecology - Progress Series*, **122**: 277-306.

Ormerod, B. and M. Angel (eds) 1996. *Ocean storage of carbon dioxide: Workshop 2 - Environmental Impact.* IEA Greenhouse Gas R&D Programme, Cheltenham. 131 pp.

Oslo Commission 1989. *Review of Sewage Sludge Disposal at Sea*, Oslo Commission, London, 84 pp.

OSPARCOM 1993. *North Sea Quality Status Report 1993*. Oslo and Paris Commissions, London.

Oudin, E. and G. Constantinou 1984. Black smoker chimney fragments in Cyprus sulphide deposits. *Nature*, **308**(5957): 349-353.

Ozturgut, E. 1981. *Suspended particulate measurement techniques and accuracy and precision of chemical analyses used in the DOMES project*. Final Report to NOAA by Science Applications, Inc., Seattle, WA. 12 pp.

Pearce, K.W. and J.D. Vincent 1963. *Investigation into the effects of deep sea pressures on waste materials and disposal containers.* AERE-M1254. United Kingdom Atomic Energy Authority, Harwell.

Pearson, T.H. 1986. Disposal of sewage in dispersive and non-dispersive areas: Contrasting case histories in British coastal waters. pp. 577-595 In: G. Kullenberg (ed.) *Scientific Basis for the Role of the Oceans as a Waste Disposal Option.* D. Reidel, Dordrecht.

Phillips, D.J.H. and P.S. Rainbow 1990. *Biomonitoring of trace aquatic contaminants.* Elsevier Science Publishers, Barking U.K.

Phillips, D.J.H. 1995. The chemistries and environmental fates of trace metals and organochlorides in aquatic ecosystems. *Marine Pollution Bulletin*, **31**: 193-200.

Pimentel, D., M. Herdendorf, S. Eisenfeld, L. Olander, M. Carroquino, C. Corson, J. McDade, Y.Chung, W. Cannon, J. Roberts, L. Bluman and J. Gregg. 1994. Achieving a secure energy future: environmental and economic issues. *Ecological Economies,* **9**: 201-219.

Plüger, W.L., P.M. Herzig, K.P. Becker, G. Deissmann, D. Schöps, J. Lange, A. Jenisch, S. Ladage, H.H. Richnow, T. Schulze and W. Michaelis 1990. Discovery of hydrothermal fields at the Central Indian Ridge. *Marine Mining*, **9**(1): 73-86.

Pörtner, H.-O. and A. Reipschläger 1996. Ocean disposal of anthropogenic CO_2: Physiological effects on tolerant and intolerant animals. pp 57-81 In: B. Ormerod and M. Angel (eds) *Ocean Storage of CO_2 :Workshop 2. Environmental Impact.* IEA Greenhouse Gas R&D Programme, Cheltenham.

Pratt, R.M. and P.F. McFarlin 1966. Manganese pavements on the Blake Plateau. *Science*, **151**: 1080-1082.

Pryor, T.A. 1995. New described super-nodule resource. *Sea Technology*, **36**(9): 15, 17-18.

Quay, P.D., B. Tilbrook and C.S. Wong 1992. Oceanic uptake of fossil fuel CO_2: Carbon-13 evidence. *Science*, **256**: 74-79.

Rad, U. von and H.-R. Kudrass 1984. Phosphorite deposits on the Chatham Rise of New Zealand: results of a prospection cruise with R/V Sonne. *Geologisches Jahrbuch Reihe,* Hannover, BGR, **D 65**: 252.

Read, P.A. 1977. *An investigation on the influence of water quality on the intertidal macrobenthos of selected areas of the south shore of the outer Firth of Forth.* PhD thesis, Napier College of Commerce and Technology, Edinburgh

Rhein, M., L. Stramma and U. Send 1995. The Atlantic deep western boundary current: Water masses and transports near the equator. *J. Geophys. Res.*, **100**: 2441-2457.

Rice, A.L., D.S.M. Billett, J. Fry, A.W.G. John, R.S. Lampitt, R.F.C. Mantoura and R.J. Morris 1986. Seasonal deposition of phytodetritus to the deep-sea floor. *Proc. Roy. Soc. Edinburgh,* **88B**: 265-279.

Rice, A.L., M.H. Thurston, and B.J. Bett 1994. The IOSDL DEEPSEAS programme: introduction and photographic evidence for the presence and absence of a seasonal input of phytodetritus at contrasting abyssal sites in the Northeastern Atlantic. *Deep-Sea Research*, **41**: 1305-1320.

Riebesell, U., D.A. Wolf Gladrow, and V.S. Smetacek 1993. Carbon dioxide limitation of marine phytoplankton growth rates. *Nature, London*, **361**: 249-251.

Riggs, S.R. 1979. Petrology of the Tertiary phosphorite system of Florida. *Economic Geology*, **74**: 195-220.

Riggs, S.R. 1984. Paleoceanographic model of Meogene phosphorite deposition. U.S. Atlantic continental margin. *Science*, **223**: 123-131.

Riggs, S.R., A.C. Hine, S.W. Snyder, D.W. Lewis, M.D. Ellington and T.L. Stewart 1982. Phosphate exploration and resource potential on the North Carolina continental shelf. *Proc. Offshore Technol. Conf.*, Houston, Texas, May 3-6, 1982, Vol. **2** (OTC Paper 4295): 737-748.

Roberts, A.E. 1989. Geology and resources of Miocene coast ranges and Cenozoic offshore continental shelf phosphate deposits of California, USA. pp. 24-35 In: A.J.G. Notholt, R.P. Sheldon and D.F. Davidson (eds), *Phosphorite deposits of the world: Phosphate rock resources.* Cambridge, Cambridge University Press, **2**.

Robertson, A. (ed.) 1996. Offshore disposal: Results of the 106 Mile Dumpsite Study. Part 3: Biological fates and effects. *J. mar. Environm. Eng.*, **3** (2-4): 99-326.

Robertson, J.E., C. Robinson, D.R. Turner, P. Holligan, A. Watson, P. Boyd, E. Fernandez and M. Finch 1994. The impact of a coccolithophore bloom on ocean carbon uptake in the northeast Atlantic during summer 1991. *Deep-Sea Research I*, **41**: 297-314.

Roe, H.S.J. 1988. Midwater biomass profiles over the Madeira Abyssal Plain and the contribution of copepods. *Hydrobiologia*, **167/168**: 169-181.

Roe, H.S.J. et al. 1987. *Great Meteor East: a biological characterisation. [Ocean disposal of high level radioactive waste a research report prepared for the Department of the Environment].* Institute of Oceanographic Sciences Deacon Laboratory Report, **248**: 332 pp. and microfiche.

Roe, H.S.J., D.S.M. Billett and R.S. Lampitt 1990. Benthic/midwater interactions on the Madeira Abyssal Plain; evidence for biological transport pathways. *Progress in Oceanography*, **24**: 127-140.

Roels, O.A. 1974. Will nodule mining disturb the marine environment? *MTS Journal*, **8**(8): 17-20.

Roels, O.A., A.F. Amos, C. Garside, K.C. Haines, T.C. Malone, A.Z. Paul and G.E. Rice 1972. Environmental impact of two manganese nodule mining tests. pp. 129-146 In: *Manganese Nodule Deposits in the Pacific*, Symposium/Workshop Proc., Honolulu, Hawaii, 16 - 17 Oct. 1972. 220 pp.

Roels, O.A., A.F. Amos, O.R. Anderson, C. Garside, K.C. Haines, T.C. Malone, A.Z. Paul and G.E. Rice 1973. *The environmental impact of deep-sea mining, progress report.* NOAA Tech. Rep. ERL 290-OD 11. 185 pp.

Rona, P.A. 1975. Relation of offshore and onshore mineral resources to plate tectonics. *Proc., Offshore Technol. Conf.*, Vol. **2** (OTC Paper 2317): 713-716.

Rona, P.A. 1982. Polymetallic sulfides at seafloor spreading centers: a global overview. *MTS Journal*, **16**(3): 81-86.

Rona, P.A. 1983. Hydrothermal mineralization at seafloor spreading centers: Atlantic Ocean and Indian Ocean. *Proc., OCEANS '83,* San Francisco, California, 29 Aug. - 1 Sept. 1983, Vol. **2**: 817.

Rona, P.A. 1985. Hydrothermal mineralization at slow-spreading centers: Red Sea, Atlantic Ocean, and Indian Ocean. *Marine Mining*, **5**(2): 117-145.

Rona, P.A. 1992. Seafloor hydrothermal mineralization: a new discovery. Abstracts, *23rd Underwater Mining Institute*, Washington, DC, Sept. 27 - 29, 1992.

Rona, P.A., G. Klinkhammer, T.A. Nelsen, J.H. Trefry and H. Elderfield 1986. Black smokers, massive sulphides and vent biota at the Mid-Atlantic Ridge. *Nature*, **321**(6065): 33-37.

Roskill, S.W. 1961. *The War at Sea, Volume III- The Offensive, Part II.* HMSO, London, 479 pp.

Rowe, G.T. and V. Pariente (eds) 1992. *Deep-Sea Food Chains and the Global Carbon Cycle.* NATO ASI Series, Kluwer Academic Publishers, Dordrecht. 400 pp.

Rowe, G.T., N.R. Merrett, J. Shepherd, G. Needler, B. Hargrave and M. Marietta 1986. Estimates of direct biological transport of radioactive waste in the deep sea with special reference to organic carbon budgets. *Oceanologica Acta*, **9**: 199-208.

Rowland, T.J. 1985. Non-energy marine mineral resources of the world oceans. *MTS Journal*, **19**(4): 6-17.

Ruttenberg, K.C. and R.A. Berner 1993. Authigeneic apatite formation and burial in sediments from non-upwelling, continental margin environments. *Geochimica Cosmochimica Acta*, **57**: 991-1007.

Santoro, E.D. and T.J. Fikslin 1987. Chemical and toxicological characteristics of sewage sludge ocean dumped in the New York Bight. *Marine Pollution Bulletin*, **18**: 394-399.

Sarmiento, J.L., J.C. Orr and U. Siegenthaler 1992. A perturbation simulation of CO_2 uptake in an ocean general circulation model. *Journal of Geophysical Research* , **97**(C3): 2621-3645.

Saunders, P. 1992. *Managing Radioactive Waste in the European Community.* Brussels: CEC, 40 pp.

Sayles, F.L., S.P. Smith and J.E. Goudreau 1996. Deep ocean sludge disposal, sediment oxygen consumption and sediment redox profiles at deep water municipal dump site 106. *J. Mar. Environm. Eng.*, **3** (2-4): 99-130.

Schlitzer, R., W. Roether, H. Oster, H.G. Junghans, M. Hausmann, H. Johannsen and A. Michelato 1991. Chlorofluoromethane and oxygen in the Eastern Mediterranean.. *Deep-Sea Research,* **38**: 1531-1551.

Schoell, M. and M. Hartmann 1978. Changing hydrothermal activity in the Atlantis II Deep geothermal system. *Nature*, **274** (5673): 784-785.

Schriever, G. 1995. DISCOL - disturbance and recolonization experiment of a manganese nodule area of the southeastern Pacific. *Proc. ISOPE Ocean Mining Symposium,* Nov. 21-22, 1995, Tsukuba, Japan: 163-166.

Schriever, G. and A. Ahnert 1997. Results of the large scale deep-sea environmental impact study DISCOL during eight years of investigation. *Proc. Seventh Intern. Offshore Polar Engineer. Conf.*, Honululu. **1**: 438-444.

Schriever, G., H. Bluhm, A. Boetius, C. Borowski, C. Bussau and H. Thiel 1992. DISCOL - precautionary large-scale environmental impact studies for future polymetallic nodule mining from the deep sea. *Proc. XV Congreso Mundial de Mineria*, Madrid, Spain: 1311-1319.

Schriever, G., C. Bussau and H. Thiel 1991. DISCOL - precautionary environmental impact studies for future manganese nodule mining and first results on meiofauna abundance. *Proc. Adv. Mar. Technol. Conf.*, Tokyo, Japan, **4**: 47-57.

Scott, S.D. 1983. Basalt and sedimentary hosted seafloor polymetallic sulfide deposits and their ancient analogues. *Proc., OCEANS '83*, San Francisco, California, 29 Aug. - 1 Sept. 1983, **2**: 818-824.

Scott, S.D. 1985. Seafloor polymetallic sulfide deposits: modern and ancient. *Marine Mining*, **5**(2): 191-212.

Scott, S.D. 1992a. Polymetallic sulfide riches from the deep: fact or fallacy? pp. 87-115 In: K.J. Hsu and J. Thiede (eds) *Use and Misuse of the Seafloor.* John Wiley and Sons, Ltd.

Scott, S.D. 1992b. Environmental considerations for the future mining of polymetallic sulfides on the deep ocean floor. Abstract, *23rd Underwater Mining Institute*, Washington, DC, Sept. 27 - 29, 1992.

Scott, S.D. 1995. Recovery of deep seabed minerals from Ocean Cities. *Proc. "Ocean Cities '95"*, UNESCO, Monaco.

Scott, S.D., R.A. Binns and PACMANUS Expedition participants 1992. PACMANUS: an actively-forming submarine polymetallic sulfide deposit in felsic volcanic rocks of the Manus back-arc, Papua, New Guinea. Abstract, *23rd Underwater Mining Institute*, Washington, DC, Sept. 27 - 29, 1992.

SEBA 1995. Oslo and Paris Conventions for the prevention of marine pollution: Working group on sea-bed activities (SEBA). *Summ. Rec. SEBA* 95/8/1-E.

SEBA 1996. Oslo and Paris Conventions for the prevention of marine pollution: Working groups in sea-bed activities (SEBA). *Summ. Rec. SEBA* 96/17/1-E.

Segschneider, J. and J. Sündermann 1997. Large scale transport of particle reactive tracers - numerical simulations. *Proc. Seventh Intern. Offshore Polar Engineer. Conf.*, Honululu. **1**: 461-467.

Seibold, E. 1978. Deep sea manganese nodules. The challenge since "Challenger". *Episodes*, **4**: 3-8.

Seki, S., A. Ito and H. Amano 1980. Integrity test of full size packages of cement - solidified radioactive wastes under deep sea conditions. *Nuclear and Chemical Waste Management*, **1**: 129-138.

Sheldon, R.P. 1980. Episodicity of phosphate deposition and deep ocean circulation - a hypothesis. pp. 239-247 In: Y.K. Bentor (ed.), Marine phosphorites - geochemistry, occurrence, genesis, SEPM Special Publication, Tulsa. **29**.

Sibuet, M. and D. Coic 1989. Photograhic prospection of the NEA dumpsite in the northeast Atlantic: quantitative distribution of epibenthic megafauna. pp. 167-177 In: *Interim oceanographic description of the North-east Atlantic site for the disposal of low-level radioactive waste*, Volume **3**. Paris: OECD Nuclear Energy Agency, 374 pp.

Sibuet, M., D. Calmet and G. Auffret 1985. Reconnaissance photographique de conteneurs en place dans la zone d,immersion des dechets faiblement radioactifs de l'Atlantique Nord-Est. *Comptes Rendus de l'Academie des Sciences*, Paris, **301**: 497-502.

Smethie, W.M. 1993. Tracing the thermohaline circulation in the western North Atlantic using chlorofluorocarbons. *Progress in Oceanography*, **31**: 51-99.

Smith, C.R. 1985. Colonization studies in the deep sea: Are results biased by experimental designs? P.E. Gibbs (ed.), *Proceed. Nineteenth Europ. Mar. Biol. Symp.*, Cambridge Univ. Press, 183-190.

Smith, C.R., P.A. Jumars and D.J. Demaster. 1986. In situ studies of megafaunal mounds indicate rapid sediment turnover and community response at the deep-sea floor, *Nature* **6085**: 251-253.

Smith, C.R., H. Kukert, R.A. Wheatcroft, P.A. Jumars and J.W. Deming 1989. Vent fauna on whale remains. *Nature*, **341**(6237): 27-28.

Smith, C.R., D.J. Hoover, S.E. Doan, R.H. Pope, D.J. Demaster, F.C. Dobbs and M.A. Altabet 1996. Phytodetritus at the abyssal seafloor across 10° of latitude in the central equatorial Pacific. *Deep-Sea Res.*, **43**: 1309-1338.

Smith, J.R., B.R. Holt and R.G. Paul 1985, Current status of leasing proposals for the U.S. Exclusive Economic Zone. *Proc. Offshore Technol. Conf.*, Vol. **2** (OTC Paper 4899): 9-18.

Smith, K.L., R.S. Kaufmann and R.J. Baldwin 1994. Coupling the near-bottom pelagic and benthic processes at abyssal depths in the eastern North Pacific. *Limnol.Oceanogr.*, **39**: 1101-1118.

Snelgrove, P.V.R., J. Grassle and R.F. Petrecca 1996. Experimental evidence for aging food patches as a factor contributing to high deep-sea macrofaunal diversity. *Limnol. Oceanogr.*, **41**: 605-614.

SOAEFD 1996. Survey of the Beaufort's Dyke Explosives disposal site - November 1995: Interim report. *Fisheries Research Services Report*, **1/96**.

Sorem, R.K. and R.H. Fewkes 1979. *Manganese nodules - research data and methods of investigation.* IFI/Plenum Press, New York, NY. 723 pp.

Spencer, C.P. 1975. The micronutrient elements. pp. 245-300 In: J.P. Riley and G. Skirrow (eds), *Chemical Oceanography.* Academic Press, London, **2**: 245-300.

Spencer, D.W. 1991. *An Abyssal ocean option for waste management.* Report of a workshop held at The Woods Hole Oceanographic Institution, January 1991. 111 pp.

Spiess, F.N., K.C. Macdonald, T. Atwater, R. Ballard, A. Carranza, D. Cordoba, C. Cox, V.M. Diaz Garcia, J. Francheteau, J. Guerrero, J. Hawkins, R. Haymon, R. Hessler, T. Juteau, M. Kastner, R. Larson, B. Luyendyk, J.D. Macdougall, S. Miller, W. Normark, J. Orcutt and C. Rangin 1980. East Pacific Rise: hot springs and geophysical experiments. *Science,* **207**(4438): 1421-1433.

Stackelberg, U. von, 1996. Growth history of manganese nodules and crusts of the Peru Basin. *Geol. Soc. Spec. Publ.*, **119**: 153-176.

Stackelberg, U. von, and shipboard scientific party 1985. Hydrothermal sulfide deposits in back-arc spreading centers in the southwest Pacific. *BGR, Hannover, Circular*, **2**: 3-14.

Stegen, G., K. Cole and R. Bacasow 1995. Biochemical impacts of CO_2 storage in the ocean. *Energy Conserv. Managem.*, **36**: 497-500.

Suess, E. 1981. Phosphate regeneration from sediments of the Peru continental margin by dissolution of fish debris. *Geochimica Cosmochimica Acta*, **45**: 577-588.

Summerhayes, C.P., A.H. Nutter and J.S. Tooms 1972. The distribution and origin of phosphate in sediments off northwest Africa. *Sedimentary Geology*, **8**: 3-28.

Summerhayes, C.P. and J.M. McArthur 1990. Moroccan offshore phosphorite deposits. pp. 159-166 In: W.C. Burnett and S.R. Riggs, *Phosphate deposits of the world. Genesis of Neogene to Recent phosphorites.* Cambridge University Press, Cambridge, **3**.

Swallow, J.C. and J. Crease 1965. Hot salty water at the bottom of the Red Sea. *Nature*, **205**(4967): 165-166.

Takada, H., J.W. Farrington, M.H. Bothner, C.G. Johnson and B. Tripp 1994. Transport of sludge-derived organic pollutants to deep-sea sediments at Deep Water Dump Site 106. *Environ. Sci. Technol.*, **28**: 10.

Takizawa, M., W.L. Straube, R.T. Hill and R.R. Colwell 1993. Near-bottom pelagic bacteria at a deep-water sewage sludge disposal site. *Applied and Environmental Microbiology*, **59** (10): 3406-3410.

Tengberg, A., F. De Bovee, P. Hall, W. Berelson, D. Chadwick, G. Ciceri, P. Crassous, A. Devol, S. Emerson, J. Gage, R. Glud, F. Graziottini, J. Gundersen, D. Hammond, W. Helder, O. Holby, R. Jahnke, A. Khripounoff, S. Lieberman, V. Nuppenau, O. Pfannkuche, C. Reimers, G. Rowe, A. Sahami, F. Sayles, M. Schurter, D. Smallman, B. Wehrli and P. De Wilde 1995. Benthic chamber and profiling landers in oceanography - A review of design, technical solutions and functioning. *Progress in Oceanography*, **35**: 253-294.

Theobald N. and N.-P. Rühl 1994. Chemische Kampfstoffmunition in der Ostsee. *Deutsche Hydrographische Zeitschrift*, Supplement **1**: 121-131.

Thiel, H. 1972. Meiofauna und Struktur der benthischen Lebensgemeinschaft des Iberischen Tiefseebeckens. *"Meteor" Forschungsergebnisse* D, **12**: 36-51.

Thiel, H. 1979. First quantitative data on Red Sea deep benthos. *Marine Ecology Progress Series*, **1**: 347-350.

Thiel, H. 1983. Meiobenthos and nanobenthos of the deep sea. pp. 167-230 In: G.T. Rowe (ed.)*The Sea*, Wiley-Interscience, New York, vol. **8**.

Thiel, H. 1991. From MESEDA to DISCOL: a new approach to deep-sea mining risk assessment. *Marine Mining*, **10**(4): 369-386.

Thiel, H. 1992. Deep-sea environmental disturbance and recovery potential. *Intern. Rev. ges. Hydrobiologie*, **77**(2): 331-339.

Thiel, H. and G. Schriever 1990. Deep-sea mining, environmental impact and the DISCOL project. *AMBIO*, **19**(5): 245-250.

Thiel, H. and E.J. Foell 1993. Environmental risk assessment for manganese nodule mining and application of the precautionary principle. pp. 226-233 In A. Couper and E. Gold (eds) *The Marine Environment and Sustainable Development - Law, Policy and Science*. Law of the Sea Institute, Honolulu, Hawaii. 688 pp.

Thiel, H., H. Weikert and L. Karbe 1985. Abschätzung der Umweltrisiken durch Abbau von Erzschlämmen aus dem Atlantis-II-Tief im Roten Meer. *Natur und Museum*, **115**(4): 98-110.

Thiel, H., H. Weikert and L. Karbe 1986. Risk assessment for mining metalliferous muds in the deep Red Sea. *AMBIO*, **15**(1): 34-41.

Thiel, H., O. Pfannkuche, G. Schriever, K. Lochte, A.J. Gooday, Ch. Hemleben, R.F.G. Mantoura, C.M. Turley, J.W. Patching and F. Riemann 1989. Phytodetritus on the deep-sea floor in a central oceanic region of the North-east Atlantic. *Biological Oceanography*, **6**: 203-239.

Thiel, H., E.J. Foell and G. Schriever 1991. Potential environmental effects of deep seabed mining. *Ber. Zentrum Meeres. u. Klimaforsch. Univ. Hamburg*, **26**: 1-243.

Thiel, H., H. Bluhm, C. Borowski, C. Bussau, A.J. Gooday, C. Maybury and G. Schriever 1992. The DISCOL Project (the impact of mining on deep-sea organisms). *Ocean Challenge*, **3**(1): 40-46.

Thiel, H., G. Schriever, C. Bussau and C. Borowski 1993. Manganese nodule crevice fauna. *Deep-Sea Research*, **40**(2): 419-423.

Thiel, H. and Forschungsverbund TUSCH 1995. The German environmental impact research for manganese nodule mining in the SE Pacific Ocean. *Proc. ISOPE Ocean Mining Symposium*, Nov. 21-22, 1995, Tsukuba, Japan, 39-45.

Thomson, J., S.E. Calvert, S. Mukherjee, W.C. Burnett and J.M. Bremner 1984. Further studies of the nature, composition andages of contemporary phosphorite from the Namibian shelf. *Earth Planetary Science Letters*, **69**: 341-353.

Thurston, M.H. 1979. Scavenging abyssal amphipods from the North-east Atlantic Ocean. *Marine Biology*, **51**: 55-68.

Tivey, M.K. and J.R. Delaney 1985. Sulfide deposits from the Endeavour Segment of the Juan de Fuca Ridge. *Marine Mining*, **5**(2): 165-179.

Tornes, J.A., J.H. Blanch, T.I. Wedervang, A.G. Andersen and A.M. Opstad 1989. *Undersokelse av skipsvrak inneholdende kjemisk ammunisjon senket i norske farvann eter annen verdenskrig*. Forsvarets Forskningsinstitutt Rapport 89/6007, Kjeller, Norway.

Toth, J.R. 1980. Deposition of submarine crusts rich in manganese and iron. *Geological Society of America Bulletin*, **91**: 44-54.

Toth, J.R. and C.A. Amerigian 1986. Development of an advanced sampling device for the investigation of marine ferromanganese crust deposits. *Proc. Offshore Technol. Conf.*, Vol. **3** (OTC Paper 5235): 127-134.

Tømes J.A., J.H. Blanch, T.I. Wedervang, A.G. Andersen and A.M. Opstad 1989. Undersøkelse av skipsvrak inneholdende kjemisk ammunisjon seuket; norske farvann etter annen Verdenskrig. Fors varets Forskningsinstitutt, Norwegian Defence Research Establishment, *FFI/Rapport*-89/6007: 24 pp.

Trueblood, D.D. 1993. US Cruise Report for BIE II, Cruise 1, July 30 - September 9, 1993, R/V Yuzhmorgeologiya. *NOAA Tech Memo NOS OCRM*, **4**: 51 pp.

Trueblood, D.D. and E. Ozturgut 1992. The Benthic Impact Experiment. Abstract, *23rd Underwater Mining Institute*, Washington, DC, Sept. 27 - 29, 1992.

Trueblood, D.D. and E. Ozturgut 1997. The benthic impact experiment: A study of the ecological impacts of deep seabed mining on abyssal benthic communities. *Proc. Seventh Intern. Offshore Polar Engineer. Conf.*, Honululu. **1**: 481-487.

Türkey, M. 1986. Crustacea. Decapoda reptantia der Tiefsee des Roten Meeres. *Senckenbergiana marit.*, **18**: 123-185.

Türkey, M. 1996. Composition of the deep Red Sea macro- and megabenthic inverbrate fauna. Zoogeographic and ecological implications. pp. 43-59 In: F. Uiblein, J. Ott and M. Stachowitsch (eds.) *Deep-sea and extreme shallow-water habitats: affinities and adaptations. Biosystematics and Ecology Series,* **11**.

Tunnicliffe, V. 1991. The biology of hydrothermal vents: ecology and evolution. *Oceanogr. Mar. Biol. Ann. Rev.*, **29**: 319-407.

Tunnicliffe, V., M. Botros, M.E. de Burgh, A. Dinet, H.P. Johnson, S.K. Juniper and R.E. McDuff 1986. Hydrothermal vents of Explorer Ridge, northeast Pacific. *Deep-Sea Research*, **33**: 401-412.

Uchida, T., T. Hondo, S. Mae and J. Kawabata 1995. Physical data of CO_2 hydrate. pp. 45-62 In N. Handa and T. Ohsumi (eds) *Direct Ocean Disposal of Carbon Dioxide*, Terra Scientific Publishing Co, Tokyo.

Uchupi, E., M.T. Muck and R.D. Ballard 1988. The geology of the Titanic site and the vicinity. *Deep-Sea Research*, **35**: 1093-1110.

UKOOA 1996. UKOOA 17th offshore round: Environmental Screening Reports. 3: South-west approaches, 4: Rockall Trough south, 5: Rockall Trough north.

Usher, J.D., J.P. Latimer and D.C. Tolefson 1987. *Study of deep seabed mining selective minesite considerations for commercial recovery of manganese nodules from the ocean floor.* Final Report to NOAA/OME under Contract No. 50-DGNC-6-00079. 45 pp. plus appendices.

Valent, P.J. and D.K. Young 1995. *Abyssal seafloor waste isolation: Environmental report.* Naval Research Laboratory, Stennis Space Center, NRL/MR/7401-95-7576. 479 pp.

Valent, P.J., D.K. Young and A.W. Palowitch 1997. Simulation of deep ocean relocation of dredged material. *Proc. Seventh Intern. Offshore Polar Engineer. Conf.*, Honululu. **1**: 505-509.

Van Dover, C.L. 1995. Ecology of Mid-Atlantic Ridge hydrothermal vents. pp. 257-294 In: L.M. Parson, C.L. Walker, D.R. Dixon (eds) *Hydrothermal vents and processes. Geol. Soc. Spec. Publ.*, **87**.

Van Dover, C.L., J.F. Grassle, B. Fry, R.H. Garritt and V.R. Starczak 1992. Stable isotopic evidence for entry of sewage-derived organic material into a deep-sea food web. *Nature* **360**: 153-156.

Veeh, H.H., W.C. Burnett and A. Soutar 1973. Contemporary phosphorites on the continental margin of Peru. *Science*, **181**: 844-845.

Veeh, H.H. S.E. Calvert and N.B. Price 1974. Accumulation of uranium in sediments and phosphorites of the south African shelf. *Marine Chemistry*, **2**: 189-202.

Vinogradov, M., A. Sagalevitch and S.V. Khetagurov 1996. Oceanographic research and technical operations on the site of nuclear submarine Komsomolets wreck. *Nauka, Moscow.* 360 pp. (in Russian).

Wasmann, P., H. Svendsen, A. Keck and H. Reigstad 1996. Selected aspects of the physical oceanography and particles fluxes in fjords of Northern Norway. *Journal of Marine Systems*, **8**: 52-71.

Weaset, R.C. and S.M. Selby (eds) 1967. *CRC Handbook of Chemistry and Physics*. The Chemical Rubber Co., Cleveland, Ohio.

Weaver, P.P.E., D.G. Masson, D.E. Gunn, R.B. Kidd, R.G. Rothwell and D.A. Maddison 1995. Sediment mass wasting in the Canary Basin. pp. 287-296 In: K.T. Pickering, R.N. Hiscott and N.H. Kenyon (eds) *Atlas of deep water environments: architectural style in turbidite sediments*. Chapman and Hall, London. 333 pp.

Weikert, H. 1982. The vertical distribution of zooplankton in relation to habitat zones in the area of the Atlantis II Deep, Central Red Sea. *Mar. Ecol. Prog. Ser.*, **8**: 129-143.

Weis, J.S. and P. Weis 1981. Effects of environmental pollutants on early fish development. *Reviews of Aquatic Science*, **1**: 45-73.

Welling, C.G. 1981. An advanced design deep sea mining system. *Proc., Offshore Technol. Conf.*, Houston, Texas, 4 - 7 May 1981, Vol. **3** (OTC Paper 4094): 247-255.

Welling, C.G. 1982. Polymetallic sulfides: an industry viewpoint. *MTS Journal*, **16**(3): 5-7.

Wiedicke, M. and M.E. Weber 1996. Small-scale variability of seafloor features in the northern Peru Basin: results from acoustic survey methods. *Mar. Geophys. Res.*, **18**: 507- 526.

Winn, C.D., D.M. Karl and G.J. Massoth 1986. Microorganisms in deep-sea hydrothermal plumes. *Nature*, **320**(6064): 744-746.

Wong, C.S. and S. Hirai 1997. Ocean storage of carbon dioxide. A review of oceanic carbonate and CO_2 hydrate chemistry. *IEA Greenhouse Gas R & D Programme*, Chaltenam, 90 pp.

Woods Hole Policy Center 1993. *Optimal strategies for waste management: Economic perspectives on the ocean option.* Report from Woods Hole Oceanographic Institution, Woods Hole, MA. 10 pp.

Woolsey, J.R. 1992. Offshore minerals for flue gas desulfurization and by-product strategic metals and superphosphate fertilizer. Abstract, *23rd Underwater Mining Institute*, Washington, DC, Sept. 27 - 29, 1992.

Yablokov, A.V., V.K. Karasov, V.M. Rumyantsev, M.Y. Kokeyev, O.I. Petrov, V.N. Lystov, A.F. Yemelyanenkov and P.M. Rubtsov 1993. *Matters related to the disposal of radioactive wastes at sea.* Office of the President of the Russian Federation. 80 pp.

Zaiger, K.K. 1994. Potential marine mining by *in situ* leaching and recovery of metals from cobalt-rich ferromanganese ocean crust. *MTS 94 Conf. Proc.,* Mar. Technol. Soc., Washington, DC: 226-232.

Zielke, W., J.A. Jankowski, J. Sündermann and J. Segschneider 1995. Numerical modeling of sediment transport caused by deep-sea mining. *Proc. of the ISOPE-Ocean Mining Symposium*, Tsukuba, Japan: 157-162.

European Commission

Marine science and technology — Environmental risks from large-scale ecological research in the deep sea: a desk study

Luxembourg: Office for Official Publications of the European Communities

1998 — XIV, 210 pp. — 21 x 29.7 cm

ISBN 92-828-3517-0

Price (excluding VAT) in Luxembourg: ECU 8

14

15

CG-12-98-489-EN-C

NOTICE TO THE READER

Information on European Commission publications in the areas of research and innovation can be obtained from:

- **CORDIS, the Community R & D Information Service**

  For more information, contact:
  CORDIS Customer Service, BP 2373, L-1023 Luxembourg
  Tel. (352) 44 10 12-2240; fax (352) 44 10 12-2248; e-mail: helpdesk@cordis.lu
  or visit the website at http://www.cordis.lu/

- **Euroabstracts**

  The European Commission's periodical on research publications, issued every two months.
  For more information, contact:
  RTD help desk, European Commission, DG XIII, L-2920 Luxembourg
  Fax (352) 43 01-32084; e-mail: rtd-helpdesk@lux.dg13.cec.be

Price (excluding VAT) in Luxembourg: ECU 8

ISBN 92-828-3517-0

9 789282 835173

OFFICE FOR OFFICIAL PUBLICATIONS
OF THE EUROPEAN COMMUNITIES

L-2985 Luxembourg